Mississippi Valley Division,
Regional Planning and Environment Division South

Pearl River Basin, Mississippi, Federal Flood Risk Management Project

Appendix D - Endangered Species Act Coordination

June 2024

Pearl River Basin, Mississippi
Federal Flood Risk
Management Project
Hinds and Rankin Counties, MS
THREATENED and ENDANGERED SPECIES
REVISED BIOLOGICAL ASSESSMENT

June 2024

The purpose of this Biological Assessment (BA) is to assess the effects of the Pearl River Flood Risk Management project and determine whether the project may affect any Federally threatened, endangered, proposed or candidate species. This BA is being prepared in accordance with legal requirements set forth under Section 7 (a) 2 of the Endangered Species Act (16 U.S.C. 1536 (c)).

Table of Contents

Annexes

Annex D1 Prior Consultation

2019 Threatened and Endangered Species Biological Opinion

Annex D2 IPaC report and USFWS email confirmation

Annex D3 Species Recovery Plans and Status Assessment Reports

Gulf Sturgeon Recovery/Management Plan

Ringed Sawback Turtle Recovery Plan

Species Status Assessment Report for the Northern long-eared bat

Species Status Assessment Report for the Pearl River Map Turtle

Species Status Assessment Report for the Alligator Snapping Turtle

Species Status Assessment Report for the Louisiana pigtoe

Species Status Assessment Report for the tricolor bat

Species Status Assessment Report for the monarch butterfly

Annex D4 Project Descriptions

1 Description of The Action

1.1 Project Name

Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, MS (PR FRM).

1.2 Introduction

A Biological Assessment (BA) completed in 2019 as a part of the environmental review process for the Pearl River Basin, Mississippi Federal Flood Risk Management Project, Hinds and Rankin Counties, MS Draft Feasibility Study/Environmental Impact Statement (FS/EIS) included a review of literature and other pertinent scientific data, interviews, and coordination efforts for each of the mentioned threatened and endangered (T&E) species as originally identified by the U.S. Fish and Wildlife Service (The Service). The 2019 BA and Biological Opinion (BO) are incorporated by reference into this BA and were used during the preparation of this document.

A search on The Services' Information for Planning and Consulting (IPaC) site, conducted on March 21, 2023, resulted in a list of species that should be considered when assessing the impacts of this project. That list includes the Gulf sturgeon, ringed sawback (Ringed map) turtle, Northern long-eared bat, Pearl River map turtle, alligator snapping turtle, and monarch butterfly. Email correspondence with The Service dated March 21, 2023, confirmed this list, and concluded that the monarch butterfly, as a candidate species, has no legal regulations under the Endangered Species Act. However, On April 21, 2023, email correspondence with the Service stated that they had been informed that a listing decision on the monarch butterfly would be made very soon. Therefore, the USACE has decided to include the monarch butterfly in this BA. On April 10, 2023, the Service informed USACE via email that the Louisiana pigtoe and the tricolored bat had been recently proposed for listing. Therefore, those two species will also be discussed in this BA. Additionally, other protected species, specifically the bald eagle and migratory birds, are discussed to obtain compliance with the Bald and Golden Eagle Protection Act and the Migratory Bird Treaty Act.

The Service requested velocity and sedimentation analysis be conducted within the Pearl River at the project site and downstream. USACE has committed to conducting this analysis during pre-construction engineering and design (PED) if a weir is included in the selected alternative for implementation. That being said, re-initiation of ESA consultation will be necessary during PED to accurately assess impacts to riverine species under consideration.

1.3 Project Description

Alternative A1, a non-structural alternative, consists of elevating and floodproofing residential and non-residential structures within the future 100-year stage.

Alternative C consists of the construction of channel improvements, demolition of the exiting weir near the J. H. Fewell WTP site, and construction of a new weir with a low-flow gate structure further downstream to ensure water supply while simultaneously creating an area of surface water, e.g., lake, for recreational opportunities, Federal levee improvements (excavated material plan), and upgrading an existing non-Federal ring levee with a slurry wall around the Savannah Street WWTP. Construction of the project would require relocations and/or improvements to various public and private utilities and infrastructure, mitigation of potential HTRW and other hazardous waste sites within the floodplain, avoidance and minimization measures required under the ESA, and the creation of new habitat mitigation areas to offset losses within the project's construction footprint areas under the FWCA. The project also has the potential to impact historic properties.

The Alternative CTO (Combined There Of) provides similar flood risk reduction as the NFI Alternative C with a smaller footprint. It combines some non-structural features from Alternative A1, the construction of channel improvements, a new weir with a low-flow gate structure downstream for future potential water supply while simultaneously creating a lake area for recreational opportunities. Federal levee improvements are added to include an excavated material plan and raising an existing non-Federal ring levee (the Savannah Street WWTP Levee). A levee segment of approximately 1.5 miles is proposed on the west bank of the Pearl River in northeast Jackson near Canton Club Circle.

Modifications also include construction of a weir upstream of the location identified for Alternative C, reducing excavation limits that reduces fill areas and thus environmental impacts throughout the project footprint. The new weir would have a lower elevation than proposed for alternative C as well as a reduction in the overbank excavation limits.

See Annex D4 for details of each alternative.

1.3.1 Location

The project is located in portions of Hinds and Rankin Counties, Mississippi within what is referred to as the City of Jackson Metropolitan Area. The project area begins at river mile (RM) 293.5 and extends southward on either side of the Pearl River channel, to a point approximately 3.0 miles south of U.S. Interstate 20 to RM 284.0.

1.3.2 Description of project habitat

The Alternative C Project area consists of numerous habitat types to include the following: Emergent wetlands, lacustrine, mixed forested wetlands, mixed scrub-shrub wetlands, riverine, upland evergreen forest, upland grassland, upland mixed forest, upland pasture, and upland scrub-shrub. It is assumed that the habitats within the CTO would be the same as for Alternative C since it falls within the same footprint. The Canton Club levee is outside of the Alternative C footprint, but habitat has been identified as BLH.

1.3.3 Project proponent information

Requesting Agency

DEPT OF DEFENSE (DOD)

Army Corps of Engineers (COE)

Tammy Gilmore
7400 Leake Ave
New Orleans, LA 70118

504-862-1002

tammy.f.gilmore@usace.army.mil

Lead agency

Same as Requesting Agency.

1.3.4 Project purpose

The primary purposes of the PR FRM Project are to reduce flood risk in the Jackson metropolitan area; reduce the flood risk to critical infrastructure, including the Savanna Street Wastewater Treatment Facility; and improve access to transportation routes, evacuation routes, and critical care facilities during flood events.

1.3.5 Project type and deconstruction

Currently the project is at approximately a 20% level of design such that many specifics for many of the components below have not yet been determined.

<u>Project timeline and sequencing</u> - TBD

<u>Site preparation</u> – Prior to construction

<u>Construction access and staging</u> – Prior to, concurrent with, and post construction

<u>Post-project site restoration</u> – Post construction

Conservation and compensation activities (both on- and off-site) – Prior to, concurrent with, and post construction.

1.3.6 Anticipated environmental stressors

1.3.6.1 Animal Features

Alternative A-1

Existing terrestrial wildlife habitat and the wildlife resources within the immediate area would be directly impacted due to construction activities and associated noise. It is anticipated that the areas where there are currently existing structures would self-vegetate once the structures are removed. This newly developed habitat would support some terrestrial species.

Alternative C

Existing terrestrial wildlife habitat and wildlife resources within the Project Area would be directly impacted by the removal of forested wetlands and other terrestrial habitat that currently exists within the project area. Though the existing terrestrial habitats would be removed, aquatic habitats that replace them would be utilized by other wildlife species.

Impacts to aquatic and fisheries resources associated with sedimentation during construction poses a risk. Best Management Practices would be implemented to reduce this risk, but potential sedimentation could adversely affect food sources for aquatic species. This impact, however, would be temporary; and it is anticipated that overall available aquatic and fisheries habitat would increase as a result of the channel improvements, with the total area available for aquatic and fish habitat estimated at 2,562 acres, post-construction. However, approximately 287 acres of the current riverine system would be replaced with a far larger lake system.

Under this alternative, in spite of increased water surface, riverine obligates (fish, mussels, turtles, etc.)will not benefit because they will not use lakes or shorelines modified for recreation and could not survive under such conditions except in minimal areas where riverine flows are sustained.

Disturbance from excavation and placement of material from within and adjacent to the river over approximately two years could also result in death of individuals if they are unable to flee the construction work area. This is especially relevant to young of year species and freshwater mussels.

Compensation and mitigation measures, including habitat restoration activities, would be implemented to offset the intensity of these impacts during and after construction . A fish passage would be created around the relocated weir, which would increase the possibility for migrating aquatic species to utilize the Project Area provided connectivity to flowing waters is sustained.

CTO Alternative

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of CTO with a weir would likely eliminate riverine habitat that many aquatic and riverine species depend on. For this draft, a conservative approach is being taken, and the IMT is assuming the CTO with a weir would convert the riverine system within the project area to a lake-like system. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system. That being said, forested wetlands and other terrestrial habitat would be converted to a lacustrine habitat type.

Alternative CTO without a weir

Impacts to terrestrial wildlife would be the same as with a weir. There would be no conversion of riverine habitat and so wildlife and fisheries dependent on aquatic systems would not be impacted.

1.3.6.2 Aquatic Features

Alternative A-1

There would be no impacts to aquatic features due to implementation of Alternative A-1.

Alternative C

Existing surface water bodies within the channel improvement footprint, including the Pearl River channel itself and its tributaries, would be impacted by this alternative.

Indirect impacts to adjacent waterbodies within the project area could also be anticipated through the implementation of Alternative C. Existing interconnections to adjoining waterbodies could be affected and existing inflow and outflow functions within the areas could also be affected.

An approximate 2,562-acre lake would be created post construction. This would increase the available aquatic features within the project area. However, approximately 287 acres of the current riverine system would be replaced by this lake system. Riverine obligates (fish, mussels, turtles, etc.) will not use lakes or could not survive under such conditions.

Alternative CTO

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that many aquatic species depend on. For this draft, a conservative approach is being taken, and the IMT is assuming the CTO with a weir would convert approximately 232 acres of the riverine system within the project area to a lake-like system. An approximate 1,706-acre lake would be created post construction. That being said, the impacts to aquatic features would be similar to those of alternative C except to a lesser degree.

Alternative CTO without a weir

Impacts on aquatic features would be reduced as the portion of the Pearl River within the project footprint would not be converted to a lake like system.

1.3.6.3 Environmental Quality Features

Alternative A-1

There would be no impacts to water quality with the implementation of Alternative A-1. Construction activities would increase suspended particles (dust) into the air during construction. This would cause temporary and minimal impacts to air quality.

Alternative C

This alternative could potentially result in direct, indirect, and cumulative impacts to water quality. These impacts would be temporary increases in turbidity and suspended solids in adjacent water bodies – the Pearl River and tributaries. The impacts to water quality due to this alternative are inconclusive due to the lack of data, modeling inaccuracies, and the usage of outdated modeling methodologies of the project area. There could be changes to temperature, dissolved oxygen (DO), ultimate carbonaceous biochemical oxygen demand (CBODU), total nitrogen (TN), ammonia-nitrogen (NH3-N), nitrate-nitrite (NOX), organic nitrogen (Org-N), total phosphorus (TP), orthophosphate (PO4), organic phosphorus (Org-P), phytoplankton chlorophyll-a, and total suspended solids (TSS). These changes could increase susceptibility to impacts of Harmful Algal Blooms (HABs) and invasive aquatic species ranging from benthic species to plants to fish.

The impacts to the air quality within the Project Area as a result of the implementation of Alternative C would be short-term, minor, adverse impacts during the construction period only, other than those of future recreational activities that would be addressed in future NEPA document(s).

CTO Alternative

Alternative CTO with a weir

Impacts on environmental quality would be similar to those discussed for Alternative C.

Alternative CTO without a weir

There would be temporary increases in turbidity and suspended solids in the Pearl River and tributaries during construction. Impacts to air quality would be similar to those discussed for Alt C.

1.3.6.4 Landform (topographic) Features

Alternative A-1

There would be no impacts to landform features due to implementation of Alternative A-1.

Alternative C

The current topographic features in the project area include the Pearl River, natural ridges, Native American earthworks/mounds, existing levees, and agricultural fields. Some areas within the project footprint would be degraded to elevations lower than existing. Additionally, the disposal areas would result in increased elevations within those areas.

CTO Alternative

Alternative CTO with a weir

Changes in landform features would be similar to those discussed for Alternative C except to a lesser degree.

Alternative CTO without a weir

Changes in landform features would be the same as those discussed for CTO with a weir.

1.3.6.5 Soil and Sediment

Alternative A-1

Any potential for impacts to soils would be temporary in nature and would occur only during the period of construction during the elevation activities, demolition, and/or relocation activities.

Alternative C

Approximately 20 million cy of existing soils within the project area would be removed and placed in the designated disposal areas. Indirect impacts to soils within the Project Area could be anticipated because of ongoing operations and associated maintenance through the life of the project. There is potential for increased sedimentation in the river from the channel excavation. Best Management Practices would be implemented to reduce this.

CTO Alternative

Alternative CTO with and without a weir

Changes to soils due to this alternative would be similar to those discussed for alternative C except to a lesser degree. Approximately 14 million cy of existing soils within the project area would be removed and placed in the designated disposal areas. Sedimentation would be the same as for Alternative C.

1.4 Action Area

The Action Area consists of the Pearl River floodplain from the Ross Barnett Dam to just south of Byram and includes land in Madison, Rankin, and Hinds Counties, Mississippi. The study area is drained by several small creeks that are tributaries of the Pearl River. Small tributaries to the Pearl River within the Action Area include Town, Hanging Moss, Eubanks, Lynch, Richland, Hardy, Caney, Purple, and Hog Creeks.

The Action Area includes the Pearl River Basin between River Mile (RM) 270.0 just south of Byram (32°10'20.95"N 90°14'41.98"W), Mississippi, and RM 301.77 at the dam of Ross Barnett Reservoir (32°24'39.58"N 90° 3'0.22"W) (Figure1). The Action Area also includes riparian areas adjacent to the river where construction activities would occur.

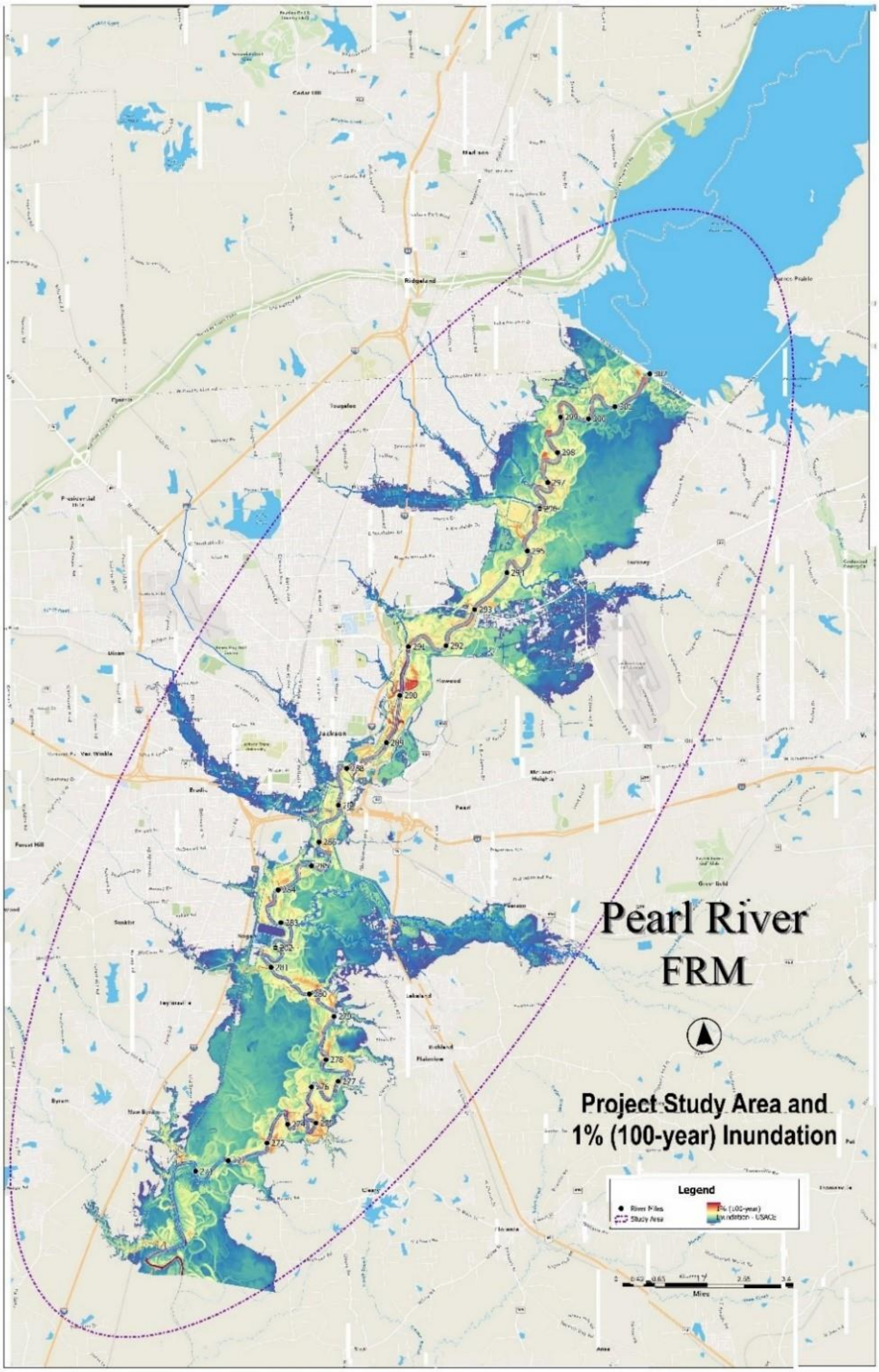


Figure 1 Action Area

1.5 Conservation Measures

Alternative A1 Not applicable

Alternative C and CTO

Conservation measures have not been identified as of yet. USACE is coordinating closely with the Service to develop conservation measures for each species as needed.

1.6 Prior Consultation History

The BA completed as a part of the environmental review process for the Draft FS/ESI included a review of literature and other pertinent scientific data, interviews and coordination efforts for each of the aforementioned T & E species as originally identified by the USFWS in correspondence dated June 8, 2004, and follow up coordination and listing reviews from 2015 through 2018. The listed species covered under that BA included the threatened Gulf sturgeon, the threatened Ringed Sawback (Ringed map) Turtle, the threatened Northern long-eared bat and the threatened Wood stork. It also includes a review of the listing information for the recently listed Pearl Darter. Though no longer listed, it also includes a review and assessment of both the American bald eagle and the Louisiana black bear.

The most recent consultation was completed on Oct 23, 2019, when The Service rendered a Biological Opinion (BO) (FWS Log #: 04EL1000- 2020-F-0109) that included an Incidental Take Statement requiring the USACE to implement reasonable and prudent measures that the Service considers necessary or appropriate to minimize the impacts of anticipated take on the ringed map turtle (*Graptemys oculifera*) and Gulf sturgeon (*Acipenser oxyrhynchus desotoi*). Since this opinion was issued, new data has revealed the presence of gulf sturgeon within the action area (Michael Andres, personal communication, January 12, 2023). In a letter dated Jan 18, 2023, the Service recommended the draft EIS reflect these recent findings, since the previous draft EIS and biological assessment incorrectly stated that gulf sturgeon were not likely to migrate into the project area for potential spawning. In addition, since only a conceptual design was provided for the fish passage channel in 2019, additional coordination is required with the Service to ensure Gulf sturgeon can successfully pass the new weir which would be considerably larger than the existing weir (2019 BO, Gulf Sturgeon RPM #2). Re-initiation of formal consultation will be required as the designs of the proposed alternatives and associated mitigation plans are currently not finalized. Development of the designs and mitigation plans will be coordinated with The Service.

1.7 Other Agency Partners and Interested Parties

- Rankin Hinds Pearl River Flood & Drainage Control District
- Environmental Protection Agency (EPA) Region 4
- Mississippi Department of Environmental Quality (MDEQ)
- Federal Emergency Management Agency (FEMA) Region IV
- Mississippi Department of Wildlife Fisheries and Parks (MDWFP)
- Mississippi Department of Marine Resources (MDMR)
- Mississippi Natural Resources Conservation Service (MNRCS)
- LA Department of Wildlife and Fisheries (LDWF)
- LA Department of Environmental Quality (LDEQ)
- Louisiana Department of Natural Resources (LDNR)
- Louisiana Coastal Protection and Restoration Authority (CPRA)
- U.S. Fish and Wildlife Service Jackson District
- USFWS Lafayette District
- Mississippi Department of Archives & History

1.8 Other Reports and Helpful Information

- 2019 Rankin Hinds Pearl River Flood and Drainage Control District Threatened and Endangered Species Biological Assessment
- 2019 USFWS Threatened and Endangered Species Biological Opinion
- Gulf Sturgeon Recovery/Management Plan
- Ringed Sawback Turtle Recovery Plan
- Species Status Assessment Report for the Northern long-eared bat
- Species Status Assessment Report for the Pearl River map Turtle
- Species Status Assessment Report for the Alligator Snapping Turtle
- Species Status Assessment Report for the Louisiana pigtoe
- Species Status Assessment Report for the tricolored bat

2 Species Effects Analysis

Alternative A1 is not expected to impact any of the listed species in the area and therefore will not be discussed further.

2.1 Gulf sturgeon (*Acipenser oxyrhynchus desotoi*)

2.1.1 Status of the species

2.1.1.1 Legal status

The Gulf Sturgeon is listed as Threatened under the Endangered Species Act (Federal Register Vol. 56, No. 189, September 30, 1991).

2.1.1.2 Recovery plans

The most recent recovery plan available for the Gulf sturgeon is dated September 1995 (Annex D3).

2.1.1.3 Life history information

The Gulf sturgeon is an anadromous fish (ascending rivers from the sea for breeding) that have historically inhabited coastal rivers from the Mississippi in Louisiana to the Tampa Bay in Florida. The Gulf sturgeon is one (1) of two (2) geographically dispersed subspecies of the Atlantic Sturgeon (*Acipenser oxyrhynchus*).

The Gulf sturgeon is characterized by a sub-cylindrical body that is imbedded with bony plates or “scutes”. The snout of the fish is greatly extended and bladelike and includes four (4) fleshy barbells in front of the mouth. The upper lobe of the tail is longer than the lower lobe. Adult specimens generally range in size from 1.8 to 2.4 meters (m) or six (6) to eight (8) feet in length. They are typically light brown to dark brown in color but are known to vary in color from grayish brown to bluish black on their back and sides, grading to white on their belly.

Age at sexual maturity ranges from 8 to 12 years for females and 7 to 9 years for males (Huff 1975). The Gulf sturgeon is a long-lived species, with some individuals reaching at least 42 years in age (Huff 1975).

The feeding habits of the Gulf sturgeon vary, depending upon the fish’s age (i.e., young-of-year, juvenile, sub-adult, adult) and is closely associated with migration and spawning habits. Throughout fall and winter, juveniles feed in the lower salinity areas in the river mouth and estuary (Sulak and Clugston 1999; Sulak et al. 2009), while subadults and adults migrate and feed in the estuaries and nearshore Gulf of Mexico habitat (Foster 1993; Foster and Clugston 1997; Edwards et al. 2003, 2007; Parkyn et al. 2007). Some Gulf sturgeon may also forage in the open Gulf of Mexico (Edwards et al. 2003).

The Gulf sturgeon typically inhabits the coastal rivers of the Gulf of Mexicoduring the warmer months of the year and generally overwinters in estuaries and bay environments within the Gulf of Mexico. The adults move into the tributary rivers for spawning in the spring and return to the Gulf waters in the fall. Spawning occurs in the upper reaches of rivers, at least 100 km (62 miles) upstream of the river mouth (Sulak et al. 2004), in habitats consisting of one or more of the following: limestone bluffs and outcroppings, cobble, limestone bedrock covered with gravel and small cobble, gravel, and sand (Marchant and Shutters 1996; Sulak and Clugston 1999; Heise et al. 1999a; Fox et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006). These hard bottom substrates are required for egg adherence and shelter for developing larvae (Sulak and Clugston 1998). Documented spawning depths range from 1.4 to 7.9 m (4.6 to 26 ft) (Fox et al. 2000; Ross et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006).

Further details on life history can be found in Annex D3.

2.1.1.4 Conservation needs

There are currently no conservation plans for the Gulf sturgeon. However, there is a Recovery Plan dated 1995 that includes an outline for recovery actions addressing threats to the Gulf sturgeon. Below are the main objectives. See Appendix D3 for further details.

- Determine essential ecosystems, identify essential habitats, assess population status, and refine life history investigations in management unit rivers.
- Protect individuals, populations, and their habitats.
- Coordinate and facilitate exchange of information on Gulf sturgeon conservation and recovery activities.

2.1.2 Environmental baseline

2.1.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

Recent studies for the Gulf sturgeon have not been conducted in this reach of the Pearl River and survey data from this area is not prevalent; however, in 2021, a Gulf sturgeon was detected above the waterworks weir in LeFleur's Bluff State Park and in 2022 the same sturgeon was detected closer to the spillway of Ross Barnet (Michael J. Andres, Ph.D., pers. comm. January 12, 2023). There are also unconfirmed sightings of Gulf sturgeon as far upstream as the City of Jackson, Mississippi, in Hinds County which is within the Action Area (Morrow et. al. 1996; Lorio 2000; Slack, pers. comm. 2002). There have been 24 Gulf sturgeon captured by commercial fishermen, eight of which being captured within the Action Area and the most recent of those captures occurring, a juvenile, in 2008.

The potential spawning habitat in the project area is believed to be minimal and significantly degraded due to the urbanization within the area and past flood control efforts. With the inclusion of the fish passage around the relocated weir, any adverse effects to potential spawning habitat thought to be associated with construction of the project would be minimized. There is no documented evidence that spawning activities occur within the project area. If there is spawning upstream of the weir, the presence of a reservoir without flow does cause large issues for juvenile fish. Post-hatching the sturgeon larvae go into a drift phase and if the velocities slow into a lake-like setting they will likely not survive. They are not great swimmers at this life stage and if they end up in a reservoir they can be exposed to excess predators among other threats.

2.1.2.2 Species conservation needs within the action area

There are currently no conservation plans for the Gulf sturgeon. However, there is a Recovery Plan dated 1995 that includes an outline for recovery actions addressing threats to the Gulf sturgeon. Below are the objectives that might be applicable to the action area. See Annex D3 for further details.

- Survey, monitor, and model populations.
- Reduce or eliminate unauthorized take.
- Identify and eliminate known or potentially harmful chemical contaminants, and water quantity and water quality problems which could impede recovery of Gulf sturgeon.
- Restore, enhance, and provide access to essential habitats.

2.1.2.3 Habitat condition (general)

Alternative C and CTO Alternative

Although designated Gulf sturgeon critical habitat is the entire PR within the project area (~287 acres (Alt C) and ~232 acres (CTO)), the extent of potential habitat for use by the Gulf sturgeon, is estimated to be approximately 230 acres (~9.5 linear miles) within the confines of the existing Pearl River channel from just north of U.S. Highway 25 southward to the proposed weir location south of U.S. Interstate 20. Bedrock and limestone outcroppings that are typical of Gulf sturgeon spawning areas in other river systems do not occur here. However, within the Pearl River drainage, spawning areas likely include soapstone, hard clay, gravel and rubble areas, and undercut banks adjacent to these substrates (W. Slack, pers. comm. 2001).

2.1.2.4 Influences

Over-fishing, associated with the commercial uses, resulted in a significant decline in Gulf sturgeon numbers throughout most of the 20th century. Incidental catch of Gulf sturgeon in other fisheries occurred at significant levels during the same time periods. Habitat losses associated with the construction of water control structures including dams and sills along the Gulf of Mexico drainage basins have contributed to a decline in populations throughout the historic range. Dam construction in several of the rivers has severely restricted the sturgeon’s access to historic migration routes and spawning areas. Water quality such as pollution, temperature, and dissolved oxygen levels are also a threat.

2.1.2.5 Additional baseline information

There is no additional baseline information.

2.1.3 Effects of the action

2.1.3.1 Indirect interactions

<u>Alternative C</u>

Until a vegetative cover is established along the excavated areas, all disturbed areas would be subject to erosion. This could potentially cause excess sediment to flow downstream approximately 1.6 miles south of the construction area and erosion could be exacerbated in that area until the riverbank has stabilized. The turbidity would be additive to any downstream riverbank erosion resulting from sediments being trapped behind the weir after its construction. Increased sediment and turbidity can result in decreased light penetration and decreased photosynthesis. Production of benthic organisms also can be reduced by high levels of sediment.

With the construction of the 1,500-foot-wide weir structure and resulting impoundment from the weir, changes to the velocity and water surface elevation would occur within the Action Area. The weir has been designed to match the current discharge of the river; therefore, there should not be significant change in discharge after the target area has filled to the top of the weir. The migratory blockage caused by the weir structure could impact the sturgeon's ability to swim north of the structure unless there are high water events; however, a fish passage channel has been included as part of the project design to minimize the impacts on aquatic species migration. Flow conditions would need to meet the needs of the species to allow for navigation of the passage. These conditions include water velocity that does not exceed the sturgeon's swim speed and enough water flow levels for the species to be able to swim through it.

Studies have shown that Gulf sturgeon cannot swim against currents greater than 1 to 2 meters per second (mps) (3 to 6 fps). Studies on fish passage attraction speed flow has shown that the recommended flow should be between 2 and 4 fps with sustained swim speed ranges for sturgeon to be in the range of 3 to 4 fps (Cheong et al. 2006; White and Mefford 2002). At this time, there is only a conceptual model of the fish passage channel, approximately 1.4 miles long of a curving channel, with the possible velocities ranging anywhere from 1 to 7 fps. The optimal velocities of 2 to 4 fps will be considered during detail design of the fish passage. Velocity analysis is needed to determine the indirect impacts to GS due to construction of the fish passage. This analysis would be conducted during PED.

<u>CTO Alternative</u>

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the GS depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same indirect impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir were not included, indirect impacts in the way of changes to water velocity, water surface elevation, and water quality may occur during high water events. This would not be much different from the current conditions during high water events and the impacts would be temporary and to a much lesser extent than with a weir. Impacts to the riparian zone would remain as excavation activities would still take place.

2.1.3.2 Direct interactions

Alternative C

Approximately 287 acres of riverine habitat would be impacted by the channel excavation. While the construction activities are being conducted, the disturbance to the sediment would increase the turbidity in the river. Increased sediment and turbidity can result in decreased light penetration and decreased photosynthesis. High levels of sediment can settle on fish spawning areas and smother fish eggs and larvae. Sediments can settle on respiratory surfaces of fish and aquatic organisms and interfere with respiration. The increased sedimentation and turbidity in the river from the channel excavation and levee relocation would have impacts on the macroinvertebrate prey for any juvenile Gulf sturgeon that would be temporarily feeding in the Action Area.

CTO Alternative

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the GS depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same direct impacts as Alternative C but to a lesser degree, if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir is not included, direct impacts to GS would be due to temporary increase in sedimentation and decrease in water quality during construction due to overbank excavation.

2.1.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.1.5 Discussion and conclusion

Alternative C and CTO with a weir

To offset or reduce impacts to Gulf sturgeon, a 1-mile fish passage would be constructed at the location of the new weir. This would allow for migration upriver that would have otherwise been cut off due to the weir.

Based upon literature review, available survey data, the current status of the species, the environmental baseline for the action area, and the effects of the action, the USACE has determined that implementation of Alternative C and CTO with a weir are likely to adversely affect but not likely to jeopardize the continued existence of, the Gulf sturgeon.

CTO Alternative without a weir

Based upon literature review, available survey data, the current status of the species, the environmental baseline for the action area, and the effects of the action, USACE has determined that implementation of Alternative CTO without a weir may affect but is not likely to adversely affect the GS or GS critical habitat.

2.2 Ringed Sawback (ringed map) Turtle (*Graptemys oculifera)*

2.2.1 Status of the species

2.2.1.1 Legal status

The ringed sawback turtle is listed as threatened under the Endangered Species Act (Federal Register Vol. 51, No. 246, December 23, 1986).

2.2.1.2 Recovery plans

The most recent recovery plan available for the ringed map turtle is dated April 1988. (Annex D3).

2.2.1.3 Life history information

The ringed map turtle is a small (7.5 to 22 cm) narrow-headed turtle with laterally compressed, black, spine-like vertebral projections and a slightly serrated posterior carapacial margin. The carapace is dark olive-green and each pleural has a broad yellow or orange circular mark.

The ringed map turtle is a wholly carnivorous species, with insects and mollusks constituting their principal diet. In addition, they are also thought to be opportunistic in their feeding habits with fish and carrion as occasional food sources.

The ringed map turtle's habitat is typically riverine with a moderate current and numerous basking structures. This species has also been observed in oxbow lakes that are connected or disconnected from the main river system, at densities 10-fold lower than within riverine systems.

Nesting habitat consists of large, high sand bars adjacent to the river. Sandbars range in size from 430 square feet (40 square meters) to over 2.2 acres (8,900 square meters) and are generally composed of 39 percent open sand, 38 percent herbaceous vegetation, and 23 percent woody vegetation (Jones 2006). Nesting is initiated in May and ends in August with multiple (2 to 3) clutches per year being common.

Average longevity estimates were 13.9 for females and 8.5 years for males. Males mature at about 4.6 years of age while females mature about 9.1 years of age (Jones 2017).

Further details on life history can be found in Annex D3.

2.2.1.4 Conservation needs

There are currently no conservation plans for the ringed map turtle. However, there is a Recovery Plan dated 1988 that includes an outline for recovery actions addressing threats to the ringed map turtle. Below are the main objectives. See Annex D3 for further details.

- Protection of a total of 150 miles of the turtle's habitat in two reaches of the Pearl River. There must be a minimum of 30 miles in either reach with the total protected area totaling 150 river miles.
- Evidence of a stable or increasing population over at least a ten-year period in these two Pearl River reaches.
- An established, continuing plan of periodic monitoring of population trends and habitat to ensure a stable population in these river reaches.

2.2.2 Environmental baseline

2.2.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

Populations are known to occur within the Pearl River system from the Neshoba County, Mississippi headwaters area, southward downstream through St. Tammany Parish, Louisiana. The ringed map turtle populations are restricted primarily to the main channel of the Pearl River and the lower portions of its largest tributary, the Bogue Chitto River. To date, the highest densities of turtles have been documented in two survey areas, above the Ross Barnett Reservoir and below the Ross Barnett Reservoir dam southward to approximately MS Highway 25, upstream of the Project Area. Ringed map turtles are found throughout all reaches of the Pearl River within the Action Area, with lower numbers in the channelized sections of the river (just south of RM 293 to approximately RM 287).

Approximately 40 percent of the proposed excavation area has little or no riparian habitat and little to no natural basking and feeding habitat, especially within the channelized portion. Selman (2018) found a greater concentration of turtles within forested riparian sites along this portion of the river. He also documented nest sites, turtle nesting crawls, and juvenile turtles all indicative of successful recruitment occurring in all stretches of the Action Area, including the area with reduced riparian habitat. It is estimated that a total of approximately 5,108 turtles occur in the Action Area.

2.2.2.2 Species conservation needs within the action area

There are currently no conservation plans for the ringed map turtle. However, there is a Recovery Plan dated 1988 that includes an outline for recovery actions addressing threats to the ringed map turtle. Below are the objectives that might be applicable to the action area. Annex D3 contains further details.

- Estimate number of ringed map turtles per mile in each of the study reaches.
- Determine seasonal and daily activity.
- Determine if the species moves any distance during its lifetime and barriers to such movement, if any.
- Protect two river reaches from activities that would cause a decline of this species' population.
- Develop and implement a monitoring plan to evaluate effectiveness of protective measures and to track population trends.

2.2.2.3 Habitat condition (general)

<u>Alternative C and CTO</u>

Habitat for the ringed map turtle is typically riverine with a moderate current and numerous basking logs. Populations are typically most abundant in areas of the river that have moderate to fast currents with deep water and sand and gravel bottoms. It is also important that the riverine habitat include numerous basking logs located in direct sunlight and with large sparsely vegetated sandbars that provide nesting habitat. The river channel itself must be wide enough to allow sunlight to penetrate for several hours a day. Nesting habitat for the turtles appears to be strictly limited to large, high sand and gravel bars located adjacent to the river channel.

The project area contains habitat that has been previously manipulated by the construction of levee's, channelization/straightening of the river, and elimination of a riparian buffer in places. Moreover, there is little natural basking habitat inside the project area.

2.2.2.4 Influences

Decline in populations of the ringed map turtle in certain areas of the Pearl River system have been attributed to habitat modifications, primarily associated with dredging and/or other navigational and flood control projects. Water quality degradation also seems to play an important role in population declines, over time, as has over-collecting of the species for the pet trade. In addition, recreational and other similar activities on the river may also cause habitat destruction, over time, especially as it relates to available nesting habitat on sandbars and/or the nesting activity itself. Predation of nests by raccoons, armadillos, and fish crows is also a threat to populations. The impact of human disturbance, primarily recreating (e.g., camping, picnicking, boating) to nesting turtles and/or nests has been pointed to as another source of decline in the population (Jones 2006; Jones 2017; Selman and Jones 2017). Direct mortality associated with recreational and commercial fishing and recreational boating has been identified as another impact to Graptemys populations (Bluté et al. 2010; Selman et al. 2013; Smith et al. 2018). Jones (2017) expressed a concern about those same activities impacting the ringed map turtle.

2.2.2.5 Additional baseline information

There is no additional baseline information.

2.2.3 Effects of the action

2.2.3.1 Indirect interactions

Alternative C

The establishment of an approximate 2,562-acre impoundment from weir construction would result in changes in the velocity and water surface elevation within the project area. Because the weir has been designed to match the current discharge of the river there should not be a significant change in discharge once flows begin overtopping the weir. The current lotic habitat would be replaced with a lentic habitat which has been proven by the Ross Barnett Reservoir to not support the persistence of the ringed map turtle.

The riparian zone would be almost eliminated, and development is likely for most of the areas of fill surrounding the improved channel. This would eliminate available habitat and increase disturbance. There is potential for existing nests to be flooded during filling of the pool area behind the weir if this occurs from May to

October. Details of how the filling would be undertaken have not been finalized but would be coordinated with the Service.

Free-flowing river reaches typically support a higher quality macroinvertebrate community while pool communities typically consist of relatively few taxa dominated by oligochaetes and chironomid larvae that are more tolerant of poorer water quality. Until recolonization of macroinvertebrates the competition for food resources within the channelized area would impact all ringed map turtles within the impoundment.

Turtles downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation on sandbars and basking material during construction. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the ringed map turtle.

Fluctuations and stratifications in the water quality (e.g., DO) like what occurs in the Ross Barnett Reservoir (larger but similar in depth) could be expected. This could result in poorer and/or reduced food sources because of decreased water quality and the potential influence of contaminants.

The fish -passage channel would provide approximately 1 mile (0.2 percent of the species range) of flowing water during low flow periods when the channelized area would experience low velocities. Depending on the width and velocities of this feature it could provide additional habitat for the ringed map turtle and would prevent isolation of the populations up and down stream of the weir.

It is anticipated that downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result primarily from an increase in turbidity decreasing potential food sources.

<u>CTO Alternative</u>

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the ringed map turtle depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same indirect impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir is not included, indirect impacts would be due to changes to water velocity, water surface elevation, and water quality during high water events. This would not be much different from the current conditions during high water events and the impacts would be temporary during each event and to a much lesser extent than with a weir.

2.2.3.2 Direct interactions

<u>Alternative C</u>

Disturbance from excavation of material from within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area would slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking, and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

<u>CTO Alternative</u>

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the ringed map turtle depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same direct impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir is not included, direct impacts are expected by the way of the species avoiding the area during construction activities. Also, temporary impacts due to increased sedimentation and decreased water quality during construction activities. Additionally, there is the potential for some individuals being directly killed during overbank excavation activities. This would be mitigated by surveying the area during construction activities and relocating individuals and nests if found.

2.2.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.2.5 Discussion and conclusion

Alternative C and CTO with a Weir

Previous consultation resulted in the following to offset or reduce direct losses of turtles due to construction. However, USACE is currently coordinating with USFWS to determine if these measures are still applicable. Eggs would be relocated outside the construction area and protected from predators and approximately 0.03 percent of the total population would be relocated from Cypress Lake to the Pearl River. In addition, approximately 1 percent of the total population would be trapped, tagged, data collected, tracked, observed, and monitored in the Action Area population.

Additional offsets to turtle losses that could be implemented as part of the Action include: (1) the creation and protection of 31 acres of nesting habitat and adjacent basking habitat and predator control; (2) the establishment and enforcement of no-wake zones to reduce boat strikes and disturbance during basking; (3) the placement of public access conditions to reduce disturbances to basking and nesting behaviors and habitats (4) the creation of an approximately 1 mile fish by-pass, and (5) the protection of 10 miles of riverbank that would prevent the development and destruction of riparian habitat utilized by the turtle and also reduce nesting and basking disturbances.

Based upon literature review and available survey data, and the effects of the action (both detrimental and mitigation activities proposed), the USACE has determined that implementation of Alternative C is likely to adversely affect but is not likely to jeopardize the continued existence of the ringed map turtle.

Alternative CTO without a weir

Based upon literature review and available survey data, and the effects of the action, although substantially less than that with a weir, the USACE has determined that implementation of Alternative CTO without a weir is likely to adversely affect but is not likely to jeopardize the continued existence of the ringed map turtle. This determination is due to the overbank excavation and the need to capture and relocate ringed map turtles.

2.3 Northern long-eared bat (NLEB) (*Myotis septentrionalis*)

2.3.1 Status of the species

2.3.1.1 Legal status

The NLEB is listed as an endangered species under the Act (87 FR 73488 November 30, 2022)

2.3.1.2 Recovery plans

There are currently no recovery plans for the NLEB. However, there is a SSA dated August 2022 (Annex D3).

2.3.1.3 Life history information

NLEB, a wide-ranging bat species, found in 37 states and 8 provinces in North America, typically overwinters in caves or mines and spends the remainder of the year in forested habitats. The NLEB individuals are typically approximately 3.0 to 3.7 inches in length with a wingspan of approximately 9.0 to 10.0 inches. The bat is distinguished by its long ears, particularly when compared to the other bats in the same genus, Myotis. The primary diet for the NLEB is insects including moths, flies, leafhoppers, caddisflies, and beetles.

Generalized annual life history diagram for NLEB (adapted from Silvis et al. 2016, p. 1).

2.3.1.4 Conservation needs

The SSA dated August 2022 includes conservation efforts for the NLEB. Below are the conservation efforts listed in the SSA. See Annex D3 for further details.

- NLEB receives varying degrees of protection through state laws as it is designated as Endangered in Arkansas, Connecticut, Delaware, Indiana, Maine, Massachusetts, Missouri, New Hampshire, Vermont; Threatened in Georgia, Illinois, Louisiana, Maryland, New York, Ohio, Pennsylvania, Tennessee, Virginia, and Wisconsin; and Special Concern in Alabama, Iowa, Michigan, Minnesota, Mississippi, Oklahoma, South Carolina, South Dakota, West Virginia, and Wyoming.
- Multiple national and international efforts are underway in attempt to reduce the impacts of white nose syndrome by determining the cause of the disease and reducing or slowing its spread.
- Operational strategies at wind power facilities.
- Forestry programs/forest management
- Bat-friendly gates to protect important hibernation sites.

2.3.2 Environmental baseline

2.3.2.1 Species presence and use

Alternative C and CTO Alternative

Although the USFWS ECOS webpage does not include the counties of Hinds and Rankin as part of the NLEB range, the Service has identified what is referred to as the White-Nose Syndrome Buffer Zone that includes all areas within 150 miles of the boundaries of U.S. counties or Canadian districts where the fungus has previously been detected. The established buffer zone includes both Hinds and Rankin Counties within the Project Area.

At this point, the Service does not have survey data that would indicate what the migration patterns are for the NLEB. More specifically, little is known whether the available summertime woodland habitat present within the Project Area is being utilized by the NLEB. No existing data is available that would indicate that the NLEB currently utilizes the Project Area during the summer migration.

2.3.2.2 Species conservation needs within the action area

The SSA dated August 2022 includes conservation efforts for the NLEB. Below are the conservation efforts listed in the SSA that might be applicable to the action area. See Annex D3 for further details.

- NLEB receives protection through Mississippi state law as it is designated as Endangered in Mississippi.

2.3.2.3 Habitat condition (general)

Alternative C and CTO Alternative

NLEBs typically roost singly or in maternity colonies underneath bark or more often in cavities or crevices of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 662; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 222; Timpone et al. 2010, p. 119). Males' and non-reproductive females' summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, p. 77; Amelon and Burhans 2006, p. 72). NLEBs are flexible in tree species selection and while they may select for certain tree species regionally, likely are not dependent on certain species of trees for roosts throughout their range; rather, many tree species that form suitable cavities or retain bark will be used by the bats opportunistically (Foster and Kurta 1999, p. 668; Silvis et al. 2016, p. 12; Hyzy 2020, p. 62).

NLEBs are thought to predominantly overwinter in hibernacula that include caves and abandoned mines. NLEBs are typically found roosting singly or in small numbers in cave or mine walls or ceilings, often in small crevices or cracks.

2.3.2.4 Influences

The Northern Long-eared Bat (NLEB) is one of the species of bats that have been most impacted by the spread of the white- nose syndrome disease and has experienced significant declines in populations because of the spread of the disease. Secondary threats to the NLEB include the disturbance of roosts and hibernation areas, forest management practices, and forest habitat modifications (development, wind power development).

2.3.2.5 Additional baseline information

There is no additional baseline information.

2.3.3 Effects of the action

2.3.3.1 Indirect interactions

Alternative C, CTO with a weir and CTO without a weir

All alternatives would remove potential roosting and foraging habitat (forests and structures such as abandoned bridges) and could result in potential adverse effects to the NLEB.

2.3.3.2 Direct interactions

Alternative C, CTO with a weir, and CTO without a weir

No direct interactions are anticipated as no existing data is available that would indicate that the NLEB currently utilizes the project area. However, if individuals were present during migration (summer months), and if construction activities were to take place at that time, it is safe to assume that the bats would avoid the area due to construction activities. Additionally, if surveys are conducted and females are found using the area during maternity pup season (May 1 – July 31), any tree removal activities would be required to take place in the non-maternity season (August 1 – April 30).

2.3.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.3.5 Discussion and conclusion

<u>Alternative C, CTO with a weir, and CTO without a weir</u>

The project would not occur near or affect any known maternity roost trees but would remove potential roosting and foraging habitat and could result in potential adverse effects. Also, no existing data is available that would indicate that the NLEB currently utilizes the Project Area. Additionally, with the implementation of tree clearing restrictions, these impacts would be minimized or avoided. Accordingly, the USACE has determined that Alternative C, CTO with a weir, and CTO without a weir, may affect, but is not likely to adversely affect the NLEB.

2.4 Pearl River Map Turtle (PRMT) *(Graptemys pearlensis)*

2.4.1 Status of the species

2.4.1.1 Legal status

The current listing of the Pearl River map turtle is “Proposed Threatened” (Federal Register Vol. 86, No. 223, November 23, 2021, p66624).

2.4.1.2 Recovery plans

There are currently no recovery plans for the PRMT. However, there is a Species Status Assessment Report (SSA) dated April 2021 (Annex D3).

2.4.1.3 Life history information

The PRMT is endemic to the Pearl River drainage in Mississippi and Louisiana. Rankin and Hinds Counties are included in the Counties with known records for the species in the state of Mississippi. The occupied range of the PRMT includes portions of the Pearl River, West Pearl River, Bogue Chitto, East Pearl River, Yockanookany River, Strong River, Holmes Bayou, Pearl Navigation Canal, Lobutcha Creek, Tuscolometa Creek, Pelahatchie Creek, Purvis Creek, Pushepatapa Creek, Topisaw Creek, Magees Creek, Hobolochitto Creek, and West Hobolochitto Creek. This species has also been reported in upper reaches of the Ross Barnett Reservoir.

The PRMT is a moderate-sized highly aquatic turtle found in the Pearl River drainage area of Louisiana and Mississippi. Female PRMTs have an average carapace length of 295 mm, with the male PRMT having an average carapace length of 121 mm. The PRMT exhibits a high-domed shell with a median keel, featuring salient spines on the rear portions of the anterior vertebral scutes; although similar visually, the spines are considerably smaller to that of the ringed map turtles. A key distinguishing feature of the PRMT is the complete dark stripe along the median keel and the large yellow blotch created by the connection of the postorbital and interorbital blotches on the head. The background color of the carapace is olive green, with vermiculation and yellow pigmentation present. The plastron is generally flat and pale yellow with dark pigmentations along the seams.

The PRMT is a wholly carnivorous species, with insects and mollusks constituting their principal diet. In addition, they are also thought to be opportunistic in their feeding habits with fish and carrion as occasional food sources. A recent study found that mature females consume mostly Asian clams (*Corbicula fluminea*), while males and unsexed juveniles eat insects, with mature males specializing in caddisfly larvae and consuming more mollusks than juveniles (Vučenović and Lindeman 2020, entire). In fecal samples from a site on the Pearl River, the diet for both sexes of all sizes combined was composed of 44 percent fish, 25 percent mollusks, and 25 percent insects (McCoy and Vogt, unpubl. data reported in Lovich et al. 2009, p. 029.4).

A study on the ringed map turtle (Graptemys oculifera), found that PRMTs were more frequently seen basking later in the afternoon than ringed map turtles, and suggested that more PRMTs might have been detected if more surveys were conducted after 3 pm (Dickerson and Reine 1996, p.8).

PRMTs excavate nests and lay their eggs on sandbars and beaches along riverbanks during the late spring and early summer months. The time from deposition to nest emergence by hatchlings in natural clutches average 69.3 days. An average clutch size of 6.4 eggs with a range of 4-9 eggs was reported for the PRMT and stated that females probably produce multiple clutches per year (Ennen et al. 2016, pp. 094.4-094.6).

Humans, alligators, alligator snapping turtles and otters are predators of adult PRMTs, with eggs and hatchlings more susceptible to small mammals, snakes, and crows. Red imported fire ants have also been documented invading turtle nests in the southeastern United States and can cause nest failure and hatchling mortality (Buhlmann and Coffman 2001, entire).

2.4.1.4 Conservation needs

The SSA dated April 2021 includes conservation measures for the PRMT. Below are the federal conservation measures listed in the SSA. See Annex D3 for further details.

- The same recovery actions that are listed for the ringed sawback turtle could benefit the PRMT (see section 2.2.1.4).
- The Clean Water Act of 1972 which encourages avoidance, minimizing and requires mitigation for unavoidable impacts to the aquatic environment and habitats. This includes protecting the riverine habitat occupied by the PRMT.
- The Endangered Species Act (Act) could offer some protection as the PRMT likely receives ancillary protection where it co- occurs with other species listed under the ESA.
- A Comprehensive Conservation Plan (CCP) has been developed under The National Wildlife Refuge System Administration Act (NWRAA) to provide the framework of fish and wildlife management on the Bogue Chitto National Wildlife Refuge (U.S. Fish and Wildlife Service 2011, entire). Within the CCP, specific actions are described to protect the ringed map turtle that will also benefit the PRMT which occurs on the Refuge.
- The Sikes Act Improvement Act (1997) led to Department of Defense guidance regarding development of Integrated Natural Resources Management Plans (INRMP) for promoting environmental conservation on military installations. There are records of the PRMT from Stennis WMA (Buhlman 2014, pp. 11-12, 31-32). The U.S. Navy has developed an INRMP for the Stennis WMA (U.S. Navy 2011, entire).

2.4.2 Environmental baseline

2.4.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

The project area is in Rankin and Hinds Counties, Mississippi which are included in the Counties with known records for the species in the state of Mississippi. This species has also been reported in upper reaches of the Ross Barnett Reservoir.

PRMTs can be found within the project area despite the lack of a well-defined riparian buffer, lack of preferred habitat, sedimentation accumulation, relatively low stream velocities, lack of basking habitat, and a smaller percentage of sandbars. It has been shown in studies that population densities for the species are higher above and below the project area.

2.4.2.2 Species conservation needs within the action area

The Species Status Assessment Report (SSA) dated April 2021 includes conservation measures for the PRMT. Below are the state conservation measures that might be applicable to the action area. See Annex D3 for further details.

- The same recovery actions that are listed for the ringed sawback turtle could benefit the PRMT (see section 2.2.2.2).
- Protections under state law are limited to licensing restrictions for take for personal use of nongame species in need of management (which includes native species of turtles). A Mississippi resident is required to obtain one of three licenses for capture and possession of PRMTs.
- The Mississippi Comprehensive Wildlife Action Plan (MMNS 2015, entire) includes recovery of species designated as Species of Greatest Conservation Need (SGCN) which includes the PRMT.

2.4.2.3 Habitat condition (general)

<u>Alternative C and CTO</u>

PRMTs occur in sand and gravel-bottomed rivers and creeks with dense accumulations of deadwood; they have not been documented in oxbow lakes or other floodplain habitats. They were notably absent from lakes where the ringed map turtle is present but do occur at the upstream reach of Ross Barnett Reservoir (Lindeman 2013, p. 298). Emergent deadwood serves as thermoregulatory basking structure, foraging structure for males and juveniles (Selman and Lindeman 2015, pp. 794-795), and as an overnight resting place for males and juveniles (Cagle 1952, p. 227). PRMT density was greater on mainstem reaches and large tributaries than on small tributaries (Lindeman 2019, pp. 13-18).

2.4.2.4 Influences

Climate change, water quality, habitat degradation, invasive species, collection, and disease all influence the persistence of the species.

Variability in climate may affect ecosystem processes and communities resulting in potential effects on community composition and individual species interactions (DeWan, et al., 2010, p. 7). These changes have the potential to impact PRMTs and/or their habitat.

The dual stressors of climate change and direct human impact have the potential to impact aquatic ecosystems by altering stream flows and nutrient cycles, eliminating habitats, and changing community structure (Moore et al. 1997, pp. 942).

Degradation of stream and wetland systems through reduced water quality and increased concentrations of contaminants can affect the occurrence and abundance of freshwater turtles (DeCatanzaro and Chow-Fraser 2010, p. 360).

Dredging and channelization modify and destroy habitat for aquatic species by destabilizing the substrate, increasing erosion and siltation, removing woody debris, decreasing habitat heterogeneity, and stirring up contaminants which settle onto the substrate (Williams et al. 1993, pp. 7-8; Buckner et al. 2002, entire; Bennett et al. 2008, pp. 467-468). Considerably low densities of PRMTs were observed in the lower reaches of the Pearl, where much channelization and flow diversion has occurred (Lindeman 2019, pp. 23-29).

Impoundment of rivers is a primary threat to aquatic species in the southeast (Folkerts 1997, p. 11; Buckner et al. 2002, entire). Dams modify habitat conditions and aquatic communities both upstream and downstream of an impoundment (Winston et al. 1991, pp. 103-104; Mulholland and Lenat 1992, pp. 193-231; Soballe et al. 1992, pp. 421-474). Dams fragment habitat for aquatic species by blocking corridors for migration and dispersal, resulting in population geographic and genetic isolation and heightened susceptibility to extinction (Neves et al. 1997, unpaginated).

The degree to which invasive species effect the PRMT has not been studied, but the diet of mature females may have been broader before the introduction of Asian Clams (Corbicula fluminea) and removal of invasive vegetation on sandbars has been suggested as nesting habitat management (Selman and Lindeman 2015, p. 794-795; Lindeman 2019, p. 33).

Exploitation of PRMTs for the pet trade domestically and in Asian markets has been documented, but the degree of impact is unclear, as it is unknown whether captive individuals were Pascagoula ringed map turtles or PRMTs (Lindeman 1998, p. 137; Cheung and Dudgeon 2006, p. 756; USFWS 2006, p. 2; Selman and Qualls 2007, p. 32-34; Ennen et al. 2016, p. 094.6).

Ranaviruses are capable of infecting turtles. Aquatic turtles share habitat with susceptible fish and amphibian populations and as a result may be more at risk of infection than terrestrial turtles (Wirth et al. 2018, p. 6).

2.4.2.5 Additional baseline information

There is no additional baseline information.

2.4.3 Effects of the action

2.4.3.1 Indirect interactions

Alternative C

The establishment of a 2,562-acre impoundment from weir construction would result in changes in the velocity and water surface elevation within the project area. Because the weir has been designed to match the current discharge of the river there should not be a significant change in discharge once flows begin overtopping the weir. The current lotic habitat would be replaced with a lentic habitat which would not support the persistence of the PRMT.

The riparian zone would be almost eliminated, and development is likely for most of the areas of fill surrounding the improved channel. This would eliminate available habitat and increase disturbance. There is potential for existing nests to be flooded during filling of the pool area behind the weir if this occurs late spring to early summer months. Details of how the filling would be undertaken have not been finalized but would be coordinated with the Service.

Free-flowing river reaches typically support a higher quality macroinvertebrate community while pool communities typically consist of relatively few taxa dominated by oligochaetes and chironomid larvae that are more tolerant of poorer water quality. Until recolonization of macroinvertebrates is the competition for food resources within the channelized area would impact any PRMTs within the impoundment. It is expected that there would be proportionally more of a decline of PRMTs than there would be of ringed sawback turtles as PRMTs are riverine obligates that have never been documented in lentic systems, while ringed map turtles have been observed in lentic environments, albeit with populations densities 10-fold less than riverine habitats.

Turtles downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation on sandbars and basking material during construction. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the PRMT.

Fluctuations and stratifications in the water quality (e.g., DO) like what occurs in the Ross Barnett Reservoir (larger but similar in depth) could be expected. This could result in poorer and/or reduced food sources because of decreased water quality and the potential influence of contaminants.

The fish -passage channel would provide approximately 1 mile (0.2 percent of the species range) of flowing water during low flow periods when the channelized area would experience low velocities. Depending on the width and velocities of this feature it could provide additional habitat for the PRMT and would prevent isolation of the populations up and down stream of the weir.

It is anticipated that downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result primarily from an increase in turbidity decreasing potential food sources.

<u>CTO Alternative</u>

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the PRMT depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same indirect impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

Indirect impacts in the way of changes to water velocity, water surface elevation, and water quality may occur during high water events. This would not be much different from the current conditions during high water events and the impacts would be temporary and to a much lesser extent than with a weir.

2.4.3.2 Direct interactions

<u>Alternative C</u>

Disturbance from excavation of material from approximately within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area would slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking, and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

<u>CTO Alternative</u>

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the PRMT depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same direct impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

Direct impacts are expected by the way of the species avoiding the area during construction activities. Also, temporary indirect impacts may occur by way of increased sedimentation and decreased water quality during construction activities. Additionally, there is the potential for some individuals being directly killed during overbank excavation activities.

2.4.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.4.5 Discussion and conclusion

Alternative C and CTO with a weir

Previous consultation resulted in the following to offset or reduce direct losses of turtles due to construction. However, USACE is currently coordinating with USFWS to determine if these measures are still applicable. Eggs would be relocated outside the construction area and protected from predators and approximately 0.03 percent of the total population would be relocated from Cypress Lake to the Pearl River. In addition, approximately 1 percent of the total population would be trapped, tagged, data collected, tracked, observed, and monitored in the Action Area population.

Additional offsets to turtle losses that could be implemented as part of the Action include: (1) the creation and protection of 31 acres of nesting habitat and adjacent basking habitat and predator control; (2) the establishment and enforcement of no-wake zones to reduce boat strikes and disturbance during basking; (3) the placement of public access conditions to reduce disturbances to basking and nesting behaviors and habitats (4) the creation of an approximately 1 mile fish by-pass, and (5) the protection of 10 miles of riverbank that would prevent the development and destruction of riparian habitat utilized by the turtle and also reduce nesting and basking disturbances.

Based upon literature review and available survey data, and the effects of the action (both detrimental and mitigation activities proposed), the USACE has determined that implementation of Alternative C or CTO with a weir is likely to adversely affect but is not likely to jeopardize the continued existence of the PRMT.

Alternative CTO without a weir

Based upon literature review and available survey data, and the effects of the action, although tremendously less than that with a weir, the USACE has determined that implementation of Alternative CTO without a weir is likely to adversely affect but is not likely to jeopardize the continued existence of the PRMT. This determination is due to the overbank excavation and the need to capture and relocate PRMTs.

2.5 Alligator Snapping Turtle (AST) *(Macrochelys temminckii)*

2.5.1 Status of the species

2.5.1.1 Legal status

The current listing of the AST is "Proposed Threatened" (Federal Register Vol. 86, No. 214/Tuesday, November 9, 2021).

2.5.1.2 Recovery plans

There are currently no recovery plans for the alligator snapping turtle. However, there is a SSA dated March 2021 (Annex D3).

2.5.1.3 Life history information

The AST is the largest freshwater species of turtle in North America and is among the most aquatic. ASTs are characterized as having a large head, long tail, and an upper jaw with a hooked beak. They have three keels with posterior elevations on the scutes of the carapace, which is dark brown and often found with algae growth adding to the overall camouflage of the turtle. The plastron is greyish brown in adults, and somewhat mottled with small whitish blotches in juveniles. The eyes are positioned on the side of the head, surrounded by small, pointed projections.

The AST is found within river systems that flow into the Gulf of Mexico, extending from just before the Suwannee River in Florida to the San Antonio River in Texas. Currently, the species is known to occur in Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas.

ASTs are usually associated with the deeper waters of large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows. Hatchlings and juveniles, in comparison, are usually associated with shallower waters. In general, the species uses shallower water in early summer and deeper depths in late summer and mid-winter, which may be a thermoregulatory shift (Fitzgerald and Nelson 2011). The presence of barnacles on some specimens may also indicate an ability to spend prolonged periods in brackish water (Jackson and Ross 1971, p.188-189).

AST males reach sexual maturity in 11-21 years and 13-21 years for females. Females have been observed to have no more than a single clutch per year in the wild, as well as not appearing to be particularly selective on nesting sites. Nesting sites have been observed across a range of distances from 8 to 656 ft from the nearest water source. ASTs exhibit temperature dependent sex determination within nest incubation temperatures. Nesting occurs between May to July with areas in the most southern ranges beginning in April and extending through May. ASTs exhibit sexual dimorphism with males being distinctively larger than females, and also displaying a larger anterior to vent tail length.

ASTs are opportunistic scavengers and consume a variety of foods. Although fish comprise the majority of their diet, crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetations have also been reported (Elsey, 2006). The AST is the only turtle that uses a modified tongue appendage as a predatory lure to attract fish into range.

Racoons, armadillos, opossums, and otters are all known to prey on AST nests. Predators of hatchlings are likely to include large fish, wading birds, otters, and alligators (Ernst and Lovich 2009, p. 149). Red imported fire ants (Solenopsis invicta) are also known to cause significant decline in hatching success.

2.5.1.4 Conservation needs

The SSA dated March 2021 includes conservation measures for the AST. Below are the conservation measures listed in the SSA. See Annex D3 for further details on each.

- Captive Rearing, Head-Starting, and Reintroductions
- Integrated Natural Resource Management Plans
- Predator exclusion structures

2.5.2 Environmental baseline

2.5.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

ASTs were historically found in 14 states: Alabama, Arkansas, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. Currently, the species is known to occur in Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. ASTs are found in deeper waters and their major tributaries; however, their habitats have been known to extend into small streams, bayous, canals, swamps, lakes, reservoirs, ponds, and oxbows. The AST is usually associated with structure more so than open water. Riparian canopy cover is an important feature for the AST, as they typically select sites with a high percentage of coverage (Howey and Dinkelaker 2009).

The Service divides the AST range into seven (7) analysis units. The analysis unit focused on in relation to the project area is the Alabama unit which encompasses eastern Mississippi, western Alabama, and small parts of Louisiana and Florida. The Pearl River is listed under the Alabama unit as a water body that currently or historically supported ASTs.

The Alabama Analysis unit has an estimated abundance of 200,000 (55.37%). It is estimated range wide that there is between 68,154 and 1,436,825 individuals with 55 percent of the turtles occurring in the Alabama analysis unit (USFWS. “Federal Register / Vol. 86, No. 214 / Tuesday, November 9, 2021).

2.5.2.2 Species conservation needs within the action area

The SSA dated March 2021 includes conservation measures for the AST. However, there are no conservation needs specific to the action area.

2.5.2.3 Habitat condition (general)

ASTs are associated with deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows); with shallower water occupied in early summer and deeper depths in late summer and mid-winter, which represent a thermoregulatory shift (Ernst and Lovich 2009, p. 141). In comparison, hatchlings and juveniles tend to occupy shallower water. ASTs are also associated with structure (e.g., tree root masses, stumps, submerged trees, etc.); and may occupy areas with a high percentage of canopy cover undercut stream banks.

2.5.2.4 Influences

Adult harvest (legal and illegal), bycatch, habitat alteration, nest predation, climate change, and disease influence the existence of the AST.

Although regulatory harvest restrictions have decreased the number of ASTs being harvested, populations have not necessarily increased in response. This lag in population response is likely due to the demography of the species, specifically delayed maturity, long generation times, and relatively low reproductive output.

ASTs can be killed or harmed incidental to other fishing and recreational activities. Threats include capture as bycatch associated with commercial harvest of other species, ingestion of fishhooks and/or drowning when captured on trotlines (a fishing line strung across a stream with multiple hooks set at intervals) and limb lines (single hooks hung from branches), drowning from entanglement in various types of fishing line, and boat propeller strikes.

Dams change the hydrology of streams and could impede dispersal and genetic interchange for this highly aquatic species, but impoundments can also provide habitat for the species (Pritchard 1989, p. 84). Other activities and processes that can alter habitat include dredging, deadhead logging, removal of riparian cover, channelization, stream bank erosion, siltation, and land use adjacent to rivers (e.g., clearing land for agriculture).

Nest predation rates for the AST are high. Small mammals and red fire ants are known to prey on the nests. In 2008, one of five AST nests investigated in Louisiana was infested by the phorid fly Megaselia scalaris (snapping turtles; Holcomb and Carr 2011b, entire).

Climate change might impact the AST in several ways, including loss of habitat to sea level rise for those populations near coastal areas, impacts of drought on habitat and water availability, and physiological impacts on sex determination. Climate conditions also appear to limit the distribution of ASTs.

Chaffin et al. (2008, entire) captured and assessed the health of 97 free-ranging ASTs across nine sites in northwestern Florida and southwestern Georgia between 2001 and 2006. Assessed ASTs had shell abnormalities, including worn, cracked, or broken scutes, fresh or healed wounds resulting from trauma, missing portions of the tail, missing portions of the beak, missing portions of claws, and leech infestation (Chaffin et al. 2008, p. 674). Protozoan parasites transmitted by leeches, were found in all but one turtle assessed. Herpes was the only pathogen detected, but none of the individuals were showing symptoms. Mercury was also detected in the blood in 93% of samples.

2.5.2.5 Additional baseline information

There is not additional baseline information.

2.5.3 Effects of the action

2.5.3.1 Indirect interactions

Alternative C

Indirect impacts associated with the project would include the potential for degradation of water quality, loss of woody debris, nesting habitat loss due to flooding, nest predation issues and increases in bycatch due to recreation increase. There are also concerns about the potential impacts of the project on other species that rely on the same habitat as the AST. For example, the project and associated infrastructure could temporarily impact local fish populations, which in turn may impact the local turtle population as these fish populations are a primary food source for the AST.

Potential benefits of the project for the AST include the creation of a new, more suitable, and desirable habitat when compared to existing conditions. The construction of the project and associated infrastructure could provide new areas of deep, permanent water with a soft substrate for nesting. However, the recreational benefits that are anticipated to be implemented by the NFI could have adverse impacts by the way of increase in fishing bycatch on trotlines, limblines, and rod/reel. Implementing fishing regulations (i.e., no set lines or commercial nets) would reduce AST mortality due to these actions.

CTO Alternative

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the AST depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same indirect impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

Indirect impacts in the way of changes to water velocity, water surface elevation, and water quality may occur during high water events. This would not be much different from the current conditions during high water events and the impacts would be temporary and to a much lesser extent than with a weir. The benefits would not be realized if a weir were not constructed.

2.5.3.2 Direct interactions

Alternative C

Disturbance from excavation of material from within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area would slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

The construction of the flood control project and associated infrastructure could temporarily alter habitat conditions, leading to a decline in the AST population. In addition, the project could also potentially impact the AST through temporary changes in water quality. Impacts include removal of natural buffers that would impact water quality, and a slight decrease and less variation of dissolved oxygen concentrations.

CTO Alternative

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the AST depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same direct impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir is not included, direct impacts are expected by the way of the species avoiding the area during construction activities. Also, direct impacts by way of increased sedimentation and decreased water quality may occur during construction activities. Additionally, there is the potential for some individuals being directly killed during overbank excavation activities.

2.5.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.5.5 Discussion and conclusion

Alternative C and CTO with a weir

While the project raises concerns about the potential adverse impacts on the AST and environment, the potential benefits should also be considered. It is possible that the project would create an overall more desirable habitat for the species when compared to current habitat options within the project area. The project in general would provide more permanent deep-water habitat, potentially increase water quality, and increase the available soft substrate for nesting. However, the recreational benefits that are anticipated to be implemented by the NFI could have adverse impacts by the way of increase in fishing bycatch on trotlines, limblines, and rod/reel. Implementing fishing regulations (i.e., no set lines or commercial nets) would reduce AST mortality due to these actions.

Based upon literature review and available survey data, and the effects of the action (both detrimental and beneficial activities proposed), the USACE has determined that implementation of Alternative C and CTO with a weir is likely to adversely affect but not likely to jeopardize the continuing existence of the AST.

Alternative CTO without a weir

Based upon literature review and available survey data, and the effects of the action, although substantially less than that with a weir, USACE has determined that implementation of Alternative CTO without a weir is likely to adversely affect but not likely to jeopardize the continuing existence of the AST. This determination is due to the fact that some individuals could be killed during overbank excavation activities.

2.6 Tricolored Bat (TCB) (*Perimyotis subflavus*)

2.6.1 Status of the species

2.6.1.1 Legal status

The current listing of the Tricolored bat is "Proposed Endangered" (88 FR 16776, March 20, 2023, p16776-16832).

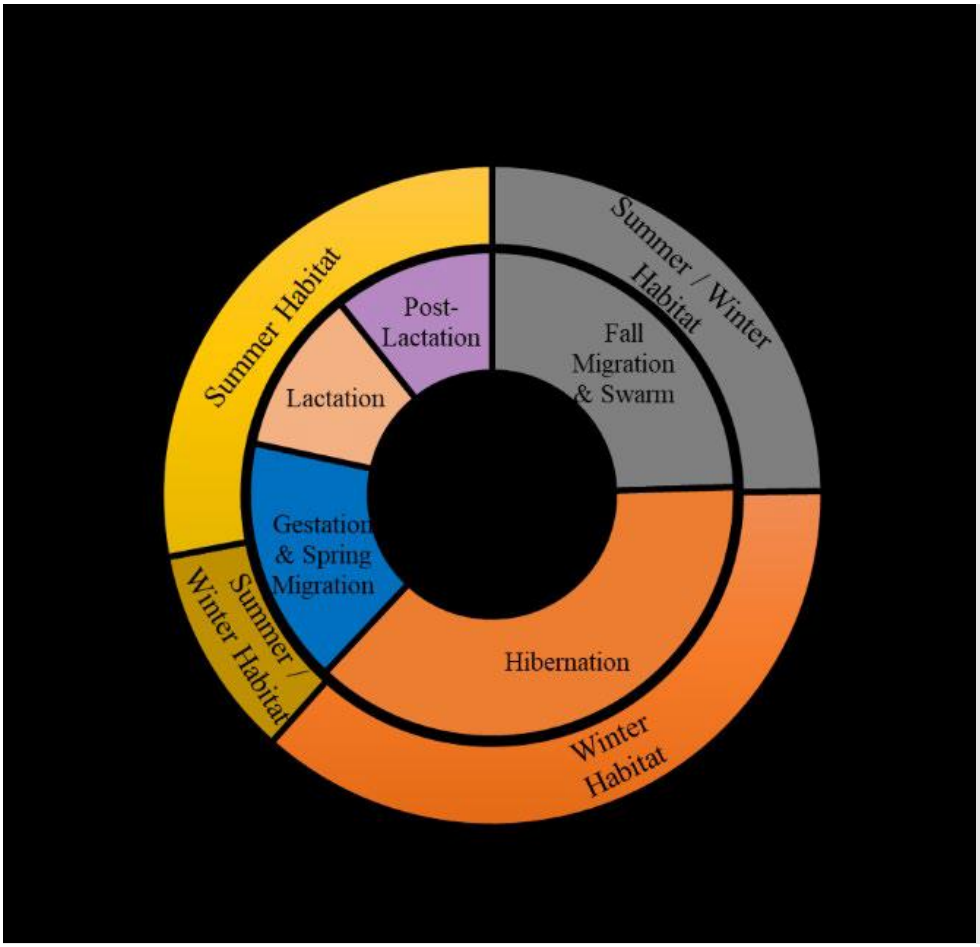
2.6.1.2 Recovery plans

There are currently no recovery plans for the tricolored bat. However, there is a SSA dated December 2021 (Annex D3).

2.6.1.3 Life history information

TCB is one of the smallest bats in eastern North America and is distinguished by its unique tricolored fur that appears dark at the base, lighter in the middle, and dark at the tip (Barbour and Davis 1969, p. 115). TCB primarily roost in foliage of live and dead trees in the spring, summer, and fall, and hibernate in caves and other subterranean habitats during the winter. TCB are opportunistic feeders feeding on small insects such as moths, beetles, flies, wasps, and flying ants.

TCB are known to occur in 39 states, one of which is Mississippi, Washington D.C., 4 Canadian Provinces, Guatemala, Honduras, Belize, Nicaragua, and Mexico.



Generalized annual life history diagram for TCB (adapted from Silvis et al. 2016, p. 1).

2.6.1.4 Conservation needs

The SSA dated December 2021 includes conservation efforts for the TCB. Below are the conservation efforts listed in the SSA. See Annex D3 for further details.

- TCB could receive varying degrees of protection through state and federal laws once the listing decision is made.
- Multiple national and international efforts are underway in attempt to reduce the impacts of white nose syndrome by determining the cause of the disease and reducing or slowing its spread.
- Operational strategies at wind power facilities.
- Forestry programs/forest management
- Bat-friendly gates to protect important hibernation sites.

2.6.2 Environmental baseline

2.6.2.1 Species presence and use

Alternative C and CTO Alternative

The tricolored bat is widespread throughout MS, and they can be found in many different habitat types throughout the year. The presence in the project area is not known at this time. However, it is safe to assume that the TCB may use the area for foraging, roosting and potentially wintering.

2.6.2.2 Species conservation needs within the action area

The SSA dated December 2021 includes conservation efforts for the TCB. However, conservation efforts within the action area have not yet been determined.

2.6.2.3 Habitat condition (general)

Alternative C and CTO Alternative

TCB seem to be opportunistic roosters and roost in live and dead leaf clusters of deciduous hardwood trees, Spanish moss, pine needles, eastern red cedar, barns, beneath porch roofs, bridges, concrete bunkers, and rarely within caves.

TCB have been documented overwintering in caves, mines, rock crevices, talus, tunnels, bunkers, basements, bridges, aqueducts, trees, earthen burrows, leaf litter, and a variety of other roosts. For bats to hibernate successfully, the most important conditions are relatively stable- low temperatures, but generally above freezing, and high humidity.

2.6.2.4 Influences

The TCB has been impacted by the spread of the WNS disease and has experienced significant declines in populations because of the spread of the disease. Other threats to the TCB include wind related mortality due to wind power development, climate change, and habitat loss.

2.6.2.5 Additional baseline information

There is no additional baseline information.

2.6.3 Effects of the action

2.6.3.1 Indirect interactions

Alternative C, CTO with a weir, and CTO without a weir

Indirect impacts would be due to the removal of potential roosting and foraging habitat (forests and structures such as abandoned bridges) and could result in potential adverse effects. Although WNS is the primary cause of decline in the

TCB population, habitat removal in the area could compound the impacts on the population.

2.6.3.2 Direct interactions

Alternative C, CTO with a weir, and CTO without a weir

Since the TCB is widespread throughout MS, there are no existing survey data for the project area, and they can be found in many different habitat types throughout the year, it is difficult to determine the direct impacts to the species at this time. However, if individuals are present at the time of construction, it is safe to assume that construction activities would cause the bats to flee the area. If construction activities take place during the winter (during hibernation), and individuals are present, then disturbance could result in increased arousals and energy expenditure during a time when food and water resources are likely scarce. Additionally, if surveys are conducted and TCB are found using the area, then tree removal activities for the project would not take place during the pup season (May 1 – July 31) or during the torpor season (December 15 - February 15).

2.6.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.6.5 Discussion and conclusion

Alternative C, CTO with a weir, and CTO without a weir

USACE has conducted literature reviews and is in coordination with the Service. Due to the lack of available survey data, and the fact that the TCB is widespread in Mississippi, the USACE will be conservative and assume that TCBs are utilizing the area.

If TCBs are utilizing the area, particularly for hibernation, all alternatives would not only remove roosting and foraging habitat but could also disturb hibernating bats potentially resulting in death of individuals. However, with the implementation of tree clearing restrictions, these impacts would be minimized or avoided. Based upon literature review and the effects of the action, the USACE has determined that implementation of Alternative C, CTO with a weir, and CTO without a weir may affect but is not likely to adversely affect the TCB.

2.7 Louisiana Pigtoe (LA pigtoe) (*Pleurobema riddellii*)

2.7.1 Status of the species

2.7.1.1 Legal status

The current listing of the Louisiana pigtoe mussel is “Proposed Threatened” (87 FR 56381, Sept 14, 2022, p56381-56393).

2.7.1.2 Recovery plans

There are currently no recovery plans for the Louisiana pigtoe. However, there is a SSA dated February 2022 (Annex D3).

2.7.1.3 Life history information

The LA pigtoe is a medium-sized freshwater mussel (shell lengths to greater than 62 mm) with a brown to black, triangular to subquadrate shell without external sculpturing, sometimes with greenish rays. They occur in gravel and coarse sandy substrates of rivers and streams. Mussels are filter feeders that rely on natural, high quality (pollutant free) flowing water of sufficient volume to support their life cycle, and that of their host fishes, which are essential for reproduction.

The range of the LA pigtoe extends into portions of east Oklahoma, southeast Arkansas, south Louisiana, and west Mississippi. Louisiana PA pigtoe currently occupies areas across seven major river basins (San Jacinto, Neches, Sabine, Big Cypress-Sulphur, Red, Calcasieu-Mermentau, and Pearl). However, within the Pearl River, the LA pigtoe is only found in the project area and a portion of the west Pearl.

Degraded water quality, altered hydrology, substrate changes, habitat fragmentation, direct mortality, invasive species, and climate change all influence the existence of the LA pigtoe. The remaining populations are in low condition and are therefore particularly vulnerable to extirpation.

2.7.1.4 Conservation needs

The SSA dated February 2022 does not include conservation efforts for the LA pigtoe.

2.7.2 Environmental baseline

2.7.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

The LA pigtoe is only found in the Pearl River within the project area and a portion of the west Pearl.

2.7.2.2 Species conservation needs within the action area

The SSA dated February 2022 does not include conservation efforts for the LA pigtoe.

2.7.2.3 Habitat condition (general)

According to the February 2022 SSA, LA pigtoe occur in medium to large streams and rivers, requiring 1) flowing water of sufficient quantity and quality 2) adequate food supply, 3) habitat that provides refugia from both high- and low-flow events, 4) appropriate substrate that is generally characterized as stable and free of excessive fine sediment, 5) access to appropriate fish hosts, and 6) habitat connectivity (i.e., lack of impoundments and other barriers to fish pass).

Louisiana Pigtoe occurs in medium to large-sized streams and rivers in flowing waters (0.3-1.4 m/s) over substrates of cobble and rock or sand, gravel, cobble, and woody debris; they are often associated with riffle, run, and sometimes larger backwater tributary habitats (Ford et al. 2016, pp. 42, 52; Howells 2010a, p. 3-4; Williams et al. 2017b, p. 21).

2.7.2.4 Influences

Degraded water quality, altered hydrology, substrate changes, habitat fragmentation, direct mortality, invasive species, and climate change all influence the existence of the Louisiana LA pigtoe.

2.7.2.5 Additional baseline information

There is no additional baseline information.

2.7.3 Effects of the action

2.7.3.1 Indirect interactions

<u>Alternative C</u>

Indirect impacts due to changes in the velocity and water surface elevation are anticipated. The current lotic habitat would be replaced with a lentic habitat which would eliminate available habitat and host fishes. LA pigtoes downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation associated with construction activities. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the LA pigtoe. It is anticipated that downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result in a river bottom shift and would bury mussel beds which would then result in suffocation of individuals. The increase in turbidity and decreasing water quality would also impact potential host fishes. The population is in low condition and are therefore particularly vulnerable to extirpation.

CTO Alternative

Alternative CTO with a weir

It is assumed that, like Alternative C, construction of the CTO with weir would likely eliminate riverine habitat that the LA pigtoe depends on. For this draft a conservative approach is being taken and therefore this alternative would have the same indirect impacts as Alternative C if a weir is included. Velocity analysis, like that conducted for Alternative C, is being conducted to better understand the potential impact of the CTO on the riverine system.

Alternative CTO without a weir

If a weir is not included, indirect impacts would be due to changes to water velocity, water surface elevation, and water quality during high water events. This would not be much different from the current conditions during high water events and the impacts would be temporary and to a much lesser extent than with a weir.

2.7.3.2 Direct interactions

Alternative C

Direct impacts by way of death are anticipated due to implementation of Alternative C. Excavation of material from within the river over approximately two years would result in death of individuals as well as displacement of host fishes.

CTO Alternative

Alternative CTO with a weir and without a weir

Temporary direct impacts by the way of increased sedimentation and decreased water quality during construction activities. Burying of individuals or mussel beds is possible during this time of increased sedimentation. Best management practices would be implemented to minimize these impacts.

2.7.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.7.5 Discussion and conclusion

Alternative C and CTO with a weir

Based upon literature review and available survey data, and the effects of the action, the USACE has determined that implementation of Alternative C or CTO with a weir is likely to adversely affect but is not likely to jeopardize the continued existence of the Louisiana pigtoe.

Alternative CTO without a weir

Based upon literature review and available survey data, and the effects of the action, the USACE has determined that implementation of Alternative CTO without a weir may affect but is not likely to adversely affect the Louisiana pigtoe.

2.8 Monarch Butterfly (*Danaus plexippus)*

2.8.1 Status of the species

2.8.1.1 Legal status

The monarch butterfly is currently a candidate species.

2.8.1.2 Recovery plans

There are currently no recovery plans for the monarch butterfly. However, there is a SSA dated September 2020 (Annex D3).

2.8.1.3 Life history information

Adult monarch butterflies are large (3 to 4 inches) and conspicuous, with bright orange wings surrounded by a black border and covered with black veins. The black border has a double row of white spots, present on the upper side of the wings. Milkweed and flowering plants are needed for monarch habitat. Adult monarchs feed on the nectar of many flowers during breeding and migration, but they can only lay eggs on milkweed plants.

Migratory individuals in eastern North America predominantly fly south or southwest to mountainous overwintering grounds in central Mexico, and migratory individuals in western North America generally fly shorter distances south and west to overwintering groves along the California coast into northern Baja California (Solensky 2004).

The eastern population of monarchs overwinter in Mexico, where this microclimate is provided by forests primarily composed of oyamel fir trees (Abies religiosa). Migratory monarchs in the western population primarily overwinter in groves along the coast of California and Baja California which include blue gum eucalyptus (Eucalyptus globulus), Monterey pine (Pinus radiata), and Monterey cypress (Hesperocyparis macrocarpa) (Griffiths and Villablanca 2015).

Monarch butterflies are found throughout North America and are highly likely to utilize portions of the project area.

2.8.1.4 Conservation needs

The Species Status Assessment Report, version 2.1 dated September 2020 discusses conservation efforts for the monarch butterfly. Below is a brief summary. See Annex D3 for further details.

- Protection, restoration, enhancement, and creation of habitat is a central aspect of recent monarch conservation strategies.
- Improved management at overwintering sites in California has also been targeted to improve the status of western North American monarch butterflies (Pelton et al. 2019; WAFWA 2019).
- The Western Monarch Butterfly Conservation Plan which includes protecting and managing 50% of all currently known and active monarch overwintering sites, including 90% of the most important overwintering sites by 2029.
- Providing a minimum of 50,000 additional acres of monarch-friendly habitat in California's Central Valley and adjacent foothills by 2029.
- It also includes overwintering and breeding habitat conservation strategies, education and outreach strategies, and research and monitoring needs.

2.8.2 Environmental baseline

2.8.2.1 Species presence and use

Alternative C and CTO Alternative

Monarch butterflies are found throughout North America and are highly likely to utilize portions of the project area.

2.8.2.2 Species conservation needs within the action area

The Species Status Assessment Report, version 2.1 dated September 2020 discusses conservation needs. However, conservation needs within the action area have not been determined yet.

2.8.2.3 Habitat condition (general)

During migration to overwintering sites, monarchs need blooming nectar plants. On their return, monarchs are laying eggs and thus need both nectar sources and milkweed. The project area contains habitat that supports blooming nectar plants to potentially include milkweed.

2.8.2.4 Influences

Loss and degradation of habitat from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence, and incompatible management of overwintering sites in California, urban development, drought, exposure to insecticides, drought, and effects of climate change are all factors in the decline of the monarch population.

2.8.2.5 Additional baseline information

See Annex D3 Monarch “Pesticide Supplemental Material.” The following website also offers additional information https://www.fws.gov/initiative/pollinators/monarchs.

2.8.3 Effects of the action

2.8.3.1 Indirect interactions

<u>Alternative C</u>

Indirect impacts are expected due to the conversion of desired habitat to open water and elimination of food source.

<u>Alternative CTO</u>

Alternative CTO with a weir

Implementation of CTO with a weir would have the same indirect impacts as Alternative C but to a lesser degree,

Alternative CTO without a weir

Indirect impacts would be the potential benefit of providing suitable habitat for the monarch butterfly if the excavated areas are allowed to self-vegetate with wildflowers which would provide a desirable food source. If the excavated areas are mowed regularly and only allowed to self-vegetate with grass the indirect impact would be the conversion of desired habitat to grassy uplands and elimination of food source.

2.8.3.2 Direct interactions

<u>Alternative C</u>

Direct impacts could be anticipated by way of collision with construction equipment. Although collision with vehicles on nearby roadways is a regular occurrence, the construction activities could increase the number of individuals impacted. However, the species is highly mobile, and the equipment is rather slow moving, so it is expected that any individuals present could escape the impact.

<u>Alternative CTO</u>

Alternative CTO with a weir

Implementation of CTO with a weir would have the same direct impacts as Alternative C but to a lesser degree,

Alternative CTO without a weir

This alternative would have the same direct impacts as Alternative C as excavation and fill would still take place. However, the impacts would be to a lesser degree.

2.8.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.8.5 Discussion and conclusion

Alternative C

Based upon literature review and available survey data, and the effects of the action, the USACE has determined that, because of the conversion of such a large area of habitat to open water, implementation of Alternative C is likely to adversely affect but is not likely to jeopardize the continued existence of the monarch butterfly.

CTO Alternative

Alternative CTO with a weir

Based upon literature review and available survey data, and the effects of the action, the USACE has determined that implementation of CTO with a weir is likely to adversely affect but is not likely to jeopardize the continued existence of the monarch butterfly.

Alternative CTO without a weir

Based upon literature review and available survey data, and the effects of the action, the USACE has determined that implementation of CTO without a weir may affect but is not likely to adversely affect the monarch butterfly.

3 Critical Habitat Effects Analysis

Alternative C and CTO Alternative

On March 19, 2003, The USFWS and NMFS published the Final Rule in the Federal Register designating critical habitat for the Gulf sturgeon.

Primary consideration must be given to the physical and biological features (PBFs) of the habitat under review that are essential to the conservation of the species and that may require special management considerations or protection.

The PBFs essential for the conservation of the Gulf sturgeon populations include those habitat components that support feeding, resting, and sheltering, reproduction, migration, and physical features necessary for maintaining the natural processes that support these habitat components.

Based upon the identified PBFs for the Gulf sturgeon, the USFWS and NMFS identified a total of fourteen (14) Critical Habitat Units. Critical Habitat Unit 1 covers the project area and includes the Pearl River System in St. Tammany and Washington Parishes in Louisiana and Walthall, Hancock, Pearl River, Marion, Lawrence, Simpson, Copiah, Hinds, Rankin, and Pike Counties in Mississippi.

Of the 7 PBFs identified for Gulf sturgeon critical habitat, riverine spawning sites and riverine aggregation (resting) areas are not present in the action area. The PBFs found in the Action Area are food, flow regime, water quality, sediment quality, and migratory pathways.

The Pearl River is included in Critical Habitat Unit 1, the Pearl and Bogue Chitto Rivers in Louisiana and Mississippi, which is currently known to support a reproducing subpopulation of Gulf sturgeon. The Action Area occurs at the top extent of this Critical Habitat Unit.

While adult sturgeon does not usually feed in freshwater, juveniles forage extensively in rivers on aquatic insects, worms, and mollusks (Mason and Clugston 1993; Huff 1975; Sulak and Clugston 1999). with the varying aquatic species within the Action Area that feed on those types of prey it can be assumed that the area does contain enough of these prey items to support the populations of species that inhabit the area.

Suitable spawning substrate within the Pearl River likely includes soapstone, hard clay, gravel, and rubble areas and undercut banks adjacent to these substrates (W. Slack, pers. comm. 2001). Specific surveys have not been conducted on the substrate of the river within the Action Area; however, grab samples were taken as part of the Wetland Delineation conducted for the EIS/Feasibility Study that did not exhibit the suitable substrates necessary for sturgeon spawning in the Pearl River.

Gulf sturgeon depend on flow regimes in the riverine environment for all life stages including migration, breeding site selection, courtship, egg fertilization, resting and staging, and for maintaining spawning sites in the suitable condition needed for egg attachment, sheltering, resting, and larval staging. Based on average flow rates from 1966 to 2013, this area of the river currently has high flows during the springtime with flows decreasing significantly during the summer.

In 2019, a water advisory was issued for the Pearl River in Jackson due to continued discharges of sanitary sewer overflows into the river. In the Action Area, there is a former creosote plant as well as two former landfills from which debris periodically washes into the river. Leachates from these landfills were found to contain heavy metals above the regulatory standards. In 2003, the EPA also found barium, cobalt, zinc, and other contaminates in the river in the Action Area.

Migratory pathways are not expected to be impacted by the construction of Alternative C due to the construction of a fish passage.

Increased turbidity and sedimentation would lead to impacts on water quality, which then leads to impacts on the prey base for juvenile sturgeon. These impacts on water quality would be temporary and would be reduced through erosion control measures.

Changes to flow regime, water surface elevation, and water quality in the Action Area are anticipated. DO and temperature are important water quality factors for sturgeon. As temperature increases, DO levels decrease which can affect the growth and respiration rates of juvenile sturgeon. Water quality modeling conducted for temperature and DO indicate post-project levels would have a slight but not significant difference from the pre-project levels.

Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat.

4 Other Protected Species

Other protected species, specifically bald eagles, and migratory birds, have potential to be present in the study area. Bald eagles are protected under the Bald and Golden Eagle Protection Act (BGEPA) and the Migratory Bird Treaty Act (MBTA).

The bald eagle was near extinction approximately forty years ago throughout most of its range. Habitat destruction and degradation, illegal shooting, and the contamination of its food source, largely as a consequence of DDT, decimated the eagle population. However, the banning of DDT, habitat protection, and conservation measures through the ESA, have afforded a remarkable recovery for the species. The bald eagle was removed from the endangered species list in 2007 but continues to be protected under the BGEPA and the MBTA.

Many of the 1,093 species of birds protected under the Migratory Bird Treaty Act are experiencing population declines due to increased threats across the landscape. Millions of acres of bird habitat are lost or degraded every year due to development, agriculture, and forestry practices. In addition, millions of birds are directly killed by human-caused sources such as collisions with man-made structures such as windows and communication towers.

Bald eagles' nest in tall trees (usually cypress or pine in this area) near water and typically in the months of October through May. Migratory birds have varying nesting behaviors and seasons depending on the species. To be conservative, the nesting season for migratory birds is February 15 through September 15. Wading/water birds typically nest in trees or shrubs near water. Shorebirds typically nest on ground level in sand, small rocks, dunes, or ground vegetation. Many migratory birds (other than wading/water birds and shorebirds) are opportunistic nesters and will nest in trees, shrubs, building overhangs, house gutters, etc.

Alternative C and CTO Alternative

Direct impacts would be attributed to avoidance of the area during construction. Indirect impacts would be the elimination of potential roosting, foraging, and nesting habitat. Cumulative impacts, including both direct and indirect impacts of the alternative along with additional impacts from other, previous projects in the area are anticipated to be minor in intensity but long-term in duration. Impacts to the bald eagle and migratory birds from Alternative C would add to the impacts that have occurred over time and are expected to continue due to ongoing development and activities in and around the Project Area. A qualified biologist would survey the area prior to construction to determine the presence of nesting birds. If eagle nests are found in the project area, the USACE MVK would apply for an incidental eagle take permit and would implement avoidance and minimization measures described in the National Bald Eagle Management Guidelines until a permit with applicable requirements is received. Coordination with The Service and MDWFP would establish buffer zones and other guidelines to be implemented for nesting migratory birds depending on the species present. These impacts are considered insignificant.

5 Summary Discussion, Conclusion, And Effect Determinations

5.1 Effect Determination Summary

Alternative C

SPECIES (COMMON NAME)	SCIENTIFIC NAME	LISTING STATUS	PRESENT IN ACTION AREA	EFFECT DETERMINATION Alt C

Gulf Sturgeon	*Acipenser oxyrhynchus desotoi*	Threatened	Yes	LAA
Ringed Sawback Turtle	*Graptemys oculifera*	Threatened	Yes	LAA
Northern Long-eared Bat	*Myotis septentrionalis*	Endangered	Yes	NLAA
Pearl River map Turtle	*Graptemys pearlensis*	Proposed Threatened	Yes	LAA
Alligator Snapping Turtle	*Macrochelys temminckii*	Proposed Threatened	Yes	LAA
Louisiana pigtoe	*Pleurobema riddellii*	Proposed endangered	Yes	LAA
Tricolored bat	*Perimyotis subflavus)*	Proposed threatened	Yes	NLAA
Monarch Butterfly	*Danaus plexippus*	Candidate	Yes	LAA

LAA- Likely to adversely affect but not likely to jeopardize the continued existence of

NLAA- May affect but not likely to adversely affect

CTO with a weir

SPECIES (COMMON NAME)	SCIENTIFIC NAME	LISTING STATUS	PRESENT IN ACTION AREA	EFFECT DETERMINATION Alt C
Gulf Sturgeon	*Acipenser oxyrhynchus desotoi*	Threatened	Yes	LAA
Ringed Sawback Turtle	*Graptemys oculifera*	Threatened	Yes	LAA
Northern Long-eared Bat	*Myotis septentrionalis*	Endangered	Yes	NLAA
Pearl River map Turtle	*Graptemys pearlensis*	Proposed Threatened	Yes	LAA
Alligator Snapping Turtle	*Macrochelys temminckii*	Proposed Threatened	Yes	LAA
Louisiana pigtoe	*Pleurobema riddellii*	Proposed endangered	Yes	LAA
Tricolored bat	*Perimyotis subflavus)*	Proposed threatened	Yes	NLAA

Monarch Butterfly	*Danaus plexippus*	Candidate	Yes	LAA

CTO without a weir

SPECIES (COMMON NAME)	SCIENTIFIC NAME	LISTING STATUS	PRESENT IN ACTION AREA	EFFECT DETERMINATION Alt C
Gulf Sturgeon	*Acipenser oxyrhynchus desotoi*	Threatened	Yes	NLAA
Ringed Sawback Turtle	*Graptemys oculifera*	Threatened	Yes	LAA
Northern Long-eared Bat	*Myotis septentrionalis*	Endangered	Yes	NLAA
Pearl River map Turtle	*Graptemys pearlensis*	Proposed Threatened	Yes	LAA
Alligator Snapping Turtle	*Macrochelys temminckii*	Proposed Threatened	Yes	LAA
Louisiana pigtoe	*Pleurobema riddellii*	Proposed endangered	Yes	NLAA
Tricolored bat	*Perimyotis subflavus)*	Proposed threatened	Yes	NLAA
Monarch Butterfly	*Danaus plexippus*	Candidate	Yes	NLAA

5.2 Summary Discussion

Threatened and Endangered species and other protected species known to occur in the action area include GS, ringed map turtle, NLEB, PRMT, AST, LA pigtoe, TCB, monarch butterfly, bald eagle, and migratory birds. GS critical habitat also occurs within the action area.

Alternative C and CTO with a weir would cause both temporary direct and long-term indirect impacts to species discussed. The project would eliminate and/or degrade habitat for GS, ringed map turtle, PRMT, the LA pigtoe, and the monarch butterfly; would eliminate potential habitat for both bat species; and would potentially create preferred habitat for the AST. Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat. Bald eagles and migratory birds could be impacted temporarily due to construction activities and long-term due to elimination of nesting and roosting habitat.

Alternative CTO without a weir would cause both temporary direct and indirect impacts to species discussed. The project would degrade habitat for the monarch butterfly and would eliminate potential habitat for both bat species. There would be temporary direct and indirect impacts to GS, all three turtle species, and the LA pigtoe. Based upon the assessment completed, it was determined that Alternative CTO with a weir would not result in an adverse modification to Gulf sturgeon critical habitat. Bald eagles and migratory birds could be impacted temporarily due to construction activities and long-term due to elimination of nesting and roosting habitat.

5.3 Conclusion

ESA consultation is ongoing. Based on currently available historical data, a review of current literature and studies, and with the employment of avoidance measures, the USACE has determined that Alternative C and CTO with a weir may affect but would not likely adversely affect the NLEB and the TCB; would likely adversely affect the GS, ringed map turtle, AST, PRMT, LA pigtoe, and monarch butterfly. Alternative CTO without a weir may affect but would not likely adversely affect the GS, NLEB, TCB, LA pigtoe, and monarch butterfly; would likely adversely affect the ringed map turtle, AST, and PRMT. Based upon the assessment completed, it was determined that Alternative C, CTO with a weir, and CTO without a weir would not result in an adverse modification to Gulf sturgeon critical habitat.

References

Amelon, S., and D. Burhans. 2006. Conservation assessment: *Myotis septentrionalis* (northern long-eared bat) in the eastern United States. Pages 69–82 in Conservation assessments for five forest bat species in the eastern United States, Thompson, F. R., III, editor. U.S. Department of Agriculture, Forest Service, North Central Research Station, General Technical Report NC-260. St. Paul, Minnesota.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. The University Press of Kentucky, Lexington, Kentucky.

Bennett, M.G., B.R. Kuhajda, J.H. Howell. 2008. Status of the frecklebelly madtom, Notorus munitus, (*Siluriformes: Ictaluridae*): a review of data from field surveys, museum records and the literature. Southeastern Naturalist.

Blute, G., M. Carrie and G. Blouin-Demers. 2010. Impact of recreational power boating on two populations of northern map turtles (*Graptemys geographica*). Aquatic Conservation Marine and Freshwater Ecosystems.

Buckner, M.M., W. Smith, J.A. Takats. 2002. Tennessee, Cumberland, and Mobile River Basins at Risk. A Biological Assessment and Vision for the World Wildlife Fund's Southeast Rivers and Streams Project. Nashville, TN. Available online: www.worldwildlife.org/what/wherewework/sers/WWFBinaryitem2738.pdf <Last accessed June 2020>

Buhlmann, K. and G. Coffman. 2001. Fire Ant Predation of Turtle Nests and Implications for the Strategy of Delayed Emergence. Journal of the Elisha Mitchell Scientific Society.

Buhlmann, K.A. 2014. Assessment of ringed map turtle (*Graptemys oculifera*) and Pearl map turtle (*Graptemys pearlensis*) abundance and habitat on the U.S. Navy's Stennis Western Maneuver Area East Pearl and Mike's Rivers, Mississippi, with recommendations for habitat enhancement. Unpublished report submitted to Stennis Space Center, Stennis Western Maneuver Area, Mississippi and Naval Facilities Engineering Command, Jacksonville, FL.

Cagle, F.R. 1952. The status of the turtles Graptemys pulchra Baur and Graptemys barbourin Carr and Marchand, with notes of their natural history. Copeia.

Carr, J.L., S.R. Holcomb, and M.J. Ray. 2011. Basking in the alligator snapping turtle, *Macrochelys temminckii* (Testudines: *Chelydridae*). IRCF Reptiles and Amphibians

Carter, T.C. and G.A. Feldhamer. 2005. Roost tree use by maternity colonies of Indiana bats and northern long-eared bats in southern Illinois. Forest Ecology and Management.

Chaffin, K., T.M. Norton, K. Gilardi, R. Poppenga, J.B. Jensen, P. Moler, C. Cray, E.S. Dierenfeld, T. Chen, M. Oliva, F.C. Origgi, S. Gibbs, L. Mazzaro, and J. Mazet. 2008.

Health assessment of free–ranging alligator snapping turtles (*Machrochelys temminckii*) in Georgia and Florida. Journal of Wildlife Diseases.

Chen, Tien-Hsi and Kuang-Yang Lue. 2009. Changes in the Population Structure and Diet of the Chinese Stripe-Necked Turtle (*Mauremys sinensis)* Inhabiting a Disturbed River in Northern Taiwan. Zoological Studies.

Cheong, T. S., M. L. Kavvas, and E. K. Anderson. 2006. Evaluation of Adult White Sturgeon Swimming Capabilities and Applications to Fishway Design. Envrion. Biol Fish.

Craft, N.M., B. Russell, and S. Travis. 2001. Identification of Gulf sturgeon spawning habitats and migratory patterns in the Yellow and Escambia River systems. Final Report to the Florida Marine Research Institute, Fish and Wildlife Conservation Commission.

DeCatanzaro, R., and P. Chow-Fraser. 2010. Relationship of road density and marsh condition to turtle assemblage characteristics in the Laurentian Great Lakes. Journal of Great Lakes Research.

DeWan, A., N. Dubois, K. Theoharides, and J. Boshoven. 2010. Understanding the impacts of climate change on fish and wildlife in North Carolina. Defenders of Wildlife Report, Washington D.C.

Dickerson D.D. and K.J. Reine. 1996. Habitat assessment and relative abundance estimates for the ringed sawback turtle (*Graptemys oculifera*) in dredging sites of the West Pearl River navigation project (Louisiana). Unpublished report. U.S. Army Engineer District, Vicksburg, Mississippi. 21 pp. + tables and figures.

Edwards, R.E., K.J. Sulak, M.T. Randall, and C.B. Grimes. 2003. Movements of Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in nearshore habitat as determined by acoustic telemetry. Gulf of Mexico Science.

Edwards, R.E., R.E., Parauka, F.M. and K.J. Sulak. 2007. New insights into marine migration and winter habitat of Gulf sturgeon. in: J. Munro, D. Hatin, J. Hightower, K. Sulak, and F. Caron (eds.). Proceedings of the Symposium on Anadromous Sturgeons. American Fisheries Society, Symposium, Bethesda, Maryland.

Ennen, J.R., J.E. Lovich, and R.L. Jones. 2016. *Graptemys pearlensis* Ennen, Lovich, Selman, and Qualls 2010 – Pearl River Map Turtle. In: Rhodin, A.G.J., Pritchard, P.C.J., van Dijk, P.P., Saumure, R.A., Buhlmann, K.A., Iverson, J.B., and Mittermeier, R.A. (Eds.). Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group, Chelonian Research Monographs.

Fitzgerald, L.A., and R.E. Nelson. 2011. Thermal biology and temperature-based habitat selection in a large aquatic ectotherm, the alligator snapping turtle, *Macroclemys temminckii*. Journal of Thermal Biology

Folkerts, G.W. 1997. State and fate of the world's aquatic fauna. Pgs. 1-16 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA.

Ford, N.B., L. Williams, M.G. Williams, J. Banta, J. Placyk, and H. Hawley. 2016. Final report: endangered species research projects for freshwater mussels, Region 2, East Texas for Texas Comptroller of Public Accounts. University of Texas at Tyler, Tyler, Texas.

Foster, A.M. and J.P. Clugston. 1997. Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society.

Foster, R.W., and A. Kurta. 1999. Roosting ecology of the Northern bat (*Myotis septentrionalis*) and comparisons with the endangered Indiana bat (*Myotis sodalis*). Journal of Mammalogy.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2000. Gulf sturgeon spawning migration and habitat in the Choctawhatchee River system. Alabama-Florida. Transactions of the American Fisheries Society129:811-826.

Griffiths J, Villablanca F. 2015. Managing monarch butterfly overwintering groves: Making room among the eucalyptus. California Fish and Game

Heise, R.J., S.T. Ross, M.F. Cashner, and W.T. Slack. 1999a. Movement and habitat use of the Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in the Pascagoula drainage of Mississippi: year 3. Museum Technical Report No. 74. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 14.

Howells, R.G. 2010a. Louisiana pigtoe (*Pleurobema riddellii*): summary of selected biological and ecological data for Texas. Biostudies, Kerrville, Texas.

Howey, C.A.F., and S.A. Dinkelacker. 2009. Habitat selection of the alligator snapping turtle (*Macrochelys temminckii*) in Arkansas. Journal of Herpetology.

Huff, J.A. 1975. Life History of the Gulf of Mexico Sturgeon, *Acipenser oxyrhynchus desotoi* in Suwannee River, Florida. Mar. Res. Publ. No. 16.

Hyzy, B.A., R.E. Russel, A. Silvis, W.M. Ford, J. Riddle, and K. Russel. 2020. Investigating maternity roost selection by northern long-eared bats at three sites in Wisconsin. Endangered Species Research.

Jackson, C.G., Jr., and A. Ross. 1971. The occurrence of barnacles on alligator snapping turtle, *Macroclemys temminckii* (Troost). Journal of Herpetology.

Jones, R. 2006. Reproduction and Nesting of the Endangered Ringed Map Turtle, *Graptemys oculifera*, in Mississippi. Chelonian Conservation and Biology.

Jones, R. 2017. Long-Term Trends in Ringed Sawback (*Graptemys oculifera*) Growth, Survivorship, Sex Ratios, and Population Sizes in the Pearl River, Mississippi. Chelonian Conservation and Biology.

Lindeman, P.V. 1998. Of deadwood and map turtles (*Graptemys*): An analysis of species status for five species in three river drainages using replicated spotting-scope counts of basking turtles. Linnaeus Fund research report. Chelonian Conservation and Biology.

Lindeman, P.V. 2013. The Map Turtle and Sawback Atlas: Ecology, Evolution, Distribution, and Conservation. University of Oklahoma Press, Norman, OK.

Lindeman, P.V. 2019. Population status of the megacephalic map turtles *Graptemys pearlensis* and *Graptemys gibbonsi*. Unpublished Final Report submitted to U.S. Fish and Wildlife Service, Jackson, MS.

Lorio, W. (ed.). 2000. Proceedings of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) status of the subspecies. Workshop held September 13-14, 2000, Mississippi State University, Science and Technology Research Center, Stennis Space Center, MS. 200 pp. and appendices.

Lovich, J.E., W. Selman, and C.J. McCoy. 2009. *Graptemys gibbonsi* Lovich and McCoy 1992. Pascagoula Map Turtle, Pearl River Map Turtle, Gibbon’s Map Turtle. Chelonian Research Monographs.

Marchant, S.R. and M.K. Shutters. 1996. Artificial substrates collect Gulf sturgeon eggs. North American Journal of Fisheries Management.

Mason, W.T. Jr. and J.P. Clugston. 1993. Foods of the Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society

Mississippi Museum of Natural Science. 2015. Mississippi State Wildlife Action Plan. Mississippi Department of Wildlife, Fisheries, and Parks, Mississippi Museum of Natural Science, Jackson, Mississippi.

Moore, M. V., M. L. Pace, J. R. Mather, P. S. Murdoch, R. W. Howarth, C. L. Folt, C. Y. Chen, H. F. Hemond, P. A. Flebbe, and C. T. Driscoll. 1997. Potential effects of climate change on freshwater ecosystems of the New England/Mid-Atlantic Coastal Region, Climate Research.

Morrow, J.V, K.J. Killgore, J.P. Kirk, and H.E. Rogillio. 1996. Distribution and population attributes of Gulf sturgeon in the lower Pearl River System, Louisiana. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Mulholland, P.J. and D.R. Lenat. 1992. Streams of the southeastern piedmont, Atlantic drainage. In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY.

Neves, R.J., A.E. Bogan, J.D. Williams, S.A. Ahlstedt, P.W. Hartfield. 1997. Status of aquatic mollusks in the southeastern United States: a downward spiral of diversity. Pgs. 43-85 In: Benz, G.W. and D.E. Collins (editors). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. Available online: http://www.sherpaguides.com/southeast/aquatic_fauna/chapter_3/ <Last accessed July 2020>

Owen, S.F., M.A. Menzel, W.M. Ford, J.W. Edwards, B.R. Chapman, K.V. Miller, and P.B. Wood. 2002. Roost tree selection by maternal colonies of Northern long-eared Myotis in an intensively managed forest. USDA Forest Service. Newtown Square, Pennsylvania. 10 pp.

Parkyn, D.C., D.J. Murie, J.E. Harris, D.E. Colle, and J.D. Holloway. 2007. Seasonal movements of Gulf of Mexico sturgeon in the Suwannee River and estuary. American Fisheries Society Symposium.

Perry, R.W., and R.E. Thill. 2007. Roost selection by male and female northern long-eared bats in a pine-dominated landscape. Forest Ecology and Management.

Pine, W.E., H.J. Flowers, K.G. Johnson, and M.L. Jones. 2006. An assessment of Gulf sturgeon movement, spawning site selection, and post-spawn holding areas in the Apalachicola River, Florida. Final Report submitted to the Florida Fish and Wildlife Conservation Commission. University of Florida, Gainesville, FL.

Plummer, M., and N. Mills. 2008. Structure of an urban population of softshell turtles (*Apalone spinifera*) before and after severe stream alteration. Herpetological Conservation.

Pritchard, P.C.H. 1989. The alligator snapping turtle: Biology and conservation. Milwaukee Public Museum, Milwaukee, WI, USA.

Ross, S.T., R.J. Heise, W.T. Slack, J.A. Ewing, III, and M. Dugo. 2000. Movement and habitat use of the Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in the Pascagoula drainage of Mississippi: year 4. Mississippi Department of Wildlife, Fisheries, and Parks and Museum of Natural Science. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 15.

Sasse, D.B., and P.J. Pekins. 1996. Summer roosting ecology of northern long-eared bats (*Myotis septentrionalis*) in the white mountain national forest. Bats and Forests Symposium October 1995, Victoria, British Columbia, Canada.

Selman, W. and C. Qualls. 2007. Distribution, status, and conservation of the Pascagoula map turtle (*Graptemys gibbonsi*). Unpublished report submitted to U.S. Fish and Wildlife Service, Jackson, MS.

Selman, W., C. Qualls, and J. Owen. 2013. Effects of Human Disturbance on the Behavior and Physiology of an Imperiled Freshwater Turtle. The Journal of Wildlife Management.

Selman, W. and P.V. Lindeman. 2015. Life history and ecology of the Pascagoula map turtle (*Graptemys gibbonsi*). Herpetological Conservation and Biology .

Selman, W. and R. Jones. 2017. Population Structure, Status, and Conservation of Two Graptemys Species from the Pearl River, Mississippi. Journal of Herpetology, 51(1).

Selman, W. 2018. Diamonds in the rough: status of two imperiled Graptemys species (*Graptemys oculifera* and *G. peralensis*) in the Pearl River of Jackson, MS.

Silvis, A., R.W. Perry, and W.M. Ford. 2016. Relationships of three species of bats impacted by White-Nose Syndrome to forest condition and management. Gen. Tech. Rep. SRS–214. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Smith, G., J. Iverson and J. Rettig. 2018. Frequency of propeller damage in a turtle community in a northern Indiana, USA, lake; a long-term study. Herpetological Conservation and Biology.

Soballe, D.M., B.L. Kimmel, R.H. Kennedy, and R.F. Gaugush. 1992. Reservoirs. Pgs. 421-474 In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY.

Solensky MJ. 2004. Overview of monarch migration. Pp. 79-83 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Sulak, K.J. and J.P. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desoto*i, in the Suwannee River, Florida, USA: a synopsis. Journal of Applied Ichthyology.

Sulak, K.J., M. Randall, J. Clugston, and W.H. Clark. 2004. Critical spawning habitat, early life history requirements, and other life history and population aspects of the Gulf sturgeon in the Suwannee River. Final Report to the Florida Fish and Wildlife Conservation Commission, Nongame Wildlife Program. U.S. Geological Survey, Gainesville, FL.

Sulak, K.J., M. T. Randall, R. E. Edwards, T. M. Summers, K. E. Luke, W. T. Smith, A. D. Norem, W. M. Harden, R. H. Lukens, F. Parauka, S. Bolden, and R. Lehnert. 2009.

Timpone, J.C., J.G. Boyles, K.L. Murray, D.P. Aubrey, and L.W. Robbins. 2010. Overlap in roosting habits of Indiana Bats (*Myotis sodalis*) and northern bats (*Myotis septentrionalis*). American Midland Naturalist.

USFWS. 2006. Status assessment. Pascagoula map turtle – *Graptemys gibbonsi.* Unpublished report, Mississippi Ecological Services Field Office, Jackson, Mississippi.

USFWS. 2011. Bogue Chitto National Wildlife Refuge Comprehensive Conservation Plan. Southeast Region, Atlanta, Georgia.

U.S. Fish and Wildlife Service. 2021. Species status assessment report for the Pearl River Map Turtle (*Graptemys pearlensis*), Version 1.1. April 2021. Atlanta, GA.

U.S. Navy. 2011. Integrated Natural Resources Management Plan. Naval Construction Battalion Center Gulfport, Stennis Western Maneuver Area, Stennis Space Center, Mississippi.

Vučenović, J. and P. V. Lindeman. 2020. The diets of the Pearl and Pascagoula Map Turtles (*Graptemys pearlensis* and *Graptemys gibbonsi*).

White R.G. and B. Mefford. 2002. Assessment of behavior and swimming ability of Yellowstone River sturgeon for design of fish passage devices. Montana Cooperative Fishery Research Unit, Montana State University; and Water Resources Research Laboratory, Bureau of Reclamation, Denver. Prepared for the Omaha District Corps of Engineers.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of the freshwater mussels of the United States and Canada. Fisheries.

Williams, J.D., A.E. Bogan, R.S. Butler, K.S. Cummings, J.T. Garner, J.L. Harris, N.A. Johnson, and G.T. Watters. 2017a. A revised list of the freshwater mussels (Mollusca: Bivalvia: Unionida) of the United States and Canada. Freshwater Mollusk Biology and Conservation.

Winston, M.R., C.M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. Transactions of the American Fisheries Society.

Wirth W., L. Schwarzkopf, L.F. Skerratt, and E. Ariel. 2018. Ranaviruses and reptiles. PeerJ 6:e6083 https://doi.org/10.7717/peerj.6083

Pearl River Basin, Mississippi, Federal Flood Risk Management Project

Annex D1 - Prior Coordination

June 2024

DEPARTMENT OF THE ARMY
U.S. ARMY CORPS OF ENGINEERS, VICKSBURG DISTRICT
4155 CLAY STREET
VICKSBURG, MS 39183-3435

April 1, 2024

Regional Planning and Environment Division South

Project Name: Pearl River Basin, Mississippi, Federal Flood Risk Management Project Hinds and Rankin Counties, MS

Mr. James Austin
Field Supervisor
Mississippi Field Office
U.S. Fish and Wildlife Service
6578 Dogwood View Parkway
Jackson, MS 39213

Dear Mr. Austin,

The U.S. Army Corps of Engineers (USACE), Vicksburg District has received your letter dated February 13, 2024 in response to our biological assessment (BA) and letter dated January 22, 2024 requesting formal consultation under Section 7 of the Endangered Species Act (ESA) of 1973 (87 Stat. 884, as amended; 16 U.S.C. 1531 et seq.) on the potential effects of the Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, Mississippi.

The USACE thanks you for your review and comments and has revised the BA to reflect consistent determinations for the alligator snapping turtle and the northern long-eared bat and has removed the 4(d)-rule language as the 4(d) rule has since been nullified. The USACE has also made other revisions per further informal coordination with the Service. Additionally, responses to the Service's comments are attached below.

The USACE is preparing a revised BA as re-initiation of ESA consultation is necessary since a combination thereof alternative is being identified and will be assessed in the draft EIS. A request to re-initiate formal consultation and conference pursuant to Section 7 of the ESA of 1973, as amended (16 U.S.C. § 1536), and the consultation procedures at 50 C.F.R. Part 402 will be submitted to the Service once enough details are available for ESA assessment.

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Eric M. Williams
Chief, Environmental Planning Branch

While the Service concurs that the proposed alternative C would result in an increase in aquatic habitat within the project area, that increase is a result of conversion of riverine, stream, slough, and forested wetland habitat to a lake-like system (as the BA mentions multiple times). Even though water discharge may be maintained within the system, it will not provide the habitat required for those species needing a riverine environment to survive and would result in the subsequent loss of those species within the action area. Also, temporal changes in sedimentation could have long-term impacts to species. A sedimentation analysis would be helpful in determining long-term impacts to habitat and species.

> Response: The USACE has committed to conducting sedimentation analysis during pre-construction engineering and design (PED).

The Service agrees that construction could lead to death of individuals if they are unable to flee the construction area. The BA notes that this is especially relevant for young of species and freshwater mussels. However, this is also relevant to inactive turtles during the winter months.

> Response: Thank you for the clarification. Please provide this information in the biological opinion (BO) as well.

The BA mentions a fish passage to be constructed around the large weir proposed in Alternative C. However, it's unclear if velocities within this conceptual design will be appropriate for species to migrate successfully. Would suitable habitat still exist upstream of this passage to support the Gulf sturgeon's life history strategies?

> Response: The USACE has committed to coordinating closely with the Service during design of the fish passage so as to ensure appropriate velocities for migration. It is anticipated that the fish passage would allow for continued migration upstream.

The BA notes that Alternative C would result in temporary and negligible changes to water quality and temperature. However, the Service believes that removal of riparian buffer, reduced flows, and larger surface area will result in higher temperatures compared to that of a riverine system. Further, nutrient loading and algae/macrophyte blooms could reduce dissolved oxygen over time if current water quality issues are not resolved. Also, the BA states that impacts could occur from reduced water quality (page 39-40); water quality advisory had been issued due to sewage; and there are dumpsites that would be inundated (page 50). A water quality analysis would be helpful in determining the degree of impacts and effects to species.

> Response: The BA has been revised to include consistent analysis throughout. Water quality analysis would be conducted during PED if required per the Service's BO.

The BA states that Action Area includes portions of the Pearl River to 1.6 miles downstream of the proposed weir at RM 284. However, the Service considers potential impacts that may occur downstream due to the action, possibly beyond this identified boundary.

> Response: Action area has been revised to reflect what is in the current EIS instead of what was in the 2019 BO.

If there's 287 acres of Pearl River within the Project Area, why does the BA only mention 230 acres of potential habitat for the Gulf sturgeon (page 17) and only 207 acres of open water impacted by excavation (page 18)?

> Response: Existing data was used during development of the current BA. 287 acres is from NFI mitigation plan, 230 acres is from the previous BA and BO, and 207 acres from previous BA and BO. GS CH is approximately 287 acres with approximately 230 acres of that estimated to be of use to the GS based on 2019 BA and BO. The current BA has been revised for clarity.

The ringed map turtle is a sponge specialist that also consumes insects and occasionally mollusks and is not "wholly carnivorous" (Lindeman et al. 2024).

> Response: Thank you for the clarification. Please provide this information in the BO as well.

Ringed map turtles are also found in the Strong River and several other tributaries.

Response: Thank you for the clarification. Please provide this information in the BO as well.

There is 2.5 miles of suitable habitat that could be directly impacted by dredging upstream of LeFleur's Bluff State Park. Also, this location partially overlaps the only ringed map turtle population that is currently stable or increasing in size (estimated 728 individuals (range 574-1,475)). There is also a significant amount of basking structure from LeFleur's Bluff State Park north to upper project area, and throughout other portions of the action area.

Response: Thank you for the clarification. Please provide this information in the BO as well.

The BA states that the Corps is unaware of any future state, tribal, local or private non-Federal actions unrelated to the proposed action that are reasonably certain to occur in the Action Area. However, the BA also states here that the riparian zone would be eliminated, and development is planned for most of the areas of fill surrounding the improved channel.

Response: Perhaps "planned" was a poor choice of words. BA has been revised to state "development is likely…".

Ringed map turtle eggs relocation: the Service recommends continued collaboration on this action, since egg incubation and timed release could be more successful option. Further, the Service has discussed issues with creating sandbars, placing debris, and enforcement of regulations.

Response: The USACE will continue to coordinate with the Service throughout the planning and construction phases.

The NLEB is a nocturnal species. Thus, avoiding daytime construction activities may be difficult. Will there be surveys of bridges and culverts that are planned for replacement? As mentioned, the 4(D) rule has been nullified due to the uplisting of this species.

Response: Construction other than clearing of trees can be avoided by the NLEB. Surveys would be conducted if required per the Service's BO.

May have mixed ringed map turtle with Pearl River map turtle. Also, ringed map turtles are not considered a "generalist" species. The Service would need more information regarding downstream impacts (i.e., sedimentation), before determining if "habitat immediately downstream of the weir would remain suitable for species." Water quality statements aren't verified on reproduction and young. Also, Pearl River map turtles aren't found throughout the Ross Barnett Reservoir, only at the upper areas where riverine characteristics persist.

Response: The species terminology in the BA has been revised for clarity. Thank you for the clarification. Please provide this information in the BO as well. Understood, the USACE has committed to conducting sedimentation analysis during PED.

Alligator snapping turtles are somewhat selective of nesting sites: usually within 20 m of water (average nest distance of 12 m from water source), >1 m above water surface, in an area with partially open canopy cover. There is high quality habitat in the 2.5 miles around LeFleur's Bluff State Park and north to Ross Barnett Reservoir (RBR).

Response: Thank you for the clarification. Please provide this information in the BO as well.

Reinvasion potential for LA pigtoe is low, as the upstream population left below RBR would be separated by a lake making it difficult for their small-bodied host fish to cross. Chance of reinvasion from the west Pearl is low due to the distance for host fish to migrate. Also, mention of the burying of mussel beds is significant. For several species (i.e., Pearl River map turtle, and to a lesser extent, alligator snapping turtle), burying of mussel beds downstream of the weir could directly impact the prey source for these species.

Response: Thank you for the clarification. Please provide this information in the BO as well.

Flow regime will likely be impacted (even if discharge is maintained, impacts to velocity will occur) due to conversion to a lake-like environment. Migratory pathways could be impacted if the fish passage doesn't function as necessary. Sedimentation was also mentioned in the mussel section to have significant downstream impacts. The BA states that sedimentation would be impacted and would lead to water quality impacts.

Response: The BA has been revised to reflect consistent analysis.

Discrepancies in table determinations, text within the BA, and letter.

Response: Revisions have been made to address inconsistencies.

United States Department of the Interior

FISH AND WILDLIFE SERVICE
Mississippi Ecological Services Field Office
6578 Dogwood View Parkway, Suite A
Jackson, Mississippi 39213
Phone: (601)965-4900 Fax: (601)965-4340

February 13, 2024

IN REPLY REFER TO:
2022-0006708E

Eric Williams, Chief
Environmental Planning Branch
U.S. Army Corps of Engineers
7400 Leake Avenue
New Orleans, Louisiana 70118

Dear Mr. Williams:

The U.S. Fish and Wildlife Service (Service) acknowledges receipt of your January 22, 2024, biological assessment (BA) and letter requesting formal consultation under Section 7 of the Endangered Species Act of 1973 (87 Stat. 884, as amended; 16 U.S.C. 1531 et seq.) on the potential effects of the Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, Mississippi. The U.S. Army Corps of Engineers (Corps), Vicksburg District, is requesting concurrence for their determination of effects to the alligator snapping turtle (*Macrochelys temminckii*) and the northern long-eared bat (*Myotis septentrionalis;* NLEB); and formal conference on the proposed listing of the tricolored bat (*Perimyotis subflavus*), Louisiana pigtoe (*Pleurobema riddellii*), and the Candidate species, Monarch butterfly (*Danaus plexippus*). They also request formal consultation for the Pearl River map turtle (*Graptemys pearlensis*), ringed map turtle (*Graptemys oculifera*), Gulf sturgeon (*Acipenser oxyrhynchus desotoi*), and its Critical Habitat. Our comments are submitted under the authority of the Endangered Species Act (ESA) of 1973 (87 Stat. 884, as amended; 16 U.S.C. 1531 et seq.).

The Corps states that the purpose of the project is to reduce flood risk in the Jackson metropolitan area, reduce the flood risk of critical infrastructure, including the Savanna Street Wastewater Treatment Facility, and improve access to transportation routes, evacuation routes, and critical care facilities during flood events. The Service previously determined that Alternative C (channel excavation plan, including construction of a large weir) would be the most ecologically damaging of those presented. Thus, the BA focuses on impacts to listed and other protected species from Alternative C. Both A1 and a newly developed CTO (combination thereof) alternative are mentioned, but assumed to have less severe environmental impacts without construction of the weir. Although the BA focusses on Alternative C, the Corps is still considering both A1 and a "yet to be defined" CTO alternative, and will re-initiate consultation if a CTO is selected, or if a weir is included in the selected alternative for implementation. The

Corps committed to conducting further analyses (i.e., velocity and sedimentation analyses within the Pearl River at the project site and downstream) during pre-construction and design to accurately determine impacts to riverine species. After receiving the necessary information, the Service will provide associated recommendations. We will continue to cooperate with the Corps in providing information and collaboration as necessary for any future developments regarding Alternative A1 and CTO.

Before we proceed with impact assessments and recommendations, the Service requests the Corps review their determinations for the alligator snapping turtle and the Northern long-eared bat. Although the letter states the proposed project may affect, but is not likely to adversely affect the alligator snapping turtle, the BA describes both direct and indirect impacts to the species and conversely determined that the project is likely to adversely affect the alligator snapping turtle. Furthermore, the letter states the project may affect, but is not likely to adversely affect the northern long-eared bat. However, the BA states that the project would remove potential roosting and foraging habitat and could result in potential adverse effects to the species. The Corps was correct in stating that the 4(d) rule did not prohibit incidental take due to otherwise lawful activities, such as forest clearing. However, the 4(d) rule was nullified when the northern long-eared bat was uplisted to endangered in November 2022. Therefore, any incidental take of this species is prohibited and would require appropriate permits.

The Service, via informal consultation, can provide information regarding potential impacts, species surveys, and any other information needed to help with your analysis. The Service requests clarification regarding species determinations, particularly for that of the alligator snapping turtle and the northern long-eared bat. We'll also provide additional comments and concerns to your BA in Appendix A (attached below).

If you have any questions, please contact Tamara Campbell in our office, telephone: (601) 321-1138, email: tamara_campbell@fws.gov.

Sincerely,

JAMES AUSTIN

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Date: 2024.02.13 14:24:13 -06'00'

James Austin
Field Supervisor
Mississippi Field Office

Appendix A

Specific comments on the Biological Assessment

To reduce redundancy and the length of this appendix the Service has tried to identify the first occurrence of a statement for which we have a comment, but do not continue to identify subsequent occurrences of similar statements. However, this doesn't mean that our comments are restricted to just the statement identified but are applicable to all similar statements within the document. In addition, our comments are primarily focused on the discussions regarding Alternative C as this is currently the focus of the Corps' BA, but with potential for A1 or a CTO (combination thereof).

Page 7: While the Service concurs that the proposed alternative C would result in an increase in aquatic habitat within the project area, that increase is a result of conversion of riverine, stream, slough, and forested wetland habitat to a lake-like system (as the BA mentions multiple times). Even though water discharge may be maintained within the system, it will not provide the habitat required for those species needing a riverine environment to survive and would result in the subsequent loss of those species within the action area. Also, temporal changes in sedimentation could have long-term impacts to species. A sedimentation analysis would be helpful in determining long-term impacts to habitat and species.

Page 8: The Service agrees that construction could lead to death of individuals if they are unable to flee the construction area. The BA notes that this is especially relevant for young of species and freshwater mussels. However, this is also relevant to inactive turtles during the winter months.

Page 8: The BA mentions a fish passage to be constructed around the large weir proposed in Alternative C. However, it's unclear if velocities within this conceptual design will be appropriate for species to migrate successfully. Would suitable habitat still exist upstream of this passage to support the Gulf sturgeon's life history strategies?

Page 9: The BA notes that Alternative C would result in temporary and negligible changes to water quality and temperature. However, the Service believes that removal of riparian buffer, reduced flows, and larger surface area will result in higher temperatures compared to that of a riverine system. Further, nutrient loading and algae/macrophyte blooms could reduce dissolved oxygen over time if current water quality issues are not resolved. Also, the BA states that impacts could occur from reduced water quality (page 39-40); water quality advisory had been issued due to sewage; and there are dumpsites that would be inundated (page 50). A water quality analysis would be helpful in determining the degree of impacts and effects to species.

Page 11: The BA states that Action Area includes portions of the Pearl River to 1.6 miles downstream of the proposed weir at RM 284. However, the Service considers potential impacts that may occur downstream due to the action, possibly beyond this identified boundary.

Page 17 and 18: If there's 287 acres of Pearl River within the Project Area, why does the BA only mention 230 acres of potential habitat for the Gulf sturgeon (page 17) and only 207 acres of open water impacted by excavation (page 18)?

Page 20: The ringed map turtle is a sponge specialist that also consumes insects and occasionally mollusks, and is not "wholly carnivorous" (Lindeman et al. 2024).

Page 21: Ringed map turtles are also found in the Strong River and several other tributaries.

Page 22: There is 2.5 miles of suitable habitat that could be directly impacted by dredging upstream of LeFleur's Bluff State Park. Also, this location partially overlaps the only ringed map turtle population that is currently stable or increasing in size (estimated 728 individuals (range 574-1,475)). There is also a significant amount of basking structure from LeFleur's Bluff State Park north to upper project area, and throughout other portions of the action area.

Page 23: The BA states that the Corps is unaware of any future state, tribal, local or private non-Federal actions unrelated to the proposed action that are reasonably certain to occur in the Action Area. However, the BA also states here that the riparian zone would be eliminated, and development is planned for most of the areas of fill surrounding the improved channel.

Page 25: Ringed map turtle eggs relocation: the Service recommends continued collaboration on this action, since egg incubation and timed release could be more successful option. Further, the Service has discussed issues with creating sandbars, placing debris, and enforcement of regulations.

Page 28, NLEB is a nocturnal species. Thus, avoiding daytime construction activities may be difficult. Will there be surveys of bridges and culverts that are planned for replacement? As mentioned, the 4(D) rule has been nullified due to the uplisting of this species.

Page 33 – 34: May have mixed ringed map turtle with Pearl River map turtle. Also, ringed map turtles are not considered a "generalist" species. The Service would need more information regarding downstream impacts (i.e., sedimentation), before determining if "habitat immediately downstream of the weir would remain suitable for species." Water quality statements aren't verified on reproduction and young. Also, Pearl River map turtles aren't found throughout the Ross Barnett Reservoir, only at the upper areas where riverine characteristics persist.

Page 36: Alligator snapping turtles are somewhat selective of nesting sites: usually within 20 m of water (average nest distance of 12 m from water source), >1 m above water surface, in an area with partially open canopy cover. There is high quality habitat in the 2.5 miles around LeFleur's Bluff State Park and north to Ross Barnett Reservoir (RBR).

Page 45: Reinvasion potential for LA pigtoe is low, as the upstream population left below RBR would be separated by a lake making it difficult for their small-bodied host fish to cross. Chance of reinvasion from the west Pearl is low due to the distance for host fish to migrate. Also, mention of the burying of mussel beds is significant. For several species (i.e., Pearl River map turtle, and to a lesser extent, alligator snapping turtle), burying of mussel beds downstream of the weir could directly impact the prey source for these species.

Page 51: Flow regime will likely be impacted (even if discharge is maintained, impacts to velocity will occur) due to conversion to a lake-like environment. Migratory pathways could be impacted if the fish passage doesn't function as necessary. Sedimentation was also mentioned in the mussel section to have significant downstream impacts. The BA states that sedimentation would be impacted and would lead to water quality impacts.

Discrepancies in table determinations, text within the BA, and letter.

DEPARTMENT OF THE ARMY
U.S. ARMY CORPS OF ENGINEERS, VICKSBURG DISTRICT
4155 CLAY STREET
VICKSBURG, MS 39183-3435

January 22, 2024

Regional Planning and Environment Division South

Project Name: Pearl River Basin, Mississippi, Federal Flood Risk Management Project Hinds and Rankin Counties, MS

Mr. James Austin
Field Supervisor
Mississippi Field Office
U.S. Fish and Wildlife Service
6578 Dogwood View Parkway
Jackson, MS 39213

Dear Mr. Austin,

The U.S. Army Corps of Engineers (USACE), Vicksburg District (CEMVK) has prepared this Biological Assessment (BA) to evaluate the potential impacts associated with the proposed flood risk management project, Pearl River Basin, Mississippi, Federal Flood Risk Management Project Hinds and Rankin Counties, MS. This BA provides the information required pursuant to the Endangered Species Act (ESA) and implementing regulation (50 CFR 402.13), to comply with the ESA.

The project is located in portions of Hinds and Rankin Counties, Mississippi within what is referred to as the City of Jackson Metropolitan Area. The project area begins at river mile (RM) 293.5 and extends southward on either side of the Pearl River channel, to a point approximately 3.0 miles south of U.S. Interstate 20 to RM 284.0.

The purpose of the project is to reduce flood risk in the Jackson metropolitan area; reduce the flood risk of critical infrastructure, including the Savanna Street Wastewater Treatment Facility; and improve access to transportation routes, evacuation routes, and critical care facilities during flood events.

Based on currently available historical data, a review of current literature and studies, and with the employment of avoidance measures, the USACE has determined that the project may affect but would not likely adversely affect the northern long eared bat, and alligator snapping turtle; would likely adversely affect but would not jeopardize the continued existence of the Gulf sturgeon, ringed sawback turtle, Louisiana pigtoe, monarch butterfly, Pearl River map turtle, and tricolored bat. Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat.

CEMVK is submitting this BA as a request to initiate formal consultation and conference pursuant to Section 7 of the ESA of 1973, as amended (16 U.S.C. § 1536), and the consultation procedures at 50 C.F.R. Part 402.

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Eric M. Williams
Chief, Environmental Planning Branch

Pearl River Basin, Mississippi
Federal Flood Risk
Management Project
Hinds and Rankin Counties, MS
THREATENED and ENDANGERED SPECIES
PRELIMINARY DRAFT BIOLOGICAL ASSESSMENT

January 2024

The purpose of this Preliminary Draft Biological Assessment (BA) is to assess the effects of the Pearl River Flood Risk Management project and determine whether the project may affect any Federally threatened, endangered, proposed or candidate species. This BA is being prepared in accordance with legal requirements set forth under Section 7 of the Endangered Species Act (16 U.S.C. 1536 (c)).

Annexes

Annex D1 Prior Consultation

2019 Threatened and Endangered Species Biological Opinion

Annex D2 IPaC report and USFWS email confirmation

Annex D3 Species Recovery Plans and Status Assessment Reports

Gulf Sturgeon Recovery/Management Plan

Ringed Sawback Turtle Recovery Plan

Species Status Assessment Report for the Northern long-eared bat

Species Status Assessment Report for the Pearl River Map Turtle

Species Status Assessment Report for the Alligator Snapping Turtle

Species Status Assessment Report for the Louisiana pigtoe

Species Status Assessment Report for the tricolor bat

Species Status Assessment Report for the monarch butterfly

1 Description of The Action

1.1 Project Name

Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, MS (PR FRM)

1.2 Introduction

A Biological Assessment (BA) completed in 2019 as a part of the environmental review process for the Pearl River Basin, Mississippi Federal Flood Risk Management Project, Hinds and Rankin Counties, MS Draft Feasibility Study/Environmental Impact Statement (FS/EIS) included a review of literature and other pertinent scientific data, interviews, and coordination efforts for each of the mentioned threatened and endangered (T & E) species as originally identified by the U.S. Fish and Wildlife Service (The Service). The 2019 BA and Biological Opinion (BO) are incorporated by reference into this BA and were used during the preparation of this document.

A search on The Services' Information for Planning and Consulting (IPaC) site, conducted on March 21, 2023, resulted in a list of species that should be considered when assessing the impacts of this project. That list includes the Gulf sturgeon, ringed sawback (Ringed map) turtle, Northern long-eared bat, Pearl River map turtle, alligator snapping turtle, and monarch butterfly. Email correspondence with The Service dated March 21, 2023, confirmed this list, and concluded that the monarch butterfly, as a candidate species, has no legal regulations under the Endangered Species Act. However, On April 21, 2023, email correspondence with the Service stated that they had been informed that they expect a listing decision on the monarch butterfly in the near future. Therefore, the USACE has decided to include the monarch butterfly in this BA. On April 10, 2023, the Service informed USACE via email (attached) that the Louisiana pigtoe and the tricolored bat had been recently proposed for listing. Therefore, those two species will also be discussed in this BA. Additionally, other protected species, specifically the bald eagle and migratory birds are discussed in order to obtain compliance with the Bald and Golden Eagle Protection Act and the Migratory Bird Treaty Act.

The Service requested velocity and sedimentation analysis be conducted within the Pearl River at the project site and downstream. USACE has committed to conducting this analysis during pre-construction engineering and design (PED) if the weir is included in the selected alternative for implementation. That being said, re-initiation of ESA consultation will be necessary during PED to accurately determine impacts to riverine species.

This BA focusses on impacts to listed and other protected species due to potential implementation of Alternative C. However, a combination thereof alternative (CTO) is also being considered but has yet to be defined. This BA includes a high level, qualitative assessment of that potential alternative on listed species. USACE will reinitiate consultation if a CTO is selected for implementation.

1.3 Project Description

Alternative A1 consists of elevating and floodproofing residential and non-residential structures within the future 100-year stage.

Alternative C consists of the construction of channel improvements, demolition of the exiting weir near the J. H. Fewell WTP site and construction of a new weir with a low-flow gate structure further downstream for water supply to be continued while simultaneously creating an area of surface water for recreational opportunities, Federal levee improvements (excavated material plan), and upgrading an existing non-Federal ring levee with slurry wall around the Savannah Street WWTP. Construction of the project would require relocations and/or improvements to various public and private utilities and infrastructure, mitigating potential HTRW and other hazardous waste sites within the floodplain, avoidance and minimization measures required under the ESA, and the creation of new habitat mitigation areas to offset losses within the project's construction footprint areas. The project has the potential to impact historic properties.

The CTO Alternative has yet to be identified. However, the following measures, in any combination, may be considered for this alternative:

- Alternative A1
- Excavation of Main Channel, Federal levee improvements
- Demo of existing weir, Construction of new weir, Fish passage
- Non-federal levee improvements (Savannah Street WWTP)
- Clean out and sustained maintenance of tributaries.
- Levee setbacks
- Small-scale levees
- Bridge modifications (major evaluation will be a PED effort and not part of the EIS).
- Mitigation features (Impact assessment will be conducted in subsequent NEPA document(s).

See Appendix I of the IES for details of each alternative.

1.3.1 Location

The project is located in portions of Hinds and Rankin Counties, Mississippi within what is referred to as the City of Jackson Metropolitan Area. The project area begins at river mile (RM) 293.5 and extends southward on either side of the Pearl River channel, to a point approximately 3.0 miles south of U.S. Interstate 20 to RM 284.0.

1.3.2 Description of project habitat

The Alternative C Project area consists of numerous habitat types to include the following. It is reasonable to assume that the habitat within the CTO would be very similar, if not the same, as that for Alternative C.

Table 1. Habitat Types and Acres Present

Habitat Type	Acres
Emergent wetland	59.19 acres
Lacustrine	200.9 acres
Mixed forested wetland	911.58 acres
Mixed Scrub-Shrub Wetland	256.04 acres
Palustrine	147.20 acres
Riverine	287.16 acres
Upland Evergreen Forest	14.44 acres
Upland Grassland	151.79 acres
Upland Mixed Forest	536.47 acres
Upland Pasture	54.41 acres
Upland Scrub-Shrub	208.68 acres
Upland Urban	29.60 acres

1.3.3 Project proponent information

Requesting Agency

DEPT OF DEFENSE (DOD)

Army Corps of Engineers (COE)

Tammy Gilmore
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Lead agency

Same as Requesting Agency.

1.3.4 Project purpose

The purpose of the PR FRM Project is to reduce flood risk in the Jackson metropolitan area; reduce the flood risk of critical infrastructure, including the Savanna Street Wastewater Treatment Facility; and improve access to transportation routes, evacuation routes, and critical care facilities during flood events.

1.3.5 Project type and deconstruction

Currently the project is at approximately 20% design and therefore many of the components below have not yet been determined.

Project timeline and sequencing - TBD

Site preparation – Prior to construction

Construction access and staging – Prior to, concurrent with, and post construction

Post-project site restoration – Post construction

Conservation and compensation activities (both on- and off-site) – Prior to, concurrent with, and post construction

1.3.6 Anticipated environmental stressors

1.3.6.1 Animal Features

Alternative A-1

Existing terrestrial wildlife habitat and the wildlife resources within the immediate area would be directly impacted by way of avoidance due to construction activities and associated noise. It is anticipated that the areas where there are currently existing structures would self-vegetate once the structures are removed. This newly developed habitat would support some terrestrial species.

Alternative C

Existing terrestrial wildlife habitat and the wildlife resources within the Project Area would be directly impacted by the removal of approximately 2,069 acres of terrestrial habitat that currently exists within the project area. Though the existing terrestrial habitats would be removed, the conversion to aquatic habitats in these areas would be utilized by other wildlife species.

Impacts to aquatic and fisheries resources associated with sedimentation poses a risk. Best Management Practices will be implemented to reduce this risk. The potential for sedimentation during construction could adversely affect food sources for aquatic species. However, this impact would be temporary. It is anticipated that overall available aquatic and fisheries habitat would increase as a result of the channel improvements, with the total area available for aquatic and fish habitat estimated at 1,692 acres, post-construction. However, the current riverine system would be replaced with a lake system. Riverine obligates (fish, mussels, turtles, etc.) will not use lakes or could not survive under such conditions.

Disturbance from excavation and placement of material from within and adjacent to the river over approximately two years could result in death of individuals if they are unable to flee the construction work area. This is especially relevant to young of species and freshwater mussels.

Mitigation measures, including habitat restoration activities, would be implemented to offset the intensity of these impacts during and after the construction activities are completed. A fish passage will be created around the relocated weir which will increase the possibility for migrating aquatic species to utilize the Project Area.

<u>CTO Alternative</u>

While the specific features of the CTO alternative have not been determined, this alternative could have similar impacts in the long-term compared to Alternative C if a new weir is constructed, but fewer impacts than Alternative C if a new weir is not constructed. Without a weir the impact to wildlife supported by floodplain areas in the Project Area would be less than Alternative C as there would not be a significant reduction in available habitat and habitat quality.

Even without the weir, excavation for channel improvements and levee work would result in both permanent and temporary loss of habitat and associated resources. As much as 255 acres of impact from levees and 374 acres from channel improvements would displace wildlife on a temporary basis and for some on a permanent basis. It is expected that there would be displacement of wildlife due to noise and activity in the area, even if lands are not directly impacted.

In addition to the impacts described above, any tributary structural features that are included in this alternative could potentially have an impact on wildlife, depending on the location and extent of the feature.

1.3.6.2 Aquatic Features

<u>Alternative A-1</u>

There would be no impacts to aquatic features due to implementation of Alternative A-1.

<u>Alternative C</u>

A total of approximately 1,861 acres of wetlands and "other waters of the U.S." would be impacted by Alternative C. Approximately 487 acres of existing surface water bodies within the channel improvement footprint, including the Pearl River channel itself and its tributaries, would be impacted by this alternative. Additional direct impacts to 116.95 acres of water bodies would be anticipated by the filling activities within the dredge disposal areas.

Indirect impacts to adjacent waterbodies within the project area could be anticipated through the implementation of Alternative C. Existing interconnections to adjoining waterbodies could be affected and existing inflow and outflow functions within the areas could also be affected.

A 1,692-acre lake would be created post construction. This would increase the available aquatic features within the project area. However, the current riverine system would be replaced with a lake system. Riverine obligates (fish, mussels, turtles, etc.) will not use lakes or could not survive under such conditions.

Alternative CTO

While the specific features of the CTO alternative have not been determined, this alternative is expected to have fewer impacts in the long-term compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed. Without a new weir, the area would maintain riverine characteristics. The existing weir would continue to pose an impediment to migratory patterns of aquatic species as do the multiple low-head dams/sills downstream of the Project Area.

1.3.6.3 Environmental Quality Features

Alternative A-1

There would be no impacts to water quality with the implementation of Alternative A-1. Construction activities would increase suspended particles (dust) into the air during construction. This would cause temporary and minimal impacts to air quality.

Alternative C

There would be temporary impacts to water quality locally in the Pearl River during construction. The greater volume of water in the impoundment would reduce temperature variations and result in dissolved oxygen concentrations that are slightly lower but still meeting water quality standards. Increases in productivity would also occur but growth of algae and macrophytes would continue to be light-limited. There could be negligible changes to temperature, dissolved oxygen (DO), ultimate carbonaceous biochemical oxygen demand (CBODU), total nitrogen (TN), ammonia-nitrogen (NH3-N), nitrate-nitrite (NOX), organic nitrogen (Org-N), total phosphorus (TP), orthophosphate (PO4), organic phosphorus (Org-P), phytoplankton chlorophyll-a, and total suspended solids (TSS). Overall, there would be temporary short-term, adverse impacts to water quality both during and for a short time following construction.

The impacts to the air quality within the Project Area as a result of the implementation of Alternative C would be short-term, minor, adverse impacts during the construction period only.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have fewer impacts in the long-term compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

1.3.6.4 Landform (topographic) Features

Alternative A-1

There would be no impacts to landform features due to implementation of Alternative A-1.

Alternative C

The current topographic features in the project area include the Pearl River, natural ridges, Native American earthworks/mounds, existing levees, and agricultural fields. Approximately 1,700 acres would be degraded to elevations lower than existing. Additionally, the disposal areas would result in increased elevations within those areas.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have fewer impacts in the long-term compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

1.3.6.5 Soil and Sediment

Alternative A-1

Any potential for impacts to soils would be temporary in nature and would occur only during the period of construction during the elevation activities, demolition, and/or relocation activities.

Alternative C

Approximately 2,557 acres of existing soils within the project area would be removed and placed in the designated disposal areas. Indirect impacts to soils within the Project Area could be anticipated because of ongoing operations and associated maintenance through the life of the project. There is potential for increased sedimentation in the river from the channel excavation. Best Management Practices will be implemented to reduce this.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have fewer impacts in the long-term compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

1.4 Action Area

The Action Area consists of the Pearl River floodplain from the Ross Barnett Dam to just south of Byram and includes land in Madison, Rankin, and Hinds Counties, Mississippi. The study area is drained by several small creeks that are tributaries of the Pearl River. Small tributaries to the Pearl River within the Action Area include Town, Hanging Moss, Eubanks, Lynch, Richland, Hardy, Caney, Purple, and Hog Creeks.

The Action Area includes the portion of the Pearl River from the Ross Barnett spillway (RM 301.77) to 1.6 miles downstream of the proposed project weir at RM 284 (Figure1). The Action Area also includes riparian areas adjacent to the river where construction activities will occur. The Action Area extends upstream of the proposed project to include all river miles that will be impacted by altered flow regimes, at approximately RM 301.77. The Action Area extends downstream (approximately 1.6 miles) of the proposed impoundment.

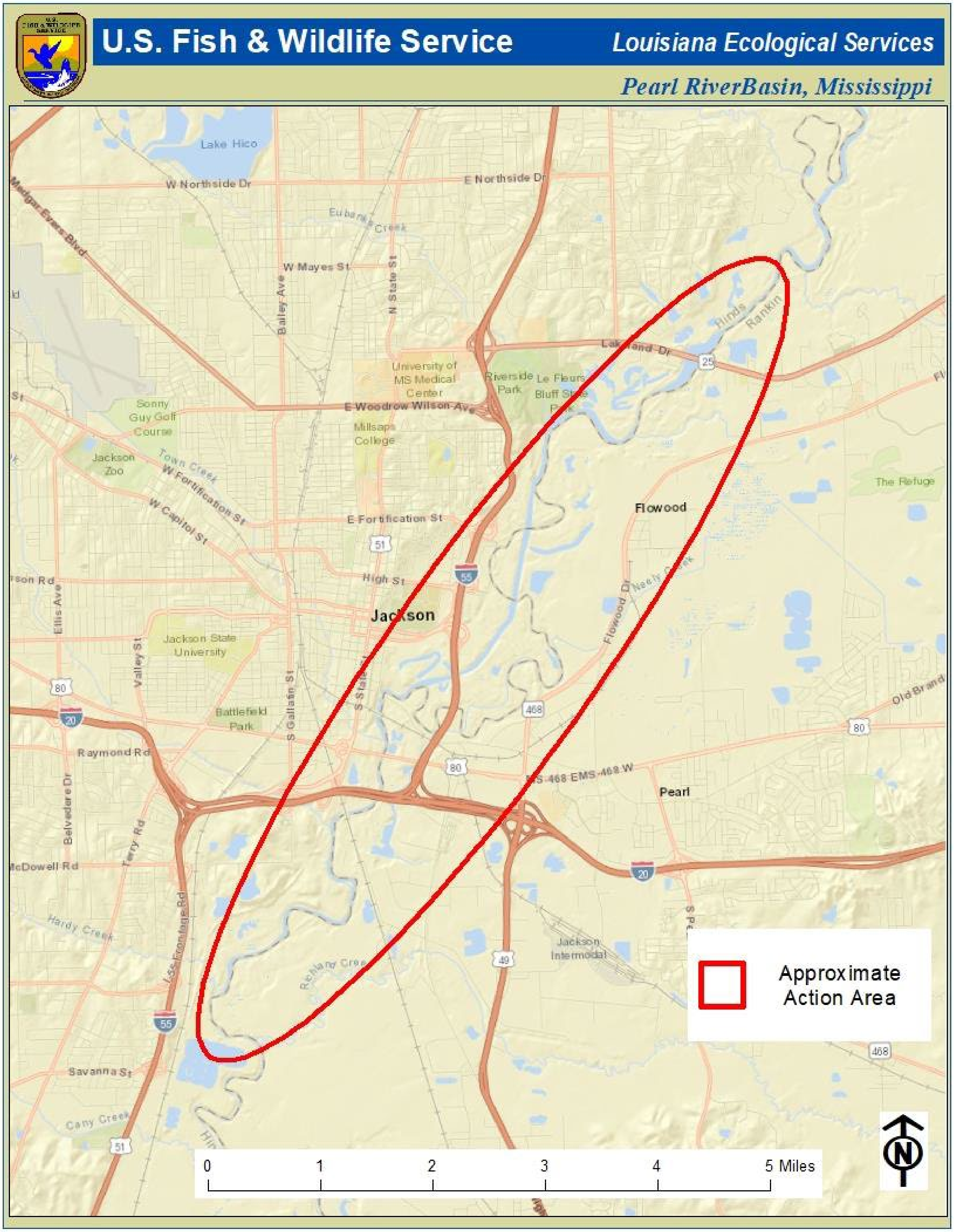


Figure 1 Action Area as Determined in the 2019 BO

1.5 Conservation Measures

Alternative A1 Not applicable

Alternative C and CTO

Conservation measures have not been identified as of yet. USACE is coordinating closely with the Service to develop conservation measures for each species as needed.

1.6 Prior Consultation History

The BA completed as a part of the environmental review process for the Draft FS/ESI included a review of literature and other pertinent scientific data, interviews and coordination efforts for each of the aforementioned T & E species as originally identified by the USFWS in correspondence dated June 8, 2004 and follow up coordination and listing reviews from 2015 through 2018. The listed species covered under that BA included the threatened Gulf sturgeon, the threatened Ringed Sawback (Ringed map) Turtle, the threatened Northern long-eared bat and the threatened Wood stork. It also includes a review of the listing information for the recently listed Pearl Darter. Though no longer listed, it also includes a review and assessment of both the American bald eagle and the Louisiana black bear.

The most recent consultation was completed on Oct 23, 2019, when The Service rendered a Biological Opinion (BO) (FWS Log #: 04EL1000- 2020-F-0109) that included an Incidental Take Statement requiring the USACE to implement reasonable and prudent measures that the Service considers necessary or appropriate to minimize the impacts of anticipated take on the ringed ringed map turtle (*Graptemys oculifera*) and Gulf sturgeon (*Acipenser oxyrhynchus desotoi*). Since this opinion was issued, new data has revealed the presence of gulf sturgeon within the action area (Michael Andres, personal communication, January 12, 2023). In a letter dated Jan 18, 2023, the Service recommended the draft EIS reflect these recent findings, since the previous draft EIS and biological assessment incorrectly stated that gulf sturgeon were not likely to migrate into the project area for potential spawning. In addition, since only a conceptual design was provided for the fish passage channel in 2019, additional coordination is required with the Service to ensure Gulf sturgeon can successfully pass the new weir which will be considerably larger than the existing weir (2019 BO, Gulf Sturgeon RPM #2). Re-initiation of formal consultation will be required as the designs of the proposed alternatives and associated mitigation plans are currently not finalized. Development of the designs and mitigation plans will be coordinated with The Service.

1.7 Other Agency Partners and Interested Parties

- Rankin County, Hinds County

- Environmental Protection Agency (EPA) Region 4
- Mississippi Department of Environmental Quality (MDEQ)
- Federal Emergency Management Agency (FEMA) Region IV
- Mississippi Department of Wildlife Fisheries and Parks (MDWFP)
- Mississippi Department of Marine Resources (MDMR)
- Mississippi Natural Resources Conservation Service (MNRCS)
- LA Department of Wildlife and Fisheries (LDWF)
- LA Department of Environmental Quality (LDEQ)
- Louisiana Department of Natural Resources (LDNR)
- Louisiana Coastal Protection and Restoration Authority (CPRA)
- U.S. Fish and Wildlife Service Jackson District
- USFWS Lafayette District
- Mississippi Department of Archives & History

1.8 Other Reports and Helpful Information

- 2019 Rankin Hinds Pearl River Flood and Drainage Control District Threatened and Endangered Species Biological Assessment
- 2019 USFWS Threatened and Endangered Species Biological Opinion
- Gulf Sturgeon Recovery/Management Plan
- Ringed Sawback Turtle Recovery Plan
- Species Status Assessment Report for the Northern long-eared bat
- Species Status Assessment Report for the Pearl River map Turtle
- Species Status Assessment Report for the Alligator Snapping Turtle
- Species Status Assessment Report for the Louisiana pigtoe
- Species Status Assessment Report for the tricolored bat

2 Species Effects Analysis

Alternative A1 is not expected to impact any of the listed species in the area and therefore will not be discussed further.

2.1 Gulf sturgeon (*Acipenser oxyrhynchus desotoi*)

2.1.1 Status of the species

2.1.1.1 Legal status

The Gulf Sturgeon is listed as Threatened under the Endangered Species Act (Federal Register Vol. 56, No. 189, September 30, 1991)

2.1.1.2 Recovery plans

The most recent recovery plan available for the Gulf sturgeon is dated September 1995 (Annex D3).

2.1.1.3 Life history information

The Gulf sturgeon is an anadromous fish (ascending rivers from the sea for breeding) that have historically inhabited coastal rivers from the Mississippi in Louisiana to the Tampa Bay in Florida. The Gulf sturgeon is one (1) of two (2) geographically dispersed subspecies of the Atlantic Sturgeon (*Acipenser oxyrhynchus*).

The Gulf sturgeon is characterized by a sub-cylindrical body that is imbedded with bony plates or “scutes”. The snout of the fish is greatly extended and bladelike and includes four (4) fleshy barbells in front of the mouth. The upper lobe of the tail is longer than the lower lobe. Adult specimens generally range in size from 1.8 to 2.4 meters (m) or six (6) to eight (8) feet in length. They are typically light brown to dark brown in color but are known to vary in color from grayish brown to bluish black on their back and sides, grading to white on their belly.

Age at sexual maturity ranges from 8 to 12 years for females and 7 to 9 years for males (Huff 1975). The Gulf sturgeon is a long-lived species, with some individuals reaching at least 42 years in age (Huff 1975).

The feeding habits of the Gulf sturgeon vary, depending upon the fish’s age (i.e., young-of-year, juvenile, sub-adult, adult) and is closely associated with migration and spawning habits. Throughout fall and winter, juveniles feed in the lower salinity areas in the river mouth and estuary (Sulak and Clugston 1999; Sulak et al. 2009), while subadults and adults migrate and feed in the estuaries and nearshore Gulf of Mexico habitat (Foster 1993; Foster and Clugston 1997; Edwards et al. 2003, 2007; Parkyn et al. 2007). Some Gulf sturgeon may also forage in the open Gulf of Mexico (Edwards et al. 2003).

The Gulf sturgeon typically inhabits the coastal rivers of the Gulf of Mexico during the warmer months of the year and generally overwinters in estuaries and bay environments within the Gulf of Mexico. The adults move into the tributary rivers for spawning in the spring and return to the Gulf waters in the fall. Spawning occurs in the upper reaches of rivers, at least 100 km (62 miles) upstream of the river mouth (Sulak et al. 2004), in habitats consisting of one or more of the following: limestone bluffs and outcroppings, cobble, limestone bedrock covered with gravel and small cobble, gravel, and sand (Marchant and Shutters 1996; Sulak and Clugston 1999; Heise et al. 1999a; Fox et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006). These hard bottom substrates are required for egg adherence and shelter for developing larvae (Sulak and Clugston 1998). Documented spawning depths range from 1.4 to 7.9 m (4.6 to 26 ft) (Fox et al. 2000; Ross et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006).

Further details on life history can be found in Annex D1.

2.1.1.4 Conservation needs

There are currently no conservation plans for the Gulf sturgeon. However, there is a Recovery Plan dated 1995 that includes an outline for recovery actions addressing threats to the Gulf sturgeon. Below are the main objectives. See Appendix D1 for further details.

- Determine essential ecosystems, identify essential habitats, assess population status, and refine life history investigations in management unit rivers.
- Protect individuals, populations, and their habitats.
- Coordinate and facilitate exchange of information on Gulf sturgeon conservation and recovery activities.

2.1.2 Environmental baseline

2.1.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

Recent studies for the Gulf sturgeon have not been conducted in this reach of the Pearl River and survey data from this area is not prevalent; however, in 2021, a Gulf sturgeon was detected above the waterworks sill in LeFleur's Bluff State Park and in 2022 the same sturgeon was detected closer to the spillway of Ross Barnet (Michael J. Andres, Ph.D., pers. comm. January 12, 2023). There are also unconfirmed sightings of Gulf sturgeon as far upstream as the City of Jackson, Mississippi, in Hinds County which is within the Action Area (Morrow et. al. 1996; Lorio 2000; Slack, pers. comm. 2002). There have been 24 Gulf sturgeon captured by commercial fishermen, eight of which being captured within the Action Area and the most recent of those captures occurring, a juvenile, in 2008.

The potential spawning habitat in the project area is believed to be minimal and significantly degraded due to the urbanization within the area and past flood control efforts. With the inclusion of the fish passage around the relocated weir, any adverse effects to potential spawning habitat thought to be associated with construction of the project would be minimized. There is no documented evidence that spawning activities occur within the project area. If there is spawning upstream of the weir, the presence of a reservoir without flow does cause large issues for juvenile fish. Post-hatching the sturgeon larvae go into a drift phase and if the velocities slow into a lake-like setting they will likely not survive. They are not great swimmers at this life stage and if they end up in a reservoir they can be exposed to excess predators among other threats.

2.1.2.2 Species conservation needs within the action area

There are currently no conservation plans for the Gulf sturgeon. However, there is a Recovery Plan dated 1995 that includes an outline for recovery actions addressing threats to the Gulf sturgeon. Below are the objectives that might be applicable to the action area. See Annex D3 for further details.

- Survey, monitor, and model populations.
- Reduce or eliminate unauthorized take.
- Identify and eliminate known or potentially harmful chemical contaminants, and water quantity and water quality problems which could impede recovery of Gulf sturgeon.
- Restore, enhance, and provide access to essential habitats.

2.1.2.3 Habitat condition (general)

Alternative C and CTO Alternative

The extent of potential habitat for the Gulf sturgeon, within the project area, is estimated to be approximately 230.0 acres (~9.5 linear miles) within the confines of the existing Pearl River channel from just north of U.S. Highway 25 southward to the proposed weir location south of U.S. Interstate 20. Bedrock and limestone outcroppings that are typical of Gulf sturgeon spawning areas in other river systems do not occur here. However, within the Pearl River drainage, spawning areas likely include soapstone, hard clay, gravel and rubble areas, and undercut banks adjacent to these substrates (W. Slack, pers. comm. 2001).

2.1.2.4 Influences

Over- fishing, associated with the commercial uses, resulted in a significant decline in Gulf sturgeon numbers throughout most of the 20th century. Incidental catch of Gulf sturgeon in other fisheries occurred at significant levels during the same time periods. Habitat losses associated with the construction of water control structures including dams and sills along the Gulf of Mexico drainage basins have contributed to a decline in populations throughout the historic range. Dam construction in several of the rivers has severely restricted the sturgeon's access to historic migration routes and spawning areas. Water quality such as pollution, temperature, and dissolved oxygen levels are also a threat.

2.1.2.5 Additional baseline information

There is no additional baseline information.

2.1.3 Effects of the action

2.1.3.1 Indirect interactions

Alternative C

Until a vegetative cover is established along the excavated areas, all disturbed areas would be subject to erosion. This could potentially cause excess sediment to flow downstream approximately 1.6 miles south of the construction area and erosion could be exacerbated in that area until the riverbank has stabilized. The turbidity would be additive to any downstream riverbank erosion resulting from sediments being trapped behind the weir after its construction. Increased sediment and turbidity can result in decreased light penetration and decreased photosynthesis. Production of benthic organisms also can be reduced by high levels of sediment.

With the construction of the 1,500-foot-wide weir structure and resulting impoundment from the weir, changes to the velocity and water surface elevation would occur within the Action Area. The weir has been designed to match the current discharge of the river; therefore, there should not be significant change in discharge after the target area has filled to the top of the weir. The migratory blockage caused by the weir structure could impact the sturgeon's ability to swim north of the structure unless there are high water events; however, a fish passage channel has been included as part of the project design to minimize the impacts on aquatic species migration. Flow conditions would need to meet the needs of the species to allow for navigation of the passage. These conditions include water velocity that does not exceed the sturgeon's swim speed and enough water flow levels for the species to be able to swim through it.

Studies have shown that Gulf sturgeon cannot swim against currents greater than 1 to 2 meters per second (mps) (3 to 6 fps). Studies on fish passage attraction speed flow has shown that the recommended flow should be between 2 and 4 fps with sustained swim speed ranges for sturgeon to be in the range of 3 to 4 fps (Cheong et al. 2006; White and Mefford 2002). At this time, there is only a conceptual model of the fish passage channel, approximately 1.4 miles long of a curving channel, with the possible velocities ranging anywhere from 1 to 7 fps. The optimal velocities of 2 to 4 fps will be considered during detail design of the fish passage. Velocity analysis is needed to determine the indirect impacts to GS due to construction of the fish passage.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.1.3.2 Direct interactions

Alternative C

Approximately 207.7 acres of open water would be impacted by the channel excavation. While the construction activities are being conducted, the disturbance to the sediment would increase the turbidity in the river. Increased sediment and turbidity can result in decreased light penetration and decreased photosynthesis.

High levels of sediment can settle on fish spawning areas and smother fish eggs and larvae. Sediments can settle on respiratory surfaces of fish and aquatic organisms and interfere with respiration. The increased sedimentation and turbidity in the river from the channel excavation and levee relocation would have impacts on the macroinvertebrate prey for any juvenile Gulf sturgeon that would be temporarily feeding in the Action Area.

<u>CTO Alternative</u>

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.1.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.1.5 Discussion and conclusion

<u>Alternative C</u>

To offset or reduce impacts to Gulf sturgeon, a 1-mile fish passage would be constructed at the location of the new weir. This would allow for migration upriver that would have otherwise been cut off due to the weir.

Based upon literature review, available survey data, the current status of the species, the environmental baseline for the action area, and the effects of the action, the USACE has determined that implementation of Alternative C is likely to adversely affect but is not likely to jeopardize the continued existence of, the Gulf sturgeon.

<u>CTO Alternative</u>

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.2 Ringed Sawback (ringed map) Turtle (*Graptemys oculifera)*

2.2.1 Status of the species

2.2.1.1 Legal status

The ringed sawback turtle is listed as threatened under the Endangered Species Act (Federal Register Vol. 51, No. 246, December 23, 1986)

2.2.1.2 Recovery plans

The most recent recovery plan available for the ringed map turtle is dated April 1988. (Annex D3)

2.2.1.3 Life history information

The ringed map turtle is a small (7.5 to 22 cm) narrow-headed turtle with laterally compressed, black, spine-like vertebral projections and a slightly serrated posterior carapacial margin. The carapace is dark olive-green and each pleural has a broad yellow or orange circular mark.

The ringed map turtle is a wholly carnivorous species, with insects and mollusks constituting their principal diet. In addition, they are also thought to be opportunistic in their feeding habits with fish and carrion as occasional food sources.

The ringed map turtle's habitat is typically riverine with a moderate current and numerous basking structures. This species has also been observed in oxbow lakes that are connected or disconnected from the main river system.

Nesting habitat consists of large, high sand bars adjacent to the river. Sandbars range in size from 430 square feet (40 square meters) to over 2.2 acres (8,900 square meters) and are generally composed of 39 percent open sand, 38 percent herbaceous vegetation, and 23 percent woody vegetation (Jones 2006). Nesting is initiated in May and ends in August with multiple (2 to 3) clutches per year being common.

Average longevity estimates were 13.9 for females and 8.5 years for males. Males mature at about 4.6 years of age while females mature about 9.1 years of age (Jones 2017).

Further details on life history can be found in Annex D1.

2.2.1.4 Conservation needs

There are currently no conservation plans for the ringed map turtle. However, there is a Recovery Plan dated 1988 that includes an outline for recovery actions addressing threats to the ringed map turtle. Below are the main objectives. See Annex D3 for further details.

- Protection of a total of 150 miles of the turtle's habitat in two reaches of the Pearl River. There must be a minimum of 30 miles in either reach with the total protected area totaling 150 river miles.

- Evidence of a stable or increasing population over at least a ten-year period in these two Pearl River reaches.
- An established, continuing plan of periodic monitoring of population trends and habitat to ensure a stable population in these river reaches.

2.2.2 Environmental baseline

2.2.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

Populations are known to occur within the Pearl River system from the Neshoba County, Mississippi headwaters area, southward downstream through St. Tammany Parish, Louisiana. The ringed map turtle populations are restricted primarily to the main channel of the Pearl River and the lower portions of its largest tributary, the Bogue Chitto River. To date, the highest densities of turtles have been documented in two survey areas, above the Ross Barnett Reservoir and below the Ross Barnett Reservoir dam southward to approximately MS Highway 25, upstream of the proposed Project Area.
ringed map turtles are found throughout all reaches of the Pearl River within the Action Area, with lower numbers in the channelized sections of the River (just south of RM 293 to approximately RM 287).

Approximately 40 percent of the proposed excavation area has little or no riparian habitat and little to no natural basking and feeding habitat, especially within the channelized portion. Selman (2018) found a greater concentration of turtles within forested riparian sites along this portion of the river. He also documented nest sites, turtle nesting crawls, and juvenile turtles all indicative of successful recruitment occurring in all stretches of the Action Area, including the area with reduced riparian habitat. It is estimated that a total of approximately 5,108 turtles occur in the Action Area.

2.2.2.2 Species conservation needs within the action area

There are currently no conservation plans for the ringed map turtle. However, there is a Recovery Plan dated 1988 that includes an outline for recovery actions addressing threats to the ringed map turtle. Below are the objectives that might be applicable to the action area. Annex D3 contains further details.

- Estimate number of ringed map turtles per mile in each of the study reaches.
- Determine seasonal and daily activity.
- Determine if the species moves any distance during its lifetime and barriers to such movement, if any.
- Protect two river reaches from activities that would cause a decline of this species' population.

- Develop and implement a monitoring plan to evaluate effectiveness of protective measures and to track population trends.

2.2.2.3 Habitat condition (general)

Alternative C and CTO

Habitat for the ringed map turtle is typically riverine with a moderate current and numerous basking logs. Populations are typically most abundant in areas of the river that have moderate to fast currents with deep water and sand and gravel bottoms. It is also important that the riverine habitat include numerous basking logs located in direct sunlight and with large sparsely vegetated sandbars that provide nesting habitat. The river channel itself must be wide enough to allow sunlight to penetrate for several hours a day. Nesting habitat for the turtles appears to be strictly limited to large, high sand and gravel bars located adjacent to the river channel.

The project area contains habitat that has been previously manipulated by the construction of levee's, channelization/straightening of the river, and elimination of a riparian buffer in places. Moreover, there is little natural basking habitat inside the project area.

2.2.2.4 Influences

Decline in populations of the ringed map turtle in certain areas of the Pearl River system have been attributed to habitat modifications, primarily associated with dredging and/or other navigational and flood control projects. Water quality degradation also seems to play an important role in population declines, over time, as has over-collecting of the species for the pet trade. In addition, recreational and other similar activities on the river may also cause habitat destruction, over time, especially as it relates to available nesting habitat on sandbars and/or the nesting activity itself. Predation of nests by raccoons, armadillos, and fish crows is also a threat to populations. The impact of human disturbance, primarily recreating (e.g., camping, picnicking, boating) to nesting turtles and/or nests has been pointed to as another source of decline in the population (Jones 2006; Jones 2017; Selman and Jones 2017). Direct mortality associated with recreational and commercial fishing and recreational boating has been identified as another impact to Graptemys populations (Bluté et al. 2010; Selman et al. 2013; Smith et al. 2018). Jones (2017) expressed a concern about those same activities impacting the ringed map turtle.

2.2.2.5 Additional baseline information

There is no additional baseline information.

2.2.3 Effects of the action

2.2.3.1 Indirect interactions

Alternative C

The establishment of an approximate 1,700-acre impoundment from weir construction will result in changes in the velocity and water surface elevation within the project area. Because the weir has been designed to match the current discharge of the river there should not be a significant change in discharge once flows begin overtopping the weir. The current lotic habitat will be replaced with a lentic habitat which has been proven by the Ross Barnett Reservoir to not support the persistence of the ringed map turtle.

The riparian zone will be almost eliminated, and development is planned for most of the areas of fill surrounding the improved channel. This would eliminate available habitat and increase disturbance. There is potential for existing nests to be flooded during filling of the pool area behind the weir if this occurs from May to October. Details of how the filling will be undertaken have not been finalized but would be coordinated with the Service.

Free-flowing river reaches typically support a higher quality macroinvertebrate community while pool communities typically consist of relatively few taxa dominated by oligochaetes and chironomid larvae that are more tolerant of poorer water quality. Until recolonization of macroinvertebrates the competition for food resources within the channelized area would impact all ringed map turtles within the impoundment.

Turtles downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation on sandbars and basking material during construction. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the ringed map turtle.

Fluctuations and stratifications in the water quality (e.g., DO) like what occurs in the Ross Barnett Reservoir (larger but similar in depth) could be expected. This could result in poorer and/or reduced food sources because of decreased water quality and the potential influence of contaminants. Modeling of the project area indicates that water quality should not significantly decline, and ringed map turtles are currently persisting in the area with the ongoing discharges. Therefore, it is assumed that while some water quality changes may occur, they would not have an adverse effect.

The fish -passage channel would provide approximately 1 mile (0.2 percent of the species range) of flowing water during low flow periods when the channelized area would experience low velocities. Depending on the width and velocities of this feature it could provide additional habitat for the ringed map turtle and would prevent isolation of the populations up and down stream of the weir.

It is anticipated that downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result primarily from an increase in turbidity decreasing potential food sources.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.2.3.2 Direct interactions

Alternative C

Disturbance from excavating 25 million cubic yards of material from approximately 1,901 acres within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area will slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking, and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.2.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.2.5 Discussion and conclusion

Alternative C

Previous consultation resulted in the following to offset or reduce direct losses of turtles due to construction. However, USACE is currently coordinating with USFWS to determine if these measures are still applicable. Up to 2,018 eggs would be relocated outside the construction area and protected from predators and 20 individuals (0.03 percent of the total population) would be relocated from Cypress Lake to the Pearl River. In addition, up to 1,600 individuals (1 percent of the total population) would be trapped, tagged, data collected, tracked, observed, and monitored in the Action Area population.

Additional offsets to turtle losses that could be implemented as part of the Action include: (1) the creation and protection of 31.4 acres of nesting habitat (estimated to produce at least 1,176 nests) and adjacent basking habitat and predator control; (2) the establishment and enforcement of no-wake zones to reduce boat strikes and disturbance during basking; (3) the placement of public access conditions to reduce disturbances to basking and nesting behaviors and habitats (4) the creation of an approximately 1 mile fish by-pass, and (5) the protection of 10 miles of riverbank that would prevent the development and destruction of riparian habitat utilized by the turtle and also reduce nesting and basking disturbances.

Based upon literature review and available survey data, and the effects of the action (both detrimental and beneficial activities proposed), the USACE has determined that implementation of Alternative is likely to adversely affect but is not likely to jeopardize the continued existence of the ringed map turtle.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.3 Northern long-eared bat (NLEB) (*Myotis septentrionalis*)

2.3.1 Status of the species

2.3.1.1 Legal status

The NLEB is listed as an endangered species under the Act (87 FR 73488 November 30, 2022)

2.3.1.2 Recovery plans

There are currently no recovery plans for the NLEB. However, there is a SSA dated August 2022 (Annex D3).

2.3.1.3 Life history information

NLEB, a wide-ranging bat species, found in 37 states and 8 provinces in North America, typically overwinters in caves or mines and spends the remainder of the year in forested habitats. The NLEB individuals are typically approximately 3.0 to 3.7 inches in length with a wingspan of approximately 9.0 to 10.0 inches. The bat is distinguished by its long ears, particularly when compared to the other bats in the same genus, Myotis. The primary diet for the NLEB is insects including moths, flies, leafhoppers, caddisflies, and beetles.

2.3.1.4 Conservation needs

The SSA dated August 2022 includes conservation efforts for the NLEB. Below are the conservation efforts listed in the SSA. See Annex D3 for further details.

- NLEB receives varying degrees of protection through state laws as it is designated as Endangered in Arkansas, Connecticut, Delaware, Indiana, Maine, Massachusetts, Missouri, New Hampshire, Vermont; Threatened in Georgia, Illinois, Louisiana, Maryland, New York, Ohio, Pennsylvania, Tennessee, Virginia, and Wisconsin; and Special Concern in Alabama, Iowa, Michigan, Minnesota, Mississippi, Oklahoma, South Carolina, South Dakota, West Virginia, and Wyoming.
- Multiple national and international efforts are underway in attempt to reduce the impacts of white nose syndrome by determining the cause of the disease and reducing or slowing its spread.
- Operational strategies at wind power facilities.
- Forestry programs/forest management
- Bat-friendly gates to protect important hibernation sites.

2.3.2 Environmental baseline

2.3.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

Although the USFWS ECOS webpage does not include the counties of Hinds and Rankin as part of the NLEB range, the Service has identified what is referred to as the White-Nose Syndrome Buffer Zone that includes all areas within 150 miles of the boundaries of U.S. counties or Canadian districts where the fungus has previously been detected. The established buffer zone includes both Hinds and Rankin Counties within the Project Area.

At this point, the Service does not have survey data that would indicate what the migration patterns are for the NLEB. More specifically, little is known whether the available summertime woodland habitat present within the Project Area is being utilized by the NLEB. No existing data is available that would indicate that the NLEB currently utilizes the Project Area during the summer migration.

2.3.2.2 Species conservation needs within the action area

The SSA dated August 2022 includes conservation efforts for the NLEB. Below are the conservation efforts listed in the SSA that might be applicable to the action area. See Annex D3 for further details.

- NLEB receives protection through Mississippi state law as it is designated as Endangered in Mississippi.

2.3.2.3 Habitat condition (general)

<u>Alternative C and CTO Alternative</u>

NLEBs typically roost singly or in maternity colonies underneath bark or more often in cavities or crevices of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 662; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 222; Timpone et al. 2010, p. 119). Males' and non-reproductive females' summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, p. 77; Amelon and Burhans 2006, p. 72). NLEBs are flexible in tree species selection and while they may select for certain tree species regionally, likely are not dependent on certain species of trees for roosts throughout their range; rather, many tree species that form suitable cavities or retain bark will be used by the bats opportunistically (Foster and Kurta 1999, p. 668; Silvis et al. 2016, p. 12; Hyzy 2020, p. 62).

NLEBs are thought to predominantly overwinter in hibernacula that include caves and abandoned mines. NLEBs are typically found roosting singly or in small numbers in cave or mine walls or ceilings, often in small crevices or cracks.

2.3.2.4 Influences

The Northern Long-eared Bat (NLEB) is one of the species of bats that have been most impacted by the spread of the white- nose syndrome disease and has experienced significant declines in populations because of the spread of the disease. Secondary threats to the NLEB include the disturbance of roosts and hibernation areas, forest management practices, and forest habitat modifications (development, wind power development).

2.3.2.5 Additional baseline information

There is no additional baseline information.

2.3.3 Effects of the action

2.3.3.1 Indirect interactions

<u>Alternative C</u>

The proposed project would remove potential roosting and foraging habitat (forests and structures such as abandoned bridges) and could result in potential adverse effects.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative could have similar impacts compared to Alternative C if clearing of existing forests is implemented, but fewer impacts than Alternative C if clearing is not included or is to a lesser degree.

2.3.3.2 Direct interactions

Alternative C and CTO Alternative

No direct interactions are anticipated as no existing data is available that would indicate that the NLEB currently utilizes the project area.

2.3.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.3.5 Discussion and conclusion

Alternative C

The proposed project will not occur near or affect any known maternity roost trees but will remove potential roosting and foraging habitat and could result in potential adverse effects. Under the final 4(d) rule, any incidental take resulting from forest conversion as a part of the channel excavation and levee realignment action of this project would be considered incidental take resulting from otherwise lawful activities and is not prohibited under the Endangered Species Act. Accordingly, the USACE has determined that the action may affect, but is not likely to adversely affect the NLEB.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.4 Pearl River Map Turtle (PRMT) *(Graptemys pearlensis)*

2.4.1 Status of the species

2.4.1.1 Legal status

The current listing of the Pearl River map turtle is “Proposed Threatened” (Federal Register Vol. 86, No. 223, November 23, 2021, p66624)

2.4.1.2 Recovery plans

There are currently no recovery plans for the PRMT. However, there is a Species Status Assessment Report (SSA) dated April 2021 (Annex D3).

2.4.1.3 Life history information

The PRMT is endemic to the Pearl River drainage in Mississippi and Louisiana. Rankin and Hinds Counties are included in the Counties with known records for the species in the state of Mississippi. The occupied range of the PRMT includes portions of the Pearl River, West Pearl River, Bogue Chitto, East Pearl River, Yockanookany River, Strong River, Holmes Bayou, Pearl Navigation Canal, Lobutcha Creek, Tuscolometa Creek, Pelahatchie Creek, Purvis Creek, Pushepatapa Creek, Topisaw Creek, Magees Creek, Hobolochitto Creek, and West Hobolochitto Creek. This species has also been reported in upper reaches of the Ross Barnett Reservoir.

The PRMT is a moderate-sized highly aquatic turtle found in the Pearl River drainage area of Louisiana and Mississippi. The PRMT exhibits sexual dimorphism with males and females differing in size. Female PRMTs have an average carapace length of 295 mm, with the male PRMT having an average carapace length of 121 mm. The PRMT exhibits a high-domed shell with a median keel, featuring salient spines on the rear portions of the anterior vertebral scutes; although similar visually, the spines are considerably smaller to that of the ringed map turtle’s. A key distinguishing feature of the PRMT is the complete dark stripe along the median keel. The background color of the carapace is olive green, with vermiculation and yellow pigmentation present. The plastron is generally flat and pale yellow with dark pigmentations along the seams.

The PRMT is a wholly carnivorous species, with insects and mollusks constituting their principal diet. In addition, they are also thought to be opportunistic in their feeding habits with fish and carrion as occasional food sources. A recent study found that mature females consume mostly Asian clams (*Corbicula fluminea*), while males and unsexed juveniles eat insects, with mature males specializing in caddisfly larvae and consuming more mollusks than juveniles (Vučenović and Lindeman 2020, entire). In fecal samples from a site on the Pearl River, the diet for both sexes of all sizes combined was composed of 44 percent fish, 25 percent mollusks, and 25 percent insects (McCoy and Vogt, unpubl. data reported in Lovich et al. 2009, p. 029.4).

A study on the ringed map turtle (Graptemys oculifera), found that PRMTs were more frequently seen basking later in the afternoon than ringed map turtles, and suggested that more PRMTs might have been detected if more surveys were conducted after 3 pm (Dickerson and Reine 1996, p.8).

PRMTs excavate nests and lay their eggs on sandbars and beaches along riverbanks during the late spring and early summer months. The time from deposition to nest emergence by hatchlings in natural clutches average 69.3 days. An average clutch size of 6.4 eggs with a range of 4-9 eggs was reported for the PRMT and stated that females probably produce multiple clutches per year (Ennen et al. 2016, pp. 094.4-094.6).

Humans, alligators, alligator snapping turtles and otters are predators of adult PRMTs, with eggs and hatchlings more susceptible to small mammals, snakes, and crows. Red imported fire ants have also been documented invading turtle nests in the southeastern United States and can cause nest failure and hatchling mortality (Buhlmann and Coffman 2001, entire).

2.4.1.4 Conservation needs

The SSA dated April 2021 includes conservation measures for the PRMT. Below are the federal conservation measures listed in the SSA. See Annex D3 for further details.

- The same recovery actions that are listed for the ringed sawback turtle could benefit the PRMT (see section 2.2.1.4).
- The Clean Water Act of 1972 which encourages avoidance, minimizing and requires mitigation for unavoidable impacts to the aquatic environment and habitats. This includes protecting the riverine habitat occupied by the PRMT.
- The Endangered Species Act (Act) could offer some protection as the PRMT likely receives ancillary protection where it co- occurs with other species listed under the ESA.
- A Comprehensive Conservation Plan (CCP) has been developed under The National Wildlife Refuge System Administration Act (NWRAA) to provide the framework of fish and wildlife management on the Bogue Chitto National Wildlife Refuge (U.S. Fish and Wildlife Service 2011, entire). Within the CCP, specific actions are described to protect the ringed map turtle that will also benefit the PRMT which occurs on the Refuge.
- The Sikes Act Improvement Act (1997) led to Department of Defense guidance regarding development of Integrated Natural Resources Management Plans (INRMP) for promoting environmental conservation on military installations. There are records of the PRMT from Stennis WMA

(Buhlman 2014, pp. 11-12, 31-32). The U.S. Navy has developed an INRMP for the Stennis WMA (U.S. Navy 2011, entire).

2.4.2 Environmental baseline

2.4.2.1 Species presence and use

Alternative C and CTO Alternative

The project area is in Rankin and Hinds Counties, Mississippi which are included in the Counties with known records for the species in the state of Mississippi. This species has also been reported in upper reaches of the Ross Barnett Reservoir.

PRMTs can be found within the project area despite the lack of a well-defined riparian buffer, lack of preferred habitat, sedimentation accumulation, relatively low stream velocities, lack of basking habitat, and a smaller percentage of sandbars. It has been shown in studies that population densities for the species are higher above and below the project area.

2.4.2.2 Species conservation needs within the action area

The Species Status Assessment Report (SSA) dated April 2021 includes conservation measures for the PRMT. Below are the state conservation measures that might be applicable to the action area. See Annex D3 for further details.

- The same recovery actions that are listed for the ringed sawback turtle could benefit the PRMT (see section 2.2.2.2).
- Protections under state law are limited to licensing restrictions for take for personal use of nongame species in need of management (which includes native species of turtles). A Mississippi resident is required to obtain one of three licenses for capture and possession of PRMTs.
- The Mississippi Comprehensive Wildlife Action Plan (MMNS 2015, entire) includes recovery of species designated as Species of Greatest Conservation Need (SGCN) which includes the PRMT.

2.4.2.3 Habitat condition (general)

Alternative C and CTO

PRMTs occur in sand and gravel-bottomed rivers and creeks with dense accumulations of deadwood; they have not been documented in oxbow lakes or other floodplain habitats. They were notably absent from lakes where the ringed map turtle is present but do occur at the upstream reach of Ross Barnett Reservoir (Lindeman 2013, p. 298). Emergent deadwood serves as thermoregulatory basking structure, foraging structure for males and juveniles (Selman and Lindeman 2015, pp. 794-795), and as an overnight resting place for males and juveniles (Cagle 1952, p. 227). PRMT density was greater on mainstem reaches and large tributaries than on small tributaries (Lindeman 2019, pp. 13-18).

2.4.2.4 Influences

Climate change, water quality, habitat degradation, invasive species, collection, and disease all influence the persistence of the species.

Variability in climate may affect ecosystem processes and communities resulting in potential effects on community composition and individual species interactions (DeWan, et al., 2010, p. 7). These changes have the potential to impact PRMTs and/or their habitat.

The dual stressors of climate change and direct human impact have the potential to impact aquatic ecosystems by altering stream flows and nutrient cycles, eliminating habitats, and changing community structure (Moore et al. 1997, pp. 942).

Degradation of stream and wetland systems through reduced water quality and increased concentrations of contaminants can affect the occurrence and abundance of freshwater turtles (DeCatanzaro and Chow-Fraser 2010, p. 360).

Dredging and channelization modify and destroy habitat for aquatic species by destabilizing the substrate, increasing erosion and siltation, removing woody debris, decreasing habitat heterogeneity, and stirring up contaminants which settle onto the substrate (Williams et al. 1993, pp. 7-8; Buckner et al. 2002, entire; Bennett et al. 2008, pp. 467-468). Considerably low densities of PRMTs were observed in the lower reaches of the Pearl, where much channelization and flow diversion has occurred (Lindeman 2019, pp. 23-29).

Impoundment of rivers is a primary threat to aquatic species in the southeast (Folkerts 1997, p. 11; Buckner et al. 2002, entire). Dams modify habitat conditions and aquatic communities both upstream and downstream of an impoundment (Winston et al. 1991, pp. 103-104; Mulholland and Lenat 1992, pp. 193-231; Soballe et al. 1992, pp. 421-474). Dams fragment habitat for aquatic species by blocking corridors for migration and dispersal, resulting in population geographic and genetic isolation and heightened susceptibility to extinction (Neves et al. 1997, unpaginated).

The degree to which invasive species effect the PRMT has not been studied, but the diet of mature females may have been broader before the introduction of Asian Clams (Corbicula fluminea) and removal of invasive vegetation on sandbars has been suggested as nesting habitat management (Selman and Lindeman 2015, p. 794-795; Lindeman 2019, p. 33).

Exploitation of PRMTs for the pet trade domestically and in Asian markets has been documented, but the degree of impact is unclear, as it is unknown whether captive individuals were Pascagoula ringed map turtles orPRMTs (Lindeman 1998, p. 137; Cheung and Dudgeon 2006, p. 756; USFWS 2006, p. 2; Selman and Qualls 2007, p. 32-34; Ennen et al. 2016, p. 094.6).

Ranaviruses are capable of infecting turtles. Aquatic turtles share habitat with susceptible fish and amphibian populations and as a result may be more at risk of infection than terrestrial turtles (Wirth et al. 2018, p. 6).

2.4.2.5 Additional baseline information

There is no additional baseline information.

2.4.3 Effects of the action

2.4.3.1 Indirect interactions

<u>Alternative C</u>

The establishment of a 1,700-acre impoundment from weir construction will result in changes in the velocity and water surface elevation within the project area. Because the weir has been designed to match the current discharge of the river there should not be a significant change in discharge once flows begin overtopping the weir. The current lotic habitat will be replaced with a lentic habitat which would not support the persistence of the PRMT.

The riparian zone will be almost eliminated, and development is planned for most of the areas of fill surrounding the improved channel. This would eliminate available habitat and increase disturbance. There is potential for existing nests to be flooded during filling of the pool area behind the weir if this occurs late spring to early summer months. Details of how the filling will be undertaken have not been finalized but would be coordinated with the Service.

Free-flowing river reaches typically support a higher quality macroinvertebrate community while pool communities typically consist of relatively few taxa dominated by oligochaetes and chironomid larvae that are more tolerant of poorer water quality. Until recolonization of macroinvertebrates is the competition for food resources within the channelized area would impact any Pear River ringed map turtles within the impoundment. It is expected that there would be more of a decline of PR ringed map turtles than there would be of ringed sawback turtles as PR ringed map turtles are riverine obligates while ringed sawback turtles tend to be more generalists.

Turtles downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation on sandbars and basking material during construction. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the PRMT.

Fluctuations and stratifications in the water quality (e.g., DO) like what occurs in the Ross Barnett Reservoir (larger but similar in depth) could be expected. This could result in poorer and/or reduced food sources because of decreased water quality and the potential influence of contaminants. Modeling of the project area indicates that water quality should not significantly decline, and PR ringed map turtles are currently persisting in the area with the ongoing discharges. Therefore, it is assumed that while some water quality changes may occur, they would not have an adverse effect.

The fish -passage channel would provide approximately 1 mile (0.2 percent of the species range) of flowing water during low flow periods when the channelized area would experience low velocities. Depending on the width and velocities of this feature it could provide additional habitat for the PR ringed map turtle and would prevent isolation of the populations up and down stream of the weir.
It is anticipated that approximately 1.6 miles downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result primarily from an increase in turbidity decreasing potential food sources.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.4.3.2 Direct interactions

Alternative C

Disturbance from excavating 25 million cubic yards of material from approximately 1,901 acres within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area will slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking, and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.4.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.4.5 Discussion and conclusion

Alternative C

Based upon literature review and available survey data, and the effects of the action (both detrimental and beneficial activities proposed), the USACE has determined that implementation of Alternative C is likely to adversely affect but is not likely to jeopardize the continued existence of the PRMT.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.5 Alligator Snapping Turtle (AST) *(Macrochelys temminckii)*

2.5.1 Status of the species

2.5.1.1 Legal status

The current listing of the AST is “Proposed Threatened” (Federal Register Vol. 86, No. 214/Tuesday, November 9, 2021)

2.5.1.2 Recovery plans

There are currently no recovery plans for the alligator snapping turtle. However, there is a SSA dated March 2021 (Annex D3).

2.5.1.3 Life history information

The AST is the largest freshwater species of turtle in North America and is among the most aquatic. ASTs are characterized as having a large head, long tail, and an upper jaw with a hooked beak. They have three keels with posterior elevations on the scutes of the carapace, which is dark brown and often found with algae growth adding to the overall camouflage of the turtle. The plastron is greyish-brown in adults, and somewhat mottled with small whitish blotches in juveniles. The eyes are positioned on the side of the head, surrounded by small, pointed projections.

The AST is found within river systems that flow into the Gulf of Mexico, extending from just before the Suwannee River in Florida to the San Antonio River in Texas. Currently, the species is known to occur in Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas.

ASTs are usually associated with the deeper waters of large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows. Hatchlings and juveniles, in comparison, are usually associated with shallower waters. In general, the species uses shallower water in early summer and deeper depths in late summer and mid-winter, which may be a thermoregulatory shift (Fitzgerald and Nelson 2011). The presence of barnacles on some specimens may also indicate an ability to spend prolonged periods in brackish water (Jackson and Ross 1971, p.188-189).

AST males reach sexual maturity in 11-21 years and 13-21 years for females. Females have been observed to have no more than a single clutch per year in the wild, as well as not appearing to be particularly selective on nesting sites. Nesting sites have been observed across a range of distances from 8 to 656 ft from the nearest water source. ASTs exhibit temperature dependent sex determination within nest incubation temperatures. Nesting occurs between May to July with areas in the most southern ranges beginning in April and extending through May. ASTs exhibit sexual dimorphism with males being distinctively larger than females, and also displaying a larger anterior to vent tail length.

ASTs are opportunistic scavengers and consume a variety of foods. Although fish comprise the majority of their diet, crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetations have also been reported (Elsey, 2006). The AST is the only turtle that uses a modified tongue appendage as a predatory lure to attract fish into range.

Racoons, armadillos, opossums, and otters are all known to prey on AST nests. Predators of hatchlings are likely to include large fish, wading birds, otters, and alligators (Ernst and Lovich 2009, p. 149). Red imported fire ants (Solenopsis invicta) are also known to cause significant decline in hatching success.

2.5.1.4 Conservation needs

The SSA dated March 2021 includes conservation measures for the AST. Below are the conservation measures listed in the SSA. See Annex D3 for further details on each.

- Captive Rearing, Head-Starting, and Reintroductions
- Integrated Natural Resource Management Plans
- Predator exclusion structures

2.5.2 Environmental baseline

2.5.2.1 Species presence and use

Alternative C and CTO Alternative

ASTs were historically found in 14 states: Alabama, Arkansas, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. Currently, the species is known to occur in Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. ASTs are found in deeper waters and their major tributaries, however their habitats have been known to extend into small streams, bayous, canals, swamps, lakes, reservoirs, ponds, and oxbows. The AST is usually associated with structure more so than open water. Riparian canopy cover is an important feature for the AST, as they typically select sites with a high percentage of coverage (Howey and Dinkelaker 2009).

The Service divides the AST range into seven (7) analysis units. The analysis unit focused on in relation to the project area is the Alabama unit which encompasses eastern Mississippi, western Alabama, and small parts of Louisiana and Florida. The Pearl River is listed under the Alabama unit as a water body that currently or historically supported ASTs.

The Alabama Analysis unit has an estimated abundance of 200,000 (55.37%). It is estimated range wide that there is between 68,154 and 1,436,825 individuals with 55 percent of the turtles occurring in the Alabama analysis unit (USFWS. “Federal Register / Vol. 86, No. 214 / Tuesday, November 9, 2021).

2.5.2.2 Species conservation needs within the action area

The SSA dated March 2021 includes conservation measures for the AST. However, there are no conservation needs specific to the action area.

2.5.2.3 Habitat condition (general)

ASTs are associated with deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows); with shallower water occupied in early summer and deeper depths in late summer and mid-winter, which represent a thermoregulatory shift (Ernst and Lovich 2009, p. 141). In comparison, hatchlings and juveniles tend to occupy shallower water. ASTs are also associated with structure (e.g., tree root masses, stumps, submerged trees, etc.); and may occupy areas with a high percentage of canopy cover undercut stream banks.

2.5.2.4 Influences

Adult harvest (legal and illegal), bycatch, habitat alteration, nest predation, climate change, and disease influence the existence of the AST.

Although regulatory harvest restrictions have decreased the amount of ASTs being harvested, populations have not necessarily increased in response. This lag in population response is likely due to the demography of the species, specifically delayed maturity, long generation times, and relatively low reproductive output.

ASTs can be killed or harmed incidental to other fishing and recreational activities. Threats include capture as bycatch associated with commercial harvest of other species, ingestion of fishhooks and/or drowning when captured on trotlines (a fishing line strung across a stream with multiple hooks set at intervals) and limb lines (single hooks hung from branches), drowning from entanglement in various types of fishing line, and boat propeller strikes.

Dams change the hydrology of streams and could impede dispersal and genetic interchange for this highly aquatic species, but impoundments can also provide habitat for the species (Pritchard 1989, p. 84). Other activities and processes that can alter habitat include dredging, deadhead logging, removal of riparian cover, channelization, stream bank erosion, siltation, and land use adjacent to rivers (e.g., clearing land for agriculture).

Nest predation rates for the AST are high. Small mammals and red fire ants are known to prey on the nests. In 2008, one of five AST nests investigated in Louisiana was infested by the phorid fly Megaselia scalaris (snapping turtles; Holcomb and Carr 2011b, entire).

Climate change might impact the AST in several ways, including loss of habitat to sea level rise for those populations near coastal areas, impacts of drought on habitat and water availability, and physiological impacts on sex determination. Climate conditions also appear to limit the distribution of ASTs.

Chaffin et al. (2008, entire) captured and assessed the health of 97 free-ranging ASTs across nine sites in northwestern Florida and southwestern Georgia between 2001 and 2006. Assessed ASTs had shell abnormalities, including worn, cracked, or broken scutes, fresh or healed wounds resulting from trauma, missing portions of the tail, missing portions of the beak, missing portions of claws, and leech infestation (Chaffin et al. 2008, p. 674). Protozoan parasites transmitted by leeches, were found in all but one turtle assessed. Herpes was the only pathogen detected, but none of the individuals were showing symptoms. Mercury was also detected in the blood in 93% of samples.

2.5.2.5 Additional baseline information

There is not additional baseline information.

2.5.3 Effects of the action

2.5.3.1 Indirect interactions

<u>Alternative C</u>

Indirect impacts associated with the project would include the potential for degradation of water quality, loss of woody debris, nesting habitat loss due to flooding, nest predation issues and increases in bycatch due to recreation increase. There are also concerns about the potential impacts of the project on other species that rely on the same habitat as the AST. For example, the project and associated infrastructure could temporarily impact local fish populations, which in turn may impact the local turtle population as these fish populations are a primary food source for the AST.

Potential benefits of the project for the AST include the creation of a new, more suitable, and desirable habitat when compared to existing conditions. The construction of the project and associated infrastructure could provide new areas of deep, permanent water with a soft substrate for nesting.

<u>CTO Alternative</u>

While the specific features of the CTO alternative have not been determined, this alternative could have less impact in the long-term compared to Alternative C without a weir, however, it will have less benefits than if a weir was constructed. If Alternative CTO includes a new weir the impacts and benefits would be similar to Alternative C.

2.5.3.2 Direct interactions

<u>Alternative C</u>

Disturbance from excavating 25 million cubic yards of material from approximately 1,901 acres within and adjacent to the river over approximately two years could result in death of individuals if they are unable to escape the construction work area. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area will slowly move away from construction activities. Turtles in the construction area are expected to be disturbed in some form of alteration of normal feeding, basking and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

The construction of the flood control project and associated infrastructure could temporarily alter habitat conditions, leading to a decline in the AST population. In addition, the project could also potentially impact the AST through temporary changes in water quality. Impacts include removal of natural buffers that would impact water quality, and a slight decrease and less variation of dissolved oxygen concentrations.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative could have less impact in the long-term compared to Alternative C without a weir, however, it will have less benefits than if a weir was constructed. If Alternative CTO includes a new weir the impacts and benefits would be similar to Alternative C.

2.5.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.5.5 Discussion and conclusion

Alternative C

Overall, while the project raises concerns about the potential adverse impacts on the AST and environment, the potential benefits should also be considered. It is possible that the project will create an overall more desirable habitat for the species when compared to current habitat options within the project area. The project in general will provide more permanent deep-water habitat, potentially increase water quality, and increase the available soft substrate for nesting.

Based upon literature review and available survey data, and the effects of the action (both detrimental and beneficial activities proposed), the USACE has determined that implementation of Alternative C is likely to adversely affect but is not likely to jeopardize the continued existence of the AST.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.6 Tricolored Bat (TCB) (*Perimyotis subflavus*)

2.6.1 Status of the species

2.6.1.1 Legal status

The current listing of the Tricolored bat is “Proposed Endangered” (88 FR 16776, March 20, 2023, p16776-16832)

2.6.1.2 Recovery plans

There are currently no recovery plans for the tricolored bat. However, there is a SSA dated December 2021 (Annex D3).

2.6.1.3 Life history information

TCB is one of the smallest bats in eastern North America and is distinguished by its unique tricolored fur that appears dark at the base, lighter in the middle, and dark at the tip (Barbour and Davis 1969, p. 115). TCB primarily roost in foliage of live and dead trees in the spring, summer, and fall, and hibernate in caves and other subterranean habitats during the winter. TCB are opportunistic feeders feeding on small insects such as moths, beetles, flies, wasps, and flying ants.

TCB are known to occur in 39 states, one of which is Mississippi, Washington D.C., 4 Canadian Provinces, Guatemala, Honduras, Belize, Nicaragua, and Mexico.

2.6.1.4 Conservation needs

The SSA dated December 2021 includes conservation efforts for the TCB. Below are the conservation efforts listed in the SSA. See Annex D3 for further details.

- TCB could receive varying degrees of protection through state and federal laws once the listing decision is made.
- Multiple national and international efforts are underway in attempt to reduce the impacts of white nose syndrome by determining the cause of the disease and reducing or slowing its spread.
- Operational strategies at wind power facilities.
- Forestry programs/forest management
- Bat-friendly gates to protect important hibernation sites.

2.6.2 Environmental baseline

2.6.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

The tricolored bat is widespread throughout MS and they can be found in many different habitat types throughout the year. The presence in the project area is not known at this time. However, it is safe to assume that the TCB may use the area for roosting and potentially wintering.

2.6.2.2 Species conservation needs within the action area

The SSA dated December 2021 includes conservation efforts for the TCB. However, conservation efforts within the action area have not yet been determined.

2.6.2.3 Habitat condition (general)

Alternative C and CTO Alternative

TCB seem to be opportunistic roosters and roost in live and dead leaf clusters of deciduous hardwood trees, Spanish moss, pine needles, eastern red cedar, barns, beneath porch roofs, bridges, concrete bunkers, and rarely within caves.

TCB have been documented overwintering in caves, mines, rock crevices, talus, tunnels, bunkers, basements, bridges, aqueducts, trees, earthen burrows, leaf litter, and a variety of other roosts. For bats to hibernate successfully, the most important conditions are relatively stable- low temperatures, but generally above freezing, and high humidity.

2.6.2.4 Influences

The TCB has been impacted by the spread of the WNS disease and has experienced significant declines in populations because of the spread of the disease. Other threats to the TCB include wind related mortality due to wind power development, climate change, and habitat loss.

2.6.2.5 Additional baseline information

There is no additional baseline information.

2.6.3 Effects of the action

2.6.3.1 Indirect interactions

Alternative C

Indirect impacts would be due to the removal of potential roosting and foraging habitat (forests and structures such as abandoned bridges) and could result in potential adverse effects.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative could have similar impacts compared to Alternative C if clearing of existing forests is implemented, but fewer impacts than Alternative C if clearing is not included or is to a lesser degree.

2.6.3.2 Direct interactions

Alternative C and CTO Alternative

Since the TCB is widespread throughout MS, there are no existing survey data for the project area, and they can be found in many different habitat types throughout the year, it is difficult to determine the direct impacts to the species at this time.

2.6.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.6.5 Discussion and conclusion

Alternative C

USACE has conducted literature reviews and is in coordination with the Service. Due to the lack of available survey data, the effects of the action on the TCB have not been finalized. Therefore, USACE has not yet made a determination for the TCB.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.7 Louisiana Pigtoe (LA pigtoe) (*Pleurobema riddellii*)

2.7.1 Status of the species

2.7.1.1 Legal status

The current listing of the Louisiana pigtoe mussel is “Proposed Threatened” (87 FR 56381, Sept 14, 2022, p56381-56393)

2.7.1.2 Recovery plans

There are currently no recovery plans for the Louisiana pigtoe. However, there is a SSA dated February 2022 (Annex D3).

2.7.1.3 Life history information

The LA pigtoe is a medium-sized freshwater mussel (shell lengths to greater than 62 mm) with a brown to black, triangular to subquadrate shell without external sculpturing, sometimes with greenish rays. They occur in gravel and coarse sandy substrates of rivers and streams. Mussels are filter feeders that rely on natural, high quality (pollutant free) flowing water of sufficient volume to support their life cycle, and that of their host fishes, which are essential for reproduction.

The range of the LA pigtoe extends into portions of east Oklahoma, southeast Arkansas, south Louisiana, and west Mississippi. Louisiana PA pigtoe currently occupies areas across seven major river basins (San Jacinto, Neches, Sabine, Big Cypress-Sulphur, Red, Calcasieu-Mermentau, and Pearl). However, the LA pigtoe is only found in the Pearl River within the project area and a portion of the west Pearl.

Degraded water quality, altered hydrology, substrate changes, habitat fragmentation, direct mortality, invasive species, and climate change all influence the existence of the Louisiana LA pigtoe. The remaining populations are in low condition and are therefore particularly vulnerable to extirpation.

2.7.1.4 Conservation needs

The SSA dated February 2022 does not include conservation efforts for the LA pigtoe.

2.7.2 Environmental baseline

2.7.2.1 Species presence and use

<u>Alternative C and CTO Alternative</u>

The LA pigtoe is only found in the Pearl River within the project area and a portion of the west Pearl.

2.7.2.2 Species conservation needs within the action area

The SSA dated February 2022 does not include conservation efforts for the LA pigtoe.

2.7.2.3 Habitat condition (general)

According to the February 2022 SSA, LA pigtoe occur in medium to large streams and rivers, requiring 1) flowing water of sufficient quantity and quality 2) adequate food supply, 3) habitat that provides refugia from both high- and low-flow events, 4) appropriate substrate that is generally characterized as stable and free of excessive fine sediment, 5) access to appropriate fish hosts, and 6) habitat connectivity (i.e., lack of impoundments and other barriers to fish pass).

Louisiana Pigtoe occur in medium to large-sized streams and rivers in flowing waters (0.3-1.4 m/s) over substrates of cobble and rock or sand, gravel, cobble, and woody debris; they are often associated with riffle, run, and sometimes larger backwater tributary habitats (Ford et al. 2016, pp. 42, 52; Howells 2010a, p. 3-4; Williams et al. 2017b, p. 21).

2.7.2.4 Influences

Degraded water quality, altered hydrology, substrate changes, habitat fragmentation, direct mortality, invasive species, and climate change all influence the existence of the Louisiana LA pigtoe.

2.7.2.5 Additional baseline information

There is no additional baseline information.

2.7.3 Effects of the action

2.7.3.1 Indirect interactions

Alternative C

Indirect impacts due to changes in the velocity and water surface elevation are anticipated. The current lotic habitat would be replaced with a lentic habitat which would eliminate available habitat and host fishes. LA pigtoes downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation associated with construction activities which may cause extirpation. However, once sediment runoff issues have dissipated due to high streamflow events, it is expected that the habitat immediately downstream of the weir would remain suitable for the LA pigtoe. It is anticipated that downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result in a river bottom shift and would bury mussel beds which would then result in suffocation of individuals. The increase in turbidity and decreasing water quality would also impact potential host fishes. The population is in low condition and are therefore particularly vulnerable to extirpation.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.7.3.2 Direct interactions

Alternative C

Direct impacts by way of death are anticipated due to implementation of Alternative C. Excavation of material from within the river over approximately two years would result in death of individuals as well as displacement of host fishes.

CTO Alternative

While the specific features of the CTO alternative have not been determined, this alternative is expected to have substantially less impacts compared to Alternative C if a new weir is not constructed, but similar impacts to Alternative C if a new weir is constructed.

2.7.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.7.5 Discussion and conclusion

Alternative C

USACE is continuing to review literature and is in coordination with the Service to finalize the effects of the action on the LA pigtoe. However, a determination has not yet been made.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

2.8 Monarch Butterfly (*Danaus plexippus)*

2.8.1 Status of the species

2.8.1.1 Legal status

The monarch butterfly is currently a candidate species.

2.8.1.2 Recovery plans

There are currently no recovery plans for the monarch butterfly. However, there is a SSA dated September 2020 (Annex D3).

2.8.1.3 Life history information

Adult monarch butterflies are large (3 to 4 inches) and conspicuous, with bright orange wings surrounded by a black border and covered with black veins. The black border has a double row of white spots, present on the upper side of the wings. Milkweed and flowering plants are needed for monarch habitat. Adult monarchs feed on the nectar of many flowers during breeding and migration, but they can only lay eggs on milkweed plants.

Migratory individuals in eastern North America predominantly fly south or southwest to mountainous overwintering grounds in central Mexico, and migratory individuals in western North America generally fly shorter distances south and west to overwintering groves along the California coast into northern Baja California (Solensky 2004).

The eastern population of monarchs overwinter in Mexico, where this microclimate is provided by forests primarily composed of oyamel fir trees (Abies religiosa). Migratory monarchs in the western population primarily overwinter in groves along the coast of California and Baja California which include blue gum eucalyptus (Eucalyptus globulus), Monterey pine (Pinus radiata), and Monterey cypress (Hesperocyparis macrocarpa) (Griffiths and Villablanca 2015)

Monarch butterflies are found throughout North America and are highly likely to utilize portions of the project area.

2.8.1.4 Conservation needs

The Species Status Assessment Report, version 2.1 dated September 2020 discusses conservation efforts for the monarch butterfly. Below is a brief summary. See Annex D3 for further details.

- Protection, restoration, enhancement and creation of habitat is a central aspect of recent monarch conservation strategies.
- Improved management at overwintering sites in California has also been targeted to improve the status of western North American monarch butterflies (Pelton et al. 2019; WAFWA 2019).
- The Western Monarch Butterfly Conservation Plan which includes
- Protecting and managing 50% of all currently known and active monarch overwintering sites, including 90% of the most important overwintering sites by 2029;
- Providing a minimum of 50,000 additional acres of monarch-friendly habitat in California’s Central Valley and adjacent foothills by 2029.
- It also includes overwintering and breeding habitat conservation strategies, education and outreach strategies, and research and monitoring needs.

2.8.2 Environmental baseline

2.8.2.1 Species presence and use

Monarch butterflies are found throughout North America and are highly likely to utilize portions of the project area.

2.8.2.2 Species conservation needs within the action area

The Species Status Assessment Report, version 2.1 dated September 2020 discusses conservation needs. However, conservation needs within the action area have not been determined yet.

2.8.2.3 Habitat condition (general)

During migration to overwintering sites, monarchs need blooming nectar plants. On their return, monarchs are laying eggs and thus need both nectar sources and milkweed. The project area contains habitat that supports blooming nectar plants to potentially include milkweed.

2.8.2.4 Influences

Loss and degradation of habitat from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence, and incompatible management of overwintering sites in California, urban development, drought, exposure to insecticides, drought, and effects of climate change are all factors in the decline of the monarch population.

2.8.2.5 Additional baseline information

See Annex D3 Monarch “Pesticide Supplemental Material.” The following website also offers additional information https://www.fws.gov/initiative/pollinators/monarchs.

2.8.3 Effects of the action

2.8.3.1 Indirect interactions

Alternative C

Indirect impacts are expected due to the conversion of desired habitat to open water and elimination of food source.

Alternative CTO

While the specific features of the CTO alternative have not been determined, this alternative is could have similar impacts compared to Alternative C if the weir is included, as the preferred habitat would be converted to open water. However, if the weir is not included, the preferred habitat would not be converted to open water and therefore the impacts would be less.

2.8.3.2 Direct interactions

Alternative C

Direct impacts could be anticipated by way of collision with construction equipment. However, the species is highly mobile and the equipment is rather slow moving, so it is expected that any individuals present could escape the impact.

Alternative CTO

While the specific features of the CTO alternative have not been determined, this alternative is could have similar impacts compared to Alternative C if the weir is included, as the preferred habitat would be converted to open water. However, if the weir is not included, the preferred habitat would not be converted to open water and therefore the impacts would be less.

2.8.4 Cumulative effects

For purposes of consultation under ESA Section 7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under Section 7 of the ESA. At this time the USACE is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area.

2.7.5 Discussion and conclusion

Alternative C

USACE is continuing to review literature and is in coordination with the Service to finalize the effects of the action on the monarch butterfly. A determination has not yet been made.

CTO Alternative

A determination cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

3 Critical Habitat Effects Analysis

On March 19, 2003, The USFWS and NMFS published the Final Rule in the Federal Register designating critical habitat for the Gulf sturgeon.

Primary consideration must be given to the physical and biological features (PBFs) of the habitat under review that are essential to the conservation of the species and that may require special management considerations or protection.

The PBFs essential for the conservation of the Gulf sturgeon populations include those habitat components that support feeding, resting, and sheltering, reproduction, migration and physical features necessary for maintaining the natural processes that support these habitat components.

Based upon the identified PBFs for the Gulf sturgeon, the USFWS and NMFS identified a total of fourteen (14) Critical Habitat Units. Critical Habitat Unit 1 covers the proposed project area and includes the Pearl River System in St. Tammany and Washington Parishes in Louisiana and Walthall, Hancock, Pearl River, Marion, Lawrence, Simpson, Copiah, Hinds, Rankin and Pike Counties in Mississippi.

Of the 7 PBFs identified for Gulf sturgeon critical habitat, riverine spawning sites and riverine aggregation (resting) areas are not present in the action area. The PBFs found in the Action Area are food, flow regime, water quality, sediment quality, and migratory pathways.

The Pearl River is included in Critical Habitat Unit 1, the Pearl and Bogue Chitto Rivers in Louisiana and Mississippi, which is currently known to support a reproducing subpopulation of Gulf sturgeon. The Action Area occurs at the top extent of this Critical Habitat Unit.

While adult sturgeon do not usually feed in freshwater, juveniles forage extensively in rivers on aquatic insects, worms, and mollusks (Mason and Clugston 1993; Huff 1975; Sulak and Clugston 1999). with the varying aquatic species within the Action Area that feed on those types of prey it can be assumed that the area does contain enough of these prey items to support the populations of species that inhabit the area.

Suitable spawning substrate within the Pearl River likely includes soapstone, hard clay, gravel, and rubble areas and undercut banks adjacent to these substrates (W. Slack, pers. comm. 2001). Specific surveys have not been conducted on the substrate of the river within the Action Area; however, grab samples were taken as part of the Wetland Delineation conducted for the EIS/Feasibility Study that did not exhibit the suitable substrates necessary for sturgeon spawning in the Pearl River.

Gulf sturgeon depend on flow regimes in the riverine environment for all life stages including migration, breeding site selection, courtship, egg fertilization, resting and staging, and for maintaining spawning sites in the suitable condition needed for egg attachment, sheltering, resting, and larval staging. Based on average flow rates from 1966 to 2013, this area of the river currently has high flows during the springtime with flows decreasing significantly during the summer.

In 2019, a water advisory was issued for the Pearl River in Jackson due to continued discharges of sanitary sewer overflows into the river. In the Action Area, there is a former creosote plant as well as two former landfills from which debris periodically washes into the river. Leachates from these landfills were found to contain heavy metals above the regulatory standards. In 2003, the EPA also found barium, cobalt, zinc, and other contaminates in the river in the Action Area.

The PBFs of flow regime, sediment quality, and migratory pathways would not be impacted by the construction of Alternative C; therefore, only the effects on the PBFs of food and water quality will be discussed.

Increased turbidity and sedimentation would lead to impacts on water quality, which then leads to impacts on the prey base for juvenile sturgeon. These impacts on water quality would be temporary and would be reduced through erosion control measures.

Changes to water velocity, water surface elevation and water quality in the Action Area would be anticipated. DO and temperature are important water quality factors for sturgeon. As temperature increases, DO levels decrease which can affect the growth and respiration rates of juvenile sturgeon. Water quality modeling conducted for temperature and DO indicated post-project levels would have a slight but not significant difference from the pre-project levels.

Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat.

2.8 Other Protected Species

Other protected species, specifically bald eagles and migratory birds, have potential to be present in the study area. Bald eagles are protected under the Bald and Golden Eagle Protection Act (BGEPA) and the Migratory Bird Treaty Act (MBTA). 1,093 species of birds are protected under the MBTA.

The bald eagle was near extinction approximately forty years ago throughout most of its range. Habitat destruction and degradation, illegal shooting, and the contamination of its food source, largely as a consequence of DDT, decimated the eagle population. However, the banning of DDT, habitat protection, and conservation measures through the ESA, have afforded a remarkable recovery for the species. The bald eagle was removed from the endangered species list in 2007 but continues to be protected under the BGEPA and the MBTA.

Many of the 1,093 species of birds protected under the Migratory Bird Treaty Act are experiencing population declines due to increased threats across the landscape. Millions of acres of bird habitat are lost or degraded every year due to development, agriculture, and forestry practices. In addition, millions of birds are directly killed by human-caused sources such as collisions with man-made structures such as windows and communication towers.

Bald eagles nest in tall trees (usually cypress or pine in this area) near water and typically in the months of October through May. Migratory birds have varying nesting behaviors and seasons depending on the species. To be conservative, the nesting season for migratory birds is February 15 through September 15. Wading/water birds typically nest in trees or shrubs near water. Shorebirds typically nest on ground level in sand, small rocks, dunes, or ground vegetation. Many migratory birds (other than wading/water birds and shorebirds) are opportunistic nesters and will nest in trees, shrubs, building overhangs, house gutters, etc.

<u>Alternative C and CTO Alternative</u>

Direct impacts would be attributed to avoidance of the area during construction. Indirect impacts would be the elimination of potential roosting, foraging, and nesting habitat. Cumulative impacts, including both direct and indirect impacts of the alternative along with additional impacts from other, previous projects in the area are anticipated to be minor in intensity but long-term in duration. Impacts to the bald eagle and migratory birds from Alternative C would add to the impacts that have occurred over time and are expected to continue due to ongoing development and activities in and around the Project Area. A qualified biologist would survey the area prior to construction to determine the presence of nesting birds. If eagle nests are found in the project area, the USACE MVK would apply for an incidental eagle take permit and would implement avoidance and minimization measures described in the National Bald Eagle Management Guidelines until a permit with applicable requirements is received. Coordination with The Service and MDWFP would establish buffer zones and other guidelines to be implemented for nesting migratory birds depending on the species present. These impacts are considered insignificant.

4 Summary Discussion, Conclusion, And Effect Determinations

4.1 Effect Determination Summary (Alternative C)

SPECIES (COMMON NAME)	SCIENTIFIC NAME	LISTING STATUS	PRESENT IN ACTION AREA	EFFECT DETERMINATION Alt C
Gulf Sturgeon	*Acipenser oxyrhynchus desotoi*	Threatened	Yes	LAA
Ringed Sawback Turtle	*Graptemys oculifera*	Threatened	Yes	LAA

Northern Long-eared Bat	*Myotis septentrionalis*	Endangered	Yes	NLAA
Pearl River map Turtle	*Graptemys pearlensis*	Proposed Threatened	Yes	LAA
Alligator Snapping Turtle	*Macrochelys temminckii*	Proposed Threatened	Yes	NLAA
Louisiana pigtoe	*Pleurobema riddellii*	Proposed endangered	Yes	TBD
Tricolored bat	*Perimyotis subflavus)*	Proposed threatened	Yes	TBD
Monarch Butterfly	*Danaus plexippus*	Candidate	Yes	TBD

CTO Alternative

Determinations for listed species cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation.

4.2 Summary Discussion

Threatened and Endangered species and other protected species known to occur in the action area include GS, ringed map turtle, NLEB, PRMT, AST, LA pigtoe, TCB, monarch butterfly, bald eagle, and migratory birds. GS critical habitat also occurs within the action area. Alternative C would cause both temporary direct and long-term indirect impacts to species discussed. The project would eliminate and/or degrade habitat for GS, all three turtle species, the LA pigtoe, and the monarch butterfly; and would eliminate potential habitat for both bat species. Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat. Bald eagles and migratory birds could be impacted temporarily due to construction activities and long-term due to elimination of nesting and roosting habitat.

Determinations cannot be made at this time for the CTO. Once measures are identified and if the CTO is recommended for implementation, the USACE will re-initiate ESA consultation. At this time, it is assumed that if a weir is included in the CTO, then there would be similar impacts as Alternative C to GS, ringed map turtle, PRMT, AST, LA pigtoe, and monarch butterfly. It is also assumed that if clearing of forested areas are included that there would be similar impacts as Alternative C to NLEB and TCB. USACE will reinitiate ESA consultation if the CTO alternative is selected for implementation.

4.3 Conclusion

ESA consultation is ongoing. Based on currently available historical data, a review of current literature and studies, and with the employment of avoidance measures, the USACE has determined that Alternative C may affect but would not likely adversely affect the NLEB, and AST; would likely adversely affect the GS, ringed map turtle, and PRMT. USACE has not yet made a determination for the LA pigtoe, TCB, or the monarch butterfly. Based upon the assessment completed, it was determined that Alternative C would not result in an adverse modification to Gulf sturgeon critical habitat. Determinations cannot be made at this time for the CTO. Once measures are identified and if the CTO is selected for implementation, the USACE will re-initiate ESA consultation. USACE is continuing close coordination with the Service to finalize ESA consultation.

References

Amelon, S., and D. Burhans. 2006. Conservation assessment: *Myotis septentrionalis* (northern long-eared bat) in the eastern United States. Pages 69–82 in Conservation assessments for five forest bat species in the eastern United States, Thompson, F. R., III, editor. U.S. Department of Agriculture, Forest Service, North Central Research Station, General Technical Report NC-260. St. Paul, Minnesota.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. The University Press of Kentucky, Lexington, Kentucky.

Bennett, M.G., B.R. Kuhajda, J.H. Howell. 2008. Status of the frecklebelly madtom, Notorus munitus, (*Siluriformes: Ictaluridae*): a review of data from field surveys, museum records and the literature. Southeastern Naturalist.

Blute, G., M. Carrie and G. Blouin-Demers. 2010. Impact of recreational power boating on two populations of northern map turtles (*Graptemys geographica*). Aquatic Conservation Marine and Freshwater Ecosystems.

Buckner, M.M., W. Smith, J.A. Takats. 2002. Tennessee, Cumberland, and Mobile River Basins at Risk. A Biological Assessment and Vision for the World Wildlife Fund's Southeast Rivers and Streams Project. Nashville, TN. Available online: www.worldwildlife.org/what/wherewework/sers/WWFBinaryitem2738.pdf <Last accessed June 2020>

Buhlmann, K. and G. Coffman. 2001. Fire Ant Predation of Turtle Nests and Implications for the Strategy of Delayed Emergence. Journal of the Elisha Mitchell Scientific Society.

Buhlmann, K.A. 2014. Assessment of ringed map turtle (*Graptemys oculifera*) and Pearl map turtle (*Graptemys pearlensis*) abundance and habitat on the U.S. Navy's Stennis Western Maneuver Area East Pearl and Mike's Rivers, Mississippi, with recommendations for habitat enhancement. Unpublished report submitted to Stennis Space Center, Stennis Western Maneuver Area, Mississippi and Naval Facilities Engineering Command, Jacksonville, FL.

Cagle, F.R. 1952. The status of the turtles Graptemys pulchra Baur and Graptemys barbourin Carr and Marchand, with notes of their natural history. Copeia.

Carr, J.L., S.R. Holcomb, and M.J. Ray. 2011. Basking in the alligator snapping turtle, *Macrochelys temminckii* (Testudines: *Chelydridae*). IRCF Reptiles and Amphibians

Carter, T.C. and G.A. Feldhamer. 2005. Roost tree use by maternity colonies of Indiana bats and northern long-eared bats in southern Illinois. Forest Ecology and Management.

Chaffin, K., T.M. Norton, K. Gilardi, R. Poppenga, J.B. Jensen, P. Moler, C. Cray, E.S. Dierenfeld, T. Chen, M. Oliva, F.C. Origgi, S. Gibbs, L. Mazzaro, and J. Mazet. 2008.

Health assessment of free–ranging alligator snapping turtles (*Machrochelys temminckii*) in Georgia and Florida. Journal of Wildlife Diseases.

Chen, Tien-Hsi and Kuang-Yang Lue. 2009. Changes in the Population Structure and Diet of the Chinese Stripe-Necked Turtle (*Mauremys sinensis)* Inhabiting a Disturbed River in Northern Taiwan. Zoological Studies.

Cheong, T. S., M. L. Kavvas, and E. K. Anderson. 2006. Evaluation of Adult White Sturgeon Swimming Capabilities and Applications to Fishway Design. Envrion. Biol Fish.

Craft, N.M., B. Russell, and S. Travis. 2001. Identification of Gulf sturgeon spawning habitats and migratory patterns in the Yellow and Escambia River systems. Final Report to the Florida Marine Research Institute, Fish and Wildlife Conservation Commission.

DeCatanzaro, R., and P. Chow-Fraser. 2010. Relationship of road density and marsh condition to turtle assemblage characteristics in the Laurentian Great Lakes. Journal of Great Lakes Research.

DeWan, A., N. Dubois, K. Theoharides, and J. Boshoven. 2010. Understanding the impacts of climate change on fish and wildlife in North Carolina. Defenders of Wildlife Report, Washington D.C.

Dickerson D.D. and K.J. Reine. 1996. Habitat assessment and relative abundance estimates for the ringed sawback turtle (*Graptemys oculifera*) in dredging sites of the West Pearl River navigation project (Louisiana). Unpublished report. U.S. Army Engineer District, Vicksburg, Mississippi. 21 pp. + tables and figures.

Edwards, R.E., K.J. Sulak, M.T. Randall, and C.B. Grimes. 2003. Movements of Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in nearshore habitat as determined by acoustic telemetry. Gulf of Mexico Science.

Edwards, R.E., R.E., Parauka, F.M. and K.J. Sulak. 2007. New insights into marine migration and winter habitat of Gulf sturgeon. in: J. Munro, D. Hatin, J. Hightower, K. Sulak, and F. Caron (eds.). Proceedings of the Symposium on Anadromous Sturgeons. American Fisheries Society, Symposium, Bethesda, Maryland.

Ennen, J.R., J.E. Lovich, and R.L. Jones. 2016. *Graptemys pearlensis* Ennen, Lovich, Selman, and Qualls 2010 – Pearl River Map Turtle. In: Rhodin, A.G.J., Pritchard, P.C.J., van Dijk, P.P., Saumure, R.A., Buhlmann, K.A., Iverson, J.B., and Mittermeier, R.A. (Eds.). Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group, Chelonian Research Monographs.

Fitzgerald, L.A., and R.E. Nelson. 2011. Thermal biology and temperature-based habitat selection in a large aquatic ectotherm, the alligator snapping turtle, *Macroclemys temminckii*. Journal of Thermal Biology

Folkerts, G.W. 1997. State and fate of the world's aquatic fauna. Pgs. 1-16 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA.

Ford, N.B., L. Williams, M.G. Williams, J. Banta, J. Placyk, and H. Hawley. 2016. Final report: endangered species research projects for freshwater mussels, Region 2, East Texas for Texas Comptroller of Public Accounts. University of Texas at Tyler, Tyler, Texas.

Foster, A.M. and J.P. Clugston. 1997. Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society.

Foster, R.W., and A. Kurta. 1999. Roosting ecology of the Northern bat (*Myotis septentrionalis*) and comparisons with the endangered Indiana bat (*Myotis sodalis*). Journal of Mammalogy.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2000. Gulf sturgeon spawning migration and habitat in the Choctawhatchee river system. Alabama-Florida. Transactions of the American Fisheries Society129:811-826.

Griffiths J, Villablanca F. 2015. Managing monarch butterfly overwintering groves: Making room among the eucalyptus. California Fish and Game

Heise, R.J., S.T. Ross, M.F. Cashner, and W.T. Slack. 1999a. Movement and habitat use of the Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in the Pascagoula drainage of Mississippi: year 3. Museum Technical Report No. 74. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 14.

Howells, R.G. 2010a. Louisiana pigtoe (*Pleurobema riddellii*): summary of selected biological and ecological data for Texas. Biostudies, Kerrville, Texas.

Howey, C.A.F., and S.A. Dinkelacker. 2009. Habitat selection of the alligator snapping turtle (*Macrochelys temminckii*) in Arkansas. Journal of Herpetology.

Huff, J.A. 1975. Life History of the Gulf of Mexico Sturgeon, *Acipenser oxyrhynchus desotoi* in Suwannee River, Florida. Mar. Res. Publ. No. 16.

Hyzy, B.A., R.E. Russel, A. Silvis, W.M. Ford, J. Riddle, and K. Russel. 2020. Investigating maternity roost selection by northern long-eared bats at three sites in Wisconsin. Endangered Species Research.

Jackson, C.G., Jr., and A. Ross. 1971. The occurrence of barnacles on alligator snapping turtle, *Macroclemys temminckii* (Troost). Journal of Herpetology.

Jones, R. 2006. Reproduction and Nesting of the Endangered Ringed Map Turtle, *Graptemys oculifera*, in Mississippi. Chelonian Conservation and Biology.

Jones, R. 2017. Long-Term Trends in Ringed Sawback (*Graptemys oculifera*) Growth, Survivorship, Sex Ratios, and Population Sizes in the Pearl River, Mississippi. Chelonian Conservation and Biology.

Lindeman, P.V. 1998. Of deadwood and map turtles (*Graptemys*): An analysis of species status for five species in three river drainages using replicated spotting-scope counts of basking turtles. Linnaeus Fund research report. Chelonian Conservation and Biology.

Lindeman, P.V. 2013. The Map Turtle and Sawback Atlas: Ecology, Evolution, Distribution, and Conservation. University of Oklahoma Press, Norman, OK.

Lindeman, P.V. 2019. Population status of the megacephalic map turtles *Graptemys pearlensis* and *Graptemys gibbonsi*. Unpublished Final Report submitted to U.S. Fish and Wildlife Service, Jackson, MS.

Lorio, W. (ed.). 2000. Proceedings of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) status of the subspecies. Workshop held September 13-14, 2000, Mississippi State University, Science and Technology Research Center, Stennis Space Center, MS. 200 pp. and appendices.

Lovich, J.E., W. Selman, and C.J. McCoy. 2009. *Graptemys gibbonsi* Lovich and McCoy 1992. Pascagoula Map Turtle, Pearl River Map Turtle, Gibbon's Map Turtle. Chelonian Research Monographs.

Marchant, S.R. and M.K. Shutters. 1996. Artificial substrates collect Gulf sturgeon eggs. North American Journal of Fisheries Management.

Mason, W.T. Jr. and J.P. Clugston. 1993. Foods of the Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society

Mississippi Museum of Natural Science. 2015. Mississippi State Wildlife Action Plan. Mississippi Department of Wildlife, Fisheries, and Parks, Mississippi Museum of Natural Science, Jackson, Mississippi.

Moore, M. V., M. L. Pace, J. R. Mather, P. S. Murdoch, R. W. Howarth, C. L. Folt, C. Y. Chen, H. F. Hemond, P. A. Flebbe, and C. T. Driscoll. 1997. Potential effects of climate change on freshwater ecosystems of the New England/Mid-Atlantic Coastal Region, Climate Research.

Morrow, J.V, K.J. Killgore, J.P. Kirk, and H.E. Rogillio. 1996. Distribution and population attributes of Gulf sturgeon in the lower Pearl River System, Louisiana. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Mulholland, P.J. and D.R. Lenat. 1992. Streams of the southeastern piedmont, Atlantic drainage. In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY.

Neves, R.J., A.E. Bogan, J.D. Williams, S.A. Ahlstedt, P.W. Hartfield. 1997. Status of aquatic mollusks in the southeastern United States: a downward spiral of diversity. Pgs. 43-85 In: Benz, G.W. and D.E. Collins (editors). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. Available online: http://www.sherpaguides.com/southeast/aquatic_fauna/chapter_3/ <Last accessed July 2020>

Owen, S.F., M.A. Menzel, W.M. Ford, J.W. Edwards, B.R. Chapman, K.V. Miller, and P.B. Wood. 2002. Roost tree selection by maternal colonies of Northern long-eared Myotis in an intensively managed forest. USDA Forest Service. Newtown Square, Pennsylvania. 10 pp.

Parkyn, D.C., D.J. Murie, J.E. Harris, D.E. Colle, and J.D. Holloway. 2007. Seasonal movements of Gulf of Mexico sturgeon in the Suwannee River and estuary. American Fisheries Society Symposium.

Perry, R.W., and R.E. Thill. 2007. Roost selection by male and female northern long-eared bats in a pine-dominated landscape. Forest Ecology and Management.

Pine, W.E., H.J. Flowers, K.G. Johnson, and M.L. Jones. 2006. An assessment of Gulf sturgeon movement, spawning site selection, and post-spawn holding areas in the Apalachicola River, Florida. Final Report submitted to the Florida Fish and Wildlife Conservation Commission. University of Florida, Gainesville, FL.

Plummer, M., and N. Mills. 2008. Structure of an urban population of softshell turtles (*Apalone spinifera*) before and after severe stream alteration. Herpetological Conservation.

Pritchard, P.C.H. 1989. The alligator snapping turtle: Biology and conservation. Milwaukee Public Museum, Milwaukee, WI, USA.

Ross, S.T., R.J. Heise, W.T. Slack, J.A. Ewing, III, and M. Dugo. 2000. Movement and habitat use of the Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in the Pascagoula drainage of Mississippi: year 4. Mississippi Department of Wildlife, Fisheries, and Parks and Museum of Natural Science. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 15.

Sasse, D.B., and P.J. Pekins. 1996. Summer roosting ecology of northern long-eared bats (*Myotis septentrionalis*) in the white mountain national forest. Bats and Forests Symposium October 1995, Victoria, British Columbia, Canada.

Selman, W. and C. Qualls. 2007. Distribution, status, and conservation of the Pascagoula map turtle (*Graptemys gibbonsi*). Unpublished report submitted to U.S. Fish and Wildlife Service, Jackson, MS.

Selman, W., C. Qualls, and J. Owen. 2013. Effects of Human Disturbance on the Behavior and Physiology of an Imperiled Freshwater Turtle. The Journal of Wildlife Management.

Selman, W. and P.V. Lindeman. 2015. Life history and ecology of the Pascagoula map turtle (*Graptemys gibbonsi*). Herpetological Conservation and Biology .

Selman, W. and R. Jones. 2017. Population Structure, Status, and Conservation of Two Graptemys Species from the Pearl River, Mississippi. Journal of Herpetology, 51(1).

Selman, W. 2018. Diamonds in the rough: status of two imperiled Graptemys species (*Graptemys oculifera* and *G. peralensis*) in the Pearl River of Jackson, MS.

Silvis, A., R.W. Perry, and W.M. Ford. 2016. Relationships of three species of bats impacted by White-Nose Syndrome to forest condition and management. Gen. Tech. Rep. SRS–214. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Smith, G., J. Iverson and J. Rettig. 2018. Frequency of propeller damage in a turtle community in a northern Indiana, USA, lake; a long term study. Herpetological Conservation and Biology.

Soballe, D.M., B.L. Kimmel, R.H. Kennedy, and R.F. Gaugush. 1992. Reservoirs. Pgs. 421-474 In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY.

Solensky MJ. 2004. Overview of monarch migration. Pp. 79-83 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Sulak, K.J. and J.P. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desoto*i, in the Suwannee River, Florida, USA: a synopsis. Journal of Applied Ichthyology.

Sulak, K.J., M. Randall, J. Clugston, and W.H. Clark. 2004. Critical spawning habitat, early life history requirements, and other life history and population aspects of the Gulf sturgeon in the Suwannee River. Final Report to the Florida Fish and Wildlife Conservation Commission, Nongame Wildlife Program. U.S. Geological Survey, Gainesville, FL.

Sulak, K.J., M. T. Randall, R. E. Edwards, T. M. Summers, K. E. Luke, W. T. Smith, A. D. Norem, W. M. Harden, R. H. Lukens, F. Parauka, S. Bolden, and R. Lehnert. 2009.

Timpone, J.C., J.G. Boyles, K.L. Murray, D.P. Aubrey, and L.W. Robbins. 2010. Overlap in roosting habits of Indiana Bats (*Myotis sodalis*) and northern bats (*Myotis septentrionalis*). American Midland Naturalist.

USFWS. 2006. Status assessment. Pascagoula map turtle – *Graptemys gibbonsi.* Unpublished report, Mississippi Ecological Services Field Office, Jackson, Mississippi.

USFWS. 2011. Bogue Chitto National Wildlife Refuge Comprehensive Conservation Plan. Southeast Region, Atlanta, Georgia.

U.S. Fish and Wildlife Service. 2021. Species status assessment report for the Pearl River Map Turtle (*Graptemys pearlensis*), Version 1.1. April 2021. Atlanta, GA.

U.S. Navy. 2011. Integrated Natural Resources Management Plan. Naval Construction Battalion Center Gulfport, Stennis Western Maneuver Area, Stennis Space Center, Mississippi.

Vučenović, J. and P. V. Lindeman. 2020. The diets of the Pearl and Pascagoula Map Turtles (*Graptemys pearlensis* and *Graptemys gibbonsi*).

White R.G. and B. Mefford. 2002. Assessment of behavior and swimming ability of Yellowstone River sturgeon for design of fish passage devices. Montana Cooperative Fishery Research Unit, Montana State University; and Water Resources Research Laboratory, Bureau of Reclamation, Denver. Prepared for the Omaha District Corps of Engineers.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of the freshwater mussels of the United States and Canada. Fisheries.

Williams, J.D., A.E. Bogan, R.S. Butler, K.S. Cummings, J.T. Garner, J.L. Harris, N.A. Johnson, and G.T. Watters. 2017a. A revised list of the freshwater mussels (Mollusca: Bivalvia: Unionida) of the United States and Canada. Freshwater Mollusk Biology and Conservation.

Winston, M.R., C.M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. Transactions of the American Fisheries Society.

Wirth W., L. Schwarzkopf, L.F. Skerratt, and E. Ariel. 2018. Ranaviruses and reptiles. PeerJ 6:e6083 https://doi.org/10.7717/peerj.6083

United States Department of the Interior

FISH AND WILDLIFE SERVICE
Louisiana Ecological Services
200 Dulles Drive
Lafayette, Louisiana 70506

October 24, 2019

Mr. Walt Dinkelacker
President, Headwaters Inc.
PO Box 2836
Ridgeland, MS 39158

Dear Mr. Dinkelacker:

This document transmits the Fish and Wildlife Service's (Service) biological opinion (enclosed), regarding the Rankin Hinds Pearl River Flood and Drainage Control District's (District) Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, Mississippi (commonly referred to as the One Lake Project). The U.S. Army Corps of Engineers' (USACE) is authorized by Congressional actions to construct a flood risk reduction project; the District has undertaken the plan formulation and environmental compliance for that project's study.

The enclosed biological opinion addresses the proposed flood risk management projects effects on the ringed map (sawback) turtle (*Graptemys oculifera*), Gulf sturgeon (*Acipenser oxyrhynchus desotoi*), wood stork (*Mycteria americana*), the Northern long-eared bat (*Myotis septentrionalis*) and the inflated heelsplitter (*Potamilus inflatus*) in accordance with section 7 of the Endangered Species Act (Act) of 1973, as amended (16 United States Code [U.S.C.] 1531 *et seq*.).

The enclosed biological opinion, is based on information provided in the District's June 17, 2019, biological assessment (BA) and the August 23, 2019, revised BA. Additional information was also provided informally during the consultation process. A complete administrative record of this consultation (Service Log No. 04EL1000-2020-F-0109) is on file at the Service's Louisiana Ecological Services Office.

The Service appreciates the District's continued cooperation in the conservation of the threatened and endangered species, and their critical habitats. If you have any questions regarding the enclosed biological opinion, please contact Mr. David Walther (337-291-3122) of this office.

Sincerely,

Joseph A. Ranson
Field Supervisor
Louisiana Ecological Services Office

Biological Opinion

Pearl River Watershed, Hinds and Rankin Counties, Mississippi Flood Reduction Project

FWS Log #: 04EL1000-2020-F-0109

Prepared by:

U.S. Fish and Wildlife Service
Louisiana Ecological Services Field Office
200 Dulles Drive
Lafayette, LA 70506

October 23, 2019

TABLE OF CONTENTS

EXECUTIVE SUMMARY

This Endangered Species Act (ESA) Biological Opinion (BO) of the U.S. Fish and Wildlife Service (Service) addresses the potential effects of the Pearl River Basin, Mississippi, Federal Flood Risk Management Project, Hinds and Rankin Counties, Mississippi being proposed by the Rankin Hinds Pearl River Flood and Drainage Control District (FDCD). The U.S. Army Corps of Engineers Vicksburg District (USACE) by a January 31, 2018, letter has agreed that the FDCD will be the designated non-federal representative for the consultation. That project is proposed to provide economic and flood control benefits to the Jackson, Mississippi, area by the deepening and widening the floodplain and the installation of a new downstream weir. The FDCD determined that the Action is likely to adversely affect the ringed map (sawback) turtle (*Graptemys oculifera*), Gulf sturgeon (*Acipenser oxyrhynchus desotoi*) and its critical habitat and requested formal consultation with the Service. The BO concludes that the Action is not likely to jeopardize the continued existence of these species and is not likely to destroy or adversely modify designated critical habitat. This conclusion fulfills the requirements applicable to the Action for completing consultation under §7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended, with respect to these species and designated critical habitats.

The FDCD also determined and requested Service concurrence that the Action is not likely to adversely affect the wood stork (*Mycteria americana*), the Northern long-eared bat (*Myotis septentrionalis*) and the inflated heelsplitter (*Potamilus inflatus*); these species have no designated critical habitat within the project area. We provide our basis for this concurrence in section 3 of the BO. This concurrence fulfills the requirements applicable to the Action for completing consultation with respect to these species and designated critical habitats.

In addition, the BA addressed the previously listed bald eagle (*Haliaeetus leucocephalus*) and Louisiana black bear (*Ursus americanus luteolus*) and the extirpated (from the Pearl River drainage basin) pearl darter (*Percina aurora*). Because the eagle and the bear are no longer listed the ESA does not apply to them and the darter is not found in the project area thus it will not be impacted by the project therefore we will not address them in this BO.

The FDCD has developed a Channel Improvement Plan, also referred to as Alternative C, or the One Lake project, that consists of excavation of approximately 25 million cubic yards from the floodplain, extending from River Mile (RM) 284.0 to RM 293.5 (approximately 9.5 miles), and ranging in width from 400 to 2,000 feet. Some existing levees will be set back and new levees constructed with large amounts of fill areas placed behind them. The elevated land mass behind the levees will range from 200 to over 1,000 feet in width. To maintain water supply at the J. H. Fewell Water Treatment Plant (WTP) located at RM 290.7, an approximately 1,500-foot-long weir will be constructed at RM 284, creating a 1,500-acre pool area at the downstream limits of the project area and providing flood risk management benefits, recreation, and long-term maintenance reduction. The approximately 200-foot-wide existing weir at the J.H. Fewell WTP will be removed. Islands will be created from RM 289.5 to RM 292.0, some of which will be used to maintain and create habitat areas for local species.

It is the Service's opinion that the project would not jeopardize the ringed map turtle or the Gulf sturgeon nor destroy or adversely modify designated critical habitat for the Gulf sturgeon to the degree that it would result in jeopardy. The Service also concurred that the proposed Action is not likely to adversely affect the Northern long-eared bat*,* the wood stork, inflated heelsplitter and the pearl darter.

The BO includes an Incidental Take Statement that requires the USACE to implement reasonable and prudent measures that the Service considers necessary or appropriate to minimize the impacts of anticipated taking on the listed species. Incidental taking of listed species that is in compliance with the terms and conditions of this statement is exempted from the prohibitions against taking under the ESA.

The Action considered in this BO includes a conservation measure to relocate turtles from Crystal Lake within the construction area to the Lakeland population area and the relocation and protection of nests prior to construction would also be done. Creation and protection of nesting, basking and feeding habitat as well as the protection of approximately 10 miles of river bank and adjoining nesting and basking habitat are also included. In addition, the monitoring of the relocated turtles, nests and the population in the Action Area through the sampling, including but not limited to the capturing, tagging, tracking, observing and taking measurements, of individuals would be undertaken. Through the Incidental Take Statement, the Service authorizes these conservation measures as an exception to the prohibitions against trapping, capturing, or collecting listed species. These conservation measures are identified as a Reasonable and Prudent Measure below, and we provide Terms and Conditions for its implementation. Sampling protocols for the ringed map turtle should significantly reduce the likelihood of any lethal or injurious incidental take from occurring.

In the Conservation Recommendations section, the BO outlines voluntary Actions that are relevant to the conservation of the listed species addressed in this BO and are consistent with the authorities of the USACE.

Reinitiating consultation is required if the USACE retains discretionary involvement or control over the Action (or is authorized by law) when:

(a) the amount or extent of incidental take is exceeded;
(b) new information reveals that the Action may affect listed species or designated critical habitat in a manner or to an extent not considered in this BO;
(c) the Action is modified in a manner that causes effects to listed species or designated critical habitat not considered in this BO; or
(d) a new species is listed or critical habitat designated that the Action may affect.

CONSULTATION HISTORY

This section lists key events and correspondence during the course of this consultation. A complete administrative record of this consultation is on file in the Service's Louisiana Ecological Services Office.

2013-04-29 - Rankin Hinds Pearl River Flood and Drainage Control District (FDCD) holds an interagency meeting to discuss the proposed feasibility and environmental impact study regarding flood damage reduction alternatives along the Pearl River in Hinds and Rankin Counties, Mississippi.

2014-04-22 – Meeting with representatives of the FDCD and Mississippi Department of Wildlife Fisheries and Parks (MDWFP) to discuss potential alternatives and potential issues related to the gulf sturgeon and ringed map turtle.

2017-08-29 – Meeting with representatives of the FDCD and the USACE to discuss the ESA section 7 consultation process for the proposed project.

2018-01-31 – The USACE attached and submits the FDCD prepared biological assessment and requests formal consultation on the proposed project and its effects on federally listed species. The USACE designates the Rankin Hinds Pearl River Flood and Drainage Control District (FDCD) as the designated non-Federal representative that we work directly with during formal consultation process.

2018-03-08 – The Service informs the FDCD that formal consultation cannot be initiated until a complete BA is submitted; the Service provides comments on the BA and requests additional information.

2019–06-17 – The FDCD provides the Service with a revised BA.

2019–07-18 – The Service provides comments on the June BA and requests additional information.

2019–07-18a – The FDCD informs the Service that the June BA initiated formal consultation.

2019–07-19 – The Service agrees that formal consultation was initiated on June 17, 2019.

2019-08 -23 – The FDCD provides the Service with a revised BA and appendices containing hydrologic data for the project, engineer drawings of the projects structures, and fish passage.

BIOLOGICAL OPINION

1. INTRODUCTION

A biological opinion (BO) is the document that states the opinion of the U.S. Fish and Wildlife Service (Service) under the Endangered Species Act of 1973, as amended (ESA), as to whether a Federal Action is likely to:

- jeopardize the continued existence of species listed as endangered or threatened; or
- result in the destruction or adverse modification of designated critical habitat.

Updates to the regulations governing interagency consultation (50 CFR part 402) become effective on October 28, 2019 [84 FR 44976]. We are applying the updated regulations to this ongoing consultation. As the preamble to the final rule adopting the regulations noted, "[t]his final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice." We have reviewed the information and analyses relied upon to complete this BO in light of the updated regulations and conclude the BO is fully consistent with the updated regulations.

The Federal Action addressed in this BO is the proposed Pearl River Watershed, Hinds and Rankin Counties, Mississippi Flood Reduction Project (the Action) being developed by the FDCD. This BO considers the effects of the Action on the Gulf sturgeon (*Acipenser oxyrhynchus desotoi*) and ringed map (sawback) turtle (*Graptemys oculifera*), and designated critical habitat for the Gulf sturgeon.

The USACE determined that the Action is not likely to adversely affect the Northern long-eared bat (*Myotis septentrionalis*), the wood stork (*Mycteria americana*), inflated heelsplitter (*Potamilus inflatus*) and the pearl darter (*Percina aurora*). The Service concurs with these determinations, for reasons we explain in section 2 of the BO.

A BO evaluates the consequences to listed species and designated critical habitat caused by a Federal action, activities that would not occur but for the Federal action, and non-Federal actions unrelated to the proposed Action that are reasonably certain to occur (cumulative effects), relative to the status of listed species and the status of designated critical habitat. A Service opinion that concludes a proposed Federal action is *not* likely to jeopardize species and is *not* likely to destroy or adversely modify critical habitat fulfills the Federal agency's responsibilities under §7(a)(2) of the ESA.

"*Jeopardize the continued existence*" means to engage in an Action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02). "*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR §402.02).

This BO uses hierarchical numeric section headings. Primary (level-1) sections are labeled sequentially with a single digit (e.g., 2. PROPOSED ACTION). Secondary (level-2) sections within each primary section are labeled with two digits (e.g., 2.1. Action Area), and so on for level-3 sections. The basis of our opinion for each listed species and each designated critical habitat identified in the first paragraph of this introduction is wholly contained in a separate level-1 section that addresses its status, environmental baseline, effects of the Action, cumulative effects, and conclusion

2. PROPOSED ACTION

Under the authority of Section 211 of the Water Resources Development Act of 1996, the USACE assigned the FDCD as the non-federal sponsor to conduct the feasibility studies, environmental impact studies, and to optionally design and construct this federally authorized flood risk management project. The FDCD is proposing the Pearl River Watershed Project in Hinds and Rankin Counties, Mississippi. The purpose of the project is to provide flood damage risk management along the Pearl River in Hinds and Rankin Counties, Mississippi. The project would provide the flood reduction benefits, as well as maintain the water supply for the City of Jackson's Fewell Water Treatment Plant, and provide potential recreational benefits. The plan is also referred to as Alternative C, the Channel Improvement Plan or One Lake.

The proposed Action (Figure 2.1) includes the construction of a weir at RM 284; excavation of approximately 25 million cubic yards from approximately RM 284.0 to RM 293.5; and widening of an approximately 9.5-mile-long reach of the Pearl River. The newly excavated channel would range in width from approximately 400 to 2,000 feet. Excavated material would be placed adjacent to and behind existing levees; some material would also be placed within the floodplain to create islands from RM 289.5 to RM 292. Islands would be created for native wildlife and sandbars, and other natural features would be created throughout the area for turtle habitat. The channel would be excavated to varying depths to facilitate aquatic species habitat. Over 4 miles of an existing levee section along the eastern floodplain would be relocated further east reconnecting some of the floodplain and an existing weir structure located at RM 291 would be removed. The existing weir is approximately 200 feet in length and provides water for the City of Jackson's Fewell Water Treatment Plant. Downstream of the proposed weir (RM 284) an existing ring levee would be upgraded around the Savannah Waste Water Treatment Plant. The plant is located on the west bank of the river between RM 281 and RM 283. To the east of the proposed weir there would be a low flow diversion channel and a fish passage channel. North of the improved channel, a total of approximately 10 miles of river bank would also be protected; this Action is in accordance with the ringed map turtle recovery plan. The relocation of ringed map turtles from Crystal Lake and the relocation of nests from the excavation area is also planned. An adaptive management and monitoring plan will be developed in conjunction with the Service and the Mississippi Department of Wildlife, Fisheries and Parks (MDWFP) which would provide ongoing monitoring, long-term management, and habitat protection benefits for the listed turtle.

The Service analyzed impacts from the Action by dividing the project into impacts primarily associated with: 1) construction of the channel (e.g., excavation) and relocation of the levee and 2) impacts associated with the construction of the weir, its appurtenances and the impacts associated with the functions of the enlarged channel. Details of those features are described below in **Section 2.2 Channel Excavation and Levee Relocation** and **Section 2.3 Weir Construction and Impoundment**. Future detailed project planning may result in changes to project features and construction methods. Such changes may necessitate future consultations pending the extent and magnitude of the potential effects of those project modifications.

2.1. Action Area

For purposes of consultation under ESA §7, the Action Area is defined as "all areas to be affected directly or indirectly by the Federal Action and not merely the immediate area involved in the Action" (50 CFR §402.02). The BA describes the project area to include 2,450 acres along the main channel of the Pearl River from RM 301.77 to 284 in Hinds and Rankin Counties, Mississippi. The Service defines the "Action Area" for this consultation to include the portion of the Pearl River from the Ross Barnett spillway (RM 301.77) to 1.6 miles downstream of the proposed project weir at RM 284 (see following sections regarding the delineation of this area) (Figure 2.2). The Action Area also includes riparian areas adjacent to the river where construction activities will occur. The Action Area extends upstream of the proposed project to include all river miles that will be impacted by altered flow regimes, at approximately RM 301.77. The Action Area extends downstream (approximately 1.6 miles) of the proposed impoundment as this represents a sufficient downstream distance outside of the construction limits to determine if geomorphology and/or water quality impacts would occur as a result of the Action.

The Pearl River is formed in Neshoba County, Mississippi, by the confluence of Nanaway and Tallahaga Creeks and flows southwesterly for 130 miles to the vicinity of Jackson, then southeasterly for 233 miles to its outlet channels, the East Pearl and West Pearl Rivers (Lee 1985). The Action Area consists of the Pearl River floodplain from the Ross Barnett Dam to just south of Byram and includes land in Madison, Rankin, and Hinds Counties, Mississippi. The study area is drained by several small creeks that are tributaries of the Pearl River. Small tributaries to the Pearl River within the Action Area include Town, Hanging Moss, Eubanks, Lynch, Richland, Hardy, Caney, Purple, and Hog Creeks.

Immediately upstream of Jackson and on the Pearl River at River Mile (RM) 301.77 is the Ross Barnett Reservoir. The Pearl River Valley Water Supply District (PRVWSD) constructed the reservoir in the mid-1960s, and they retain authority for operation and maintenance of the project. The relatively shallow impoundment (mean depth of 12 feet) inundated approximately 24 miles of the Pearl River. In the northern part of Jackson, the City of Jackson built a low weir in 1915 at approximately RM 290.7 for water supply, which still provides a large portion of the city's water supply.

The 1960 Flood Control Act authorized construction of the Jackson (i.e., Fairgrounds) and East Jackson levees to address flooding in the area; the USACE completed that project in 1968 with an extension of the Jackson levee at Fortification Street completed in 1984. The existing flood control project consists of those two earthen levees on either side of the river totaling 13.2 miles. There is also channel work associated with the levees which includes 9.3 miles of enlargement and realignment of the main river channel through the town of Jackson (approximately 5 miles of cutoffs). Maintenance includes any necessary periodic removal of vegetation along a 650-foot-wide cleared strip of floodplain along the river and complete clearing downstream of that; a total of 346 acres of the floodplain (approximately 40 percent of the riparian area) is maintained in some form of cleared or partially cleared floodplain.

Two former landfills (Gallatin Street and Jefferson Street) and the former Gulf States Creosote plant are also located within the proposed project area. The 62-acre Gallatin Street landfill contains urban and industrial trash. Leachates from within the site contain cadmium, lead, and nickel above the regulatory standards. Debris from this landfill is reported to be washing into the river. The 45-acre Jefferson Street (or Lafleurs Landing) landfill also has debris that can be eroded during high river stages. The Environmental Protection Agency's (EPA) Final Preliminary Assessment/Site Inspection (PSA/SI) done in 2003 found barium, cobalt, manganese, and zinc, as well as creosote residuals consisting of a variety of polycyclic aromatic hydrocarbons (PAHs).

Downstream of the project area, the Pearl River flows through mostly rural areas. In this area between 76 and 90 percent of the land in counties adjacent to the river is forested (Oswalt 2013). There are many tributaries to the Pearl River south of the project area, but the two largest tributaries occur in the middle portion of the watershed. The Strong River (located at approximately RM 227) flows into the Pearl River just south of Georgetown, and Silver Creek (located at approximately RM 186) joins with the Pearl just south of Monticello. The Bogue Chitto River, located at approximately RM 37, is the largest tributary in the lower Pearl River watershed.

The lower portion of the Pearl River watershed has experienced more land conversion than the middle portion but less than around Jackson. Counties along the lower portion of the Pearl River have between 51 and 75 percent forested lands (Oswalt 2013). In the lower watershed, the Pearl River has been altered by the construction of two navigation channels, the Pearl River Navigation Channel and the West Pearl River Navigation Channel. The West Pearl River Navigation Channel includes three navigation locks in the channel and three sills (i.e., weirs approximately 12 feet in height). The sills are located on the Bogue Chitto River, the Pearl River at Pools Bluff, and near the southern navigation lock. The Pearl River Navigation Project resulted in the snagging and clearing of the river between Bogalusa, Louisiana, and Columbia, Mississippi. Downstream from approximately the latitude of Bogalusa, Louisiana, the Pearl River becomes a braided river system with numerous bifurcations.

<u>Hydrology</u>

The Ross Barnett Reservoir was constructed in 1961 and was filled by 1965. Operationally, the Ross Barnett Reservoir must maintain a minimum flow of 112 million gallons of water per day or approximately 170 cubic feet per second (cfs). This discharge rate is greater than low-flow discharge rates experienced preconstruction; however, downstream discharge of the Savanna Street Wastewater Treatment Facility is based on a critical low flow of 227 cfs. Thus, the minimal discharge from the reservoir at times could be below that required for adequate dilution and flushing of the wastewater facility's discharges. The Ross Barnett Reservoir is eutrophic with low dissolved oxygen (DO) levels documented in the summer months (EPA 1975; Mississippi DEQ 2018; Phallen et al. 1988).

Prior to and after construction of the Ross Barnett Reservoir, Pearl River flows varied primarily in response to rainfall in the basin (Hasse 2006). Groundwater discharge into some of the tributary streams also contributes to flows (Lang 1972; Lee 1985). Bednar (1976) postulated that during low discharge periods aquifer recharge could further reduce flows in the project area based on information collected approximately 2.3 miles south of the proposed weir. Because that study did not examine geological formations, the potential extent of possible recharge zones within the project area is unknown. The bed and banks of the river are primarily comprised of silts, sands, sandstone, and clays, including marl, with gravel deposits also present (Monroe 1954). Some limestone outcroppings occur along the banks as well (Crider 1906). Weathering of the clays can reduce their cohesiveness allowing the Pearl River to meander naturally in the floodplain (Monroe 1954).

An analysis of data from four stream gauge stations (Edinburg, Jackson, and Monticello, Mississippi, and Bogalusa, Louisiana) on the Pearl River for pre- (up to 1960) and post-Ross Barnett Dam and Reservoir construction (1964 – 2005) revealed that the same magnitude flood and low-flow events are recurring at greater magnitudes post-construction (Hasse 2006). The analysis indicated that the increase in magnitude of post-construction low flows is an effect of the reservoir. Also revealed was an increase in the median annual rainfall amounts in the upper and middle basin, which has resulted in an increase in the flows for the lower basin. Hasse (2006) also used the *Use of the Indicators of Hydrologic Alteration* software that examined 33 primary and 44 secondary parameters to provide a statistical analysis of changes in stream flows due to landscape changes and/or water resource projects. The greatest hydrologic alteration was observed at the Jackson station immediately downstream of the dam, with the degree of alteration decreasing in a downstream direction. However, hydrologic alteration was also detected at the Edinburg station upstream of the reservoir indicating that landscape and weather pattern changes are partially responsible for some of the alterations within the basin. It was estimated that approximately one-third of the alterations at the Jackson station and one-half at the Bogalusa station were related to landscape and weather pattern changes while the remaining were attributed to the reservoir. The parameters that showed the greatest alteration downstream of the reservoir include an increase in the number of low-flow pulses but a decrease in the low-flow duration at the Jackson and Monticello stations; these stations showed the same changes for high-pulse events as well. For the Bogalusa station the annual median number of low-flow pulses decreased post-reservoir but the annual duration of low-flow pulses increased; a similar trend for high flow events was also noted. The increase in the hydrograph rise and fall rates post-reservoir construction and the increase in hydrograph reversals are typically associated with flow alterations from dams (Hasse 2006).

Tipton et al. (2004) conducted a geomorphology investigation of the middle portion of the Pearl River between its confluence with the Strong River and Monticello, Mississippi. They examined sand bar stability between 1986 and 1999 and related it to the abundance of darters. Areas experiencing greater instability were found in the lower part of their study area and those areas had fewer darters. Kennedy and Hasse (2009) also conducted a geomorphology study of the entire basin below the Ross Barnett Reservoir. Their study was multi-facetted and reported that the Ross Barnett Reservoir almost entirely removed the upper one-third of the drainage basin from contributing sediment, which has resulted in the incision and degradation of the Pearl and Strong rivers. During flood stages, the floodplain captures large quantities of suspended sediments, especially below the confluence with the Strong River. The upper Pearl River (but below the Ross Barnette Dam) is also a major contributor to sediment loads due to the instability of the river and the resulting bank erosion. Instability of the river decreases downstream but is still an important source of excess sediment. The pool created by Pools Bluff Sill acts to stabilize the channel and bank conditions in that area of the lower Pearl River. Downstream of that sill there is an increase in channel stability with most of the instability being primarily related to sand and gravel mining, but also to the navigation channel. Kennedy and Hass (2009) compared their analysis to Tipton et al. (2004) and asserted that the area of instability identified by Tipton et al. (2004) may be migrating downstream. Piller et al. (2004) reported that the Pearl River south of its confluence with the Strong River had undergone a dramatic change, with gravel substrates being replaced with unstable sand substrate following construction of the reservoir.

Conversely, the examination of data from three gauges from within and downstream of the project area (i.e., Jackson, Byram, and Rockport) was performed during the feasibility study by contractors for the FDCD to determine possible changes in discharge and stage (i.e., water level or gauge height) relationship to determine if the Pearl River had undergone any channel changes. Based on that examination it was concluded that the construction of the reservoir, land use changes, urbanization, and channel improvement could have resulted in some instability but has since re-stabilized and remained in a state of dynamic equilibrium. The Jackson gauge used in the analysis is located approximately 1.3 miles upstream of the proposed weir (i.e., within the project area), while the Byram and Rockport gauges are located approximately 14 and 40 miles downstream of the proposed weir, respectively. Because stage-discharge measurements are not taken continually the data represents periodic measurements over the years. Data from the Jackson gauge included the years from 1929-1972, 1973-1977, 1978-1989, and 1990-2010. The Byram gauge included data from only 1984 to 1993 while the Rockport gauge had data from 1940-1949, 1984-1991, and 1992-2010. Based on the examination of that data the stage-discharge relationship was determined to be stable for the Jackson and Byram gauges (Graphs 2.1 and 2.2). For the Rockport gauge (Graph 2.3) there was a slight lowering of the stages (generally less than a foot) for discharges between 32,000 and 51,000 cfs for the time period between the 1940's and 1980's and there was also a possible lowering of the stages for flows less than 4,000 cfs between the 1984-1991 and the 1992-2010 period. The Rockport gauge is located in the same reach of the river where Tipton et al. (2004) and Piller et al. (2004) reported some instability during the later time period.

For the USACE 1996 Draft Environmental Impact Statement (EIS) and Feasibility Study an examination of the river was also undertaken. That examination determined that the upper reach extending 10 miles downstream of the reservoir consisted of mostly fine to medium sands and near vertical banks that are eroding resulting in a major source of sediment to the system. The middle reach (next six miles) consists of the reach altered by previous flood control projects and that reach appeared to be stable but with some signs of degradation. The lower reach (next 15 miles) consisted of a meandering channel with areas of aggradation and degradation. It was noted that the reservoir has reduced the sediment discharge downstream of the dam with some channel degradation, but no significant instability has occurred.

In addition, the contractors for the FDCD examined Google Earth imagery from 1996-2010 to assist in determining bank erosion. For the 16-mile-long project area, eight areas of erosion were identified with six of the sites occurring between the reservoir and Highway 25; the remaining sites were downstream. That examination determined that 6.5 percent of the study area was experiencing low to moderate meander migration and no significant channel changes were seen. Examination of river banks were also conducted, and based on that examination it was determined that the Action Area is relatively stable with localized erosion and that the channel may have experienced some degradation in the past, but there was no indication of instability based on limited field observation.

Hydrologic modeling of the Action Area indicated that the range of velocities within the river varied with the cross-section of the river and floodplain and the river's discharge (Graph 2.4).

Average cross-sectional velocities varied from approximately 0.27 feet per second (fps) to 2.2 fps.

Overall the Pear River Basin has undergone alterations due to changes in the landscape (e.g., land clearing, navigation, flood control) that impact the ecological functions of the area. These ongoing impacts have led to the reduction and/or loss of habitat which has resulted in the listing of species under the ESA. Declines in other species endemic to the Pearl River and adjacent watersheds because of the ongoing alterations may result in the additional listing of other species. A comprehensive watershed assessment should be undertaken to identify pro-active measures that would ensure the protection of fish and wildlife values in the basin while achieving socio-economic needs.

2.2. Channel Excavation and Levee Relocation

The proposed Action consists of the excavation of approximately 25 million cubic yards from approximately RM 284.0 to RM 293.5. The channel widening would range in width from approximately 400 to 2,000 feet. The channel would be excavated to varying depths to facilitate aquatic species habitat. It would also include the relocation of over approximately 4 miles of a levee further away from the river thus reconnecting some of the floodplain. In addition, the construction of a 1,500-foot-wide weir structure at approximately RM 284.0 to create a 1,901-acre improved channel (i.e., lake). Earthen material removed from the floodplain and river would be used to create approximately 947 acres of elevated fill adjacent to the excavated area and levees.

Activities needed to accomplish this work would include clearing and grubbing along all of the rights-of-way (ROWs) for all project features, construction of staging areas and access roads, and hauling of earthen fill for the levee. An existing 200-foot-wide weir for drinking water retention located at RM 291 within the project footprint would also be removed. The plan also includes installation of a 12-foot by 12-foot gate structure near and east of the weir to maintain minimum flows through the river channel system. A fish by-pass channel around the weir and low flow structure would be constructed on the east bank of the river.

The project would also include the creation of islands from approximately RM 289.5 to RM 292.0 to create and maintain habitat for wildlife species common to the area. In addition, to replace the approximately 31.4 acres of sandbars that would be lost, an equal or greater acreage would be recreated for turtle nesting habitat. The sandbars would be approximately 1 to 15 acres in size with sand approximately 2 feet deep. The sandbar would be no wider than 75 yards from the water line. The central ridge of the island should be 7 to 8 feet higher than the edges and vegetated with a narrow (<20 yards) strip of river birch or black willow trees. The created sandbars in conjunction with the proposed islands would be monitored and maintained through the life of the project to ensure that vegetative cover does not overtake the created open sand nesting areas. No wake zones would be established around the sandbars and human disturbance would be prohibited. Enforcement of the wake and disturbance restrictions would be within the authority of and undertaken by members of the FDCD. The sandbars would also be surrounded by tree tops and downed trees to create at least short-term basking and foraging areas and also serve to protect turtles from predation. The tree tops and downed trees would be placed approximately 10 to 20 feet apart around the created islands and along any of the shoreline that would be available for such uses.

Approximately 10 miles of river bank would also be protected. The prioritized areas where this land would be located is; 1) north of the improved channel, 2) north of the Ross Barnett Reservoir, and 3) south of the weir. This action is in accordance with the ringed map turtle recovery plan. The relocation of ringed map turtles from Crystal Lake and the relocation of nests from the excavation area is also planned. An adaptive management and monitoring plan will be developed in conjunction with the Service and the MDWFP which would provide ongoing monitoring, long-term management, and habitat protection benefits for the listed turtle.

Capping and stabilization of the Lafleur's and Gallatin Street Landfills would be undertaken, while some mitigative measures may be required at the Gulf States site. Further investigations to be undertaken in the detailed design phase are required to fully determine the extent of remediation needed. Remediation should reduce leachates from flowing into the Pearl River.

Excavation of the 25 million cubic yards would destroy approximately 1,433.5 acres of forested or scrub-shrub wetlands, 31.41 acres of accretion (e.g., sandbar, sandbank), and 65.1 acres of emergent wetlands. A total of 1,901 acres would be excavated and 947 acres would have earthen fill placed on them. Of the 1,901 acres to be excavated, 230.80 acres currently exist as the Pearl River.

2.3. Weir Construction and Impoundment

The proposed Action also includes the construction of an approximately 1,500-foot-wide weir located at RM 284. The top elevation of the weir would be at 258 feet North American Vertical Datum 1988 (NAVD 88). The weir will create an approximately 1,500-acre impoundment stretching from RM 284 to approximately RM 293 with an average depth of 22 feet. Current average depth is 6.7 feet. A 12-foot by 12-foot gate and culvert structure would be built to the east of the new weir to maintain minimum flows through the impoundment during low flow periods. The bottom elevation of the culvert on the upstream side would be approximately 248 feet (NAVD 88) while the downstream side would connect to the existing channel at an elevation of approximately 230 feet (NAVD 88). An approximately 7,300-foot-long channel for fish passage would be constructed east of the low-flow structure and would have an upstream bottom elevation of 256 feet (NAVD 88) and the downstream bottom elevation would be 230 feet (NAVD 88) where it connects to the river channel.

Activities would also include construction of an approximately 900-foot-long embankment with a top elevation of 260 feet (NAVD 88) within the floodplain to connect the weir to the fill areas on each side; the weir would be approximately centered in this embankment. Activities would include clearing and grubbing along all the ROWs for all project features, construction of staging areas and access roads, and hauling of earthen fill for the levee. Excavation of the weir site, low-flow structure, and fish passage channel would be necessary. Placement of erosion resistant material (e.g., stone or concrete) would be needed downstream of the weir, within the low-flow channel, and in the fish passage channel.

The construction plan indicates that most of the excavation from the Pearl River floodplain would occur during the dry season when the likelihood of out-of-bank flows is reduced. This provides a progressive level of flood risk management during construction and helps to minimize impacts to water quality and quantity. With flow contained within the River, the sediment load would not be impacted by the off-line excavation process during within bank flow periods. Once constructed, the weir would fill by local rainfall events. The required minimum flows from the Ross Barnett Reservoir would be maintained at all times during construction. Once filled, the discharge over the weir and through the fish passage channel is designed to match the discharge from the Ross Barnett Reservoir.

Because the low flow structure is designed to meet the required discharge of the Ross Barnett Reservoir, there will not be a change in the discharge from the proposed project. Average monthly discharge, along with the standard deviation and minimum monthly discharge from 1966 to 2013, are presented in Table 2.1. Typically, June through October have the lowest discharge while December through April have the highest discharge. May and November have discharges that transition between the high and low periods. The percentages of months having discharges less than 1,000, 2,000, 5,000, 10,000, 20,000 and 40,000 cubic feet per second (cfs) are presented in Table 2.2. In general, discharges greater than 5,000 cfs do not occur between June and November. Discharges greater than 20,000 cfs occur infrequently between December and May; that is most discharge rates are less than 20,000 cfs during that time period.

The range of velocities and water surface elevations presented in the tables below represent various flows with the 1,000 cfs discharge typically being equaled or exceeded about 54 percent of the time, the 2,000 cfs flow would be equaled or exceeded 42 percent of the time, the 5,000 cfs flow being equaled or exceeded 26 percent of the time, and 10,000 cfs flow being equaled or exceeded 13 percent of the time. Most of the discharges have their typical reoccurrence interval presented within the profile column. The weir would elevate the water surface within the Action Area from 258.1 feet (NAVD 88) to an approximate elevation of 260.95 feet (NAVD 88) for a river discharge of 20,000 cfs just upstream of the weir. Additional changes in water surface elevation are presented in Tables 2.3 through 2.7. As shown in the tables, the weir would elevate the water surface near the weir with greater differences being experienced when the river would have normally been at low flow conditions and smaller differences during larger discharge events.

Velocity differences within the channel would also occur (velocities presented in the tables and graphs are an average over the channel's cross section) with velocities being reduced for the length of the project (Graph 2.1). This trend remains fairly constant throughout the improved channel portion (Tables 2.3 through 2.5 and Graph 2.4) with variations caused primarily by differences in the proposed cross-section of the channel. Upstream of the approximate upper limit of the pool area (between RM 293 and 294) the trend begins to diminish (Table 2.6 and Graph 2.4), but the influence of the weir is still detectable up to approximately RM 295.7.

Based on an ANOVA analysis of the 20,000 cfs and 40,000 cfs discharges the post-project velocities will be significantly reduced for the entire project area at 20,000 cfs. Post-project velocities will be significantly reduced in the improved channel reach and will increase in the upstream reach. The channelized reach is projected to have reduced velocities at all discharges below 20,000 cfs, but not at 40,000 cfs or greater; whereas the upper reach will see post-project velocity increases at 40,000 cfs and greater but not at 20,000 cfs or below. Once discharges decrease below 10,000 cfs, the improved channel's velocities would significantly decrease and lake like velocities would occur.

Between RM 293.9 (upper end of the improved channel) and RM 295.9 the river and floodplain will not be altered, but the water surface elevation will be reduced several feet for discharges between 10,000 cfs and 50,000 cfs. In this same general area there will be an increase in velocities (i.e., 1.28 feet per second [fps] to 5.85 fps) for discharges greater than 40,000 cfs (shaded area in Graph 2.4). This decrease in water surface and increase in velocities could result in scouring and destabilization of the banks (i.e., head cutting); however, analysis of the sheer strength (resistance to erosion) values within this reach would be well below the critical thresholds that would cause channel instability. This reach would be monitored for any changes in channel stability once constructed.

The weir is designed to be overtopped by the discharges occurring at the one-year frequency or greater. Studies have investigated geomorphological impacts from similar weirs. Gangloff et al. (2011) found narrower channel widths in streams with intact weirs. Helms et al. (2011) found intense sedimentation and altered geomorphology in upstream areas and immediately downstream of the weir. Pearson et al. (2016) observed that floodplains upstream of dams received larger amounts of sediment (including sand) during over bank floods. Ciski (2014) found that weirs with tops below channel banks still captured fine sediments and sand, but trapping of fines was minor and no major discontinuities in river morphology or sediment characteristics occurred. Skalak et al. (2008) discovered coarsening of downstream sediments. Ciski and Rhoads (2010) observed that if the weir does trap sediment, then downstream erosion of channel banks and the channel bed will occur through the formation of an inflection point in the water surface profile; this inflection or “nick” point would migrate toward the structure diminishing the extent of the backwater (i.e., sedimentation) zone. Sluice gates within the structure helped pass sediments downstream. Fencl et al. (2017) also found that the substrate coarsened downstream, but that a maximum of 1.6 miles downstream, the substrate returned to reference site conditions. The downstream area altered by weirs (i.e., widening and substrate changes) ranged from 0.13 to 1.6 miles with an average of 0.75 miles. The changes in the river’s width and depth depended on local factors including geology, channel confinement, slope, and height of dam compared to bank height. Sedimentation starvation below dams can reduce the effect of downstream low-head dams. Upstream areas experienced an increase in mean depths. The impacted upstream area can vary by the slope of the river and the height of the weir.

To assess the potential capture of sediments, the FDCD contracted with Tetra Tech to develop a model of the area to compare existing conditions against those with the project constructed. The Tetra Tech model was developed on behalf of the Mississippi Department of Environmental Quality (MDEQ). This model uses Environmental Fluid Dynamics Code (EFDC) and Water Quality Analysis Simulation Program (WASP) to create a dynamic one-dimensional model from Jackson, MS, to Bogalusa, LA, and simulates hydraulics and water quality from January 1, 2000, through December 31, 2017. In addition to using 18 years of data, the model accounts for many hydraulic variables, including discharge flows (Table 2.7) and total suspended solids (Table 2.7). Implementation of the project results in less than 0.3 percent change in either direction on either variable. Based on this analysis they determined that the project is not predicted to impact sediment load or downstream discharges (and thereby downstream velocities); thus, the project would not be expected to affect the amount of sediment that would or would not be picked up downstream of the project area. However, within the Engineering Appendix a preliminary sediment transport analysis was conducted. That analysis indicated a reduction of sediment transport, especially at lower flows approximately between RM 285 and 290. This would indicate a potential sediment sink within the lake portion; and the appendix did state the need for additional sediment analysis. Reduced sediment transport could result in increased downstream erosion. To address that issue, monitoring at the weir and downstream for 1.6 miles would be incorporated into the monitoring and adaptive management plan.

To assess water quality the same Tetra Tech model was used. Parameters examined included temperature, dissolved oxygen (DO), total phosphorus, total suspended solids, biological oxygen demand (BOD), total nitrogen and chlorophyll a. Slight differences were noted for many of the parameters (Table 2.7) but no significant adverse effects were revealed.

2.4. Non-Federal Activities caused by the Federal Action

A BO evaluates the effects of a proposed Federal action. "Effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 CFR §402.02).

Alternative C includes the construction of additional natural areas and parks within significant portions of the project fill areas. Non-consumptive activities, such as hiking, outdoor photography, and wildlife viewing, would increase as these areas would be publicly available. These areas would complement Lefleur's Bluff State Park. Conversion of the forestland and other habitat types that currently exist and are inaccessible to water, will occur with the implementation of Alternative C. This alternative would increase water-dependent recreational opportunities, such as fishing, boating, and canoeing through additional public access such as boat ramps. Non-consumptive uses would increase because of the inclusion of multipurpose trails, wildlife viewing areas, amphitheaters, and campgrounds. The additional public access boat ramps and pedestrian access points associated with this alternative would increase recreation within the project area. Alternative C would improve access to the riverfront, increasing the opportunity for public recreational utilization.

Activities that would not occur but for the proposed Federal action include relocation or retrofitting of existing infrastructure within the action area (i.e. roads, bridges, pipelines, powerlines), riverfront access and development. These activities are expected to increase recreational opportunities, which will stimulate community development, population, and housing. Increased recreational use of the river from the upper end of the pool to reservoir could occur similar to consequences north of the Ross Barnett Reservoir; currently recreational activities do occur but are not as great (Selman and Jones 2017). Alternative C also has the inclusion of a fish passage channel next to the weir structure so that the weir would be less of an impediment to Gulf sturgeon.

2.5. Tables and Figures for Proposed Action

Figure 2.1 Proposed Channel Improvement Plan

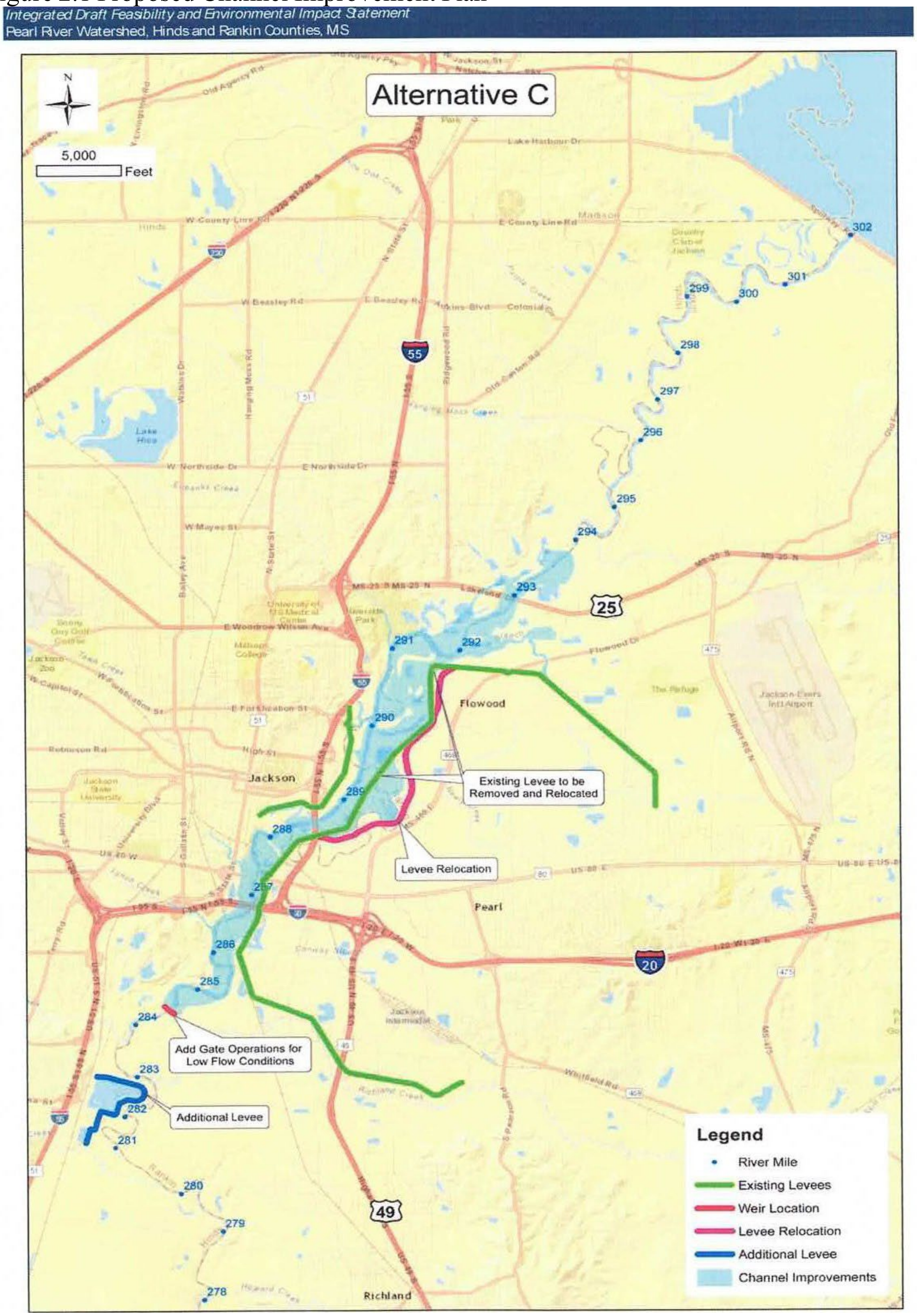
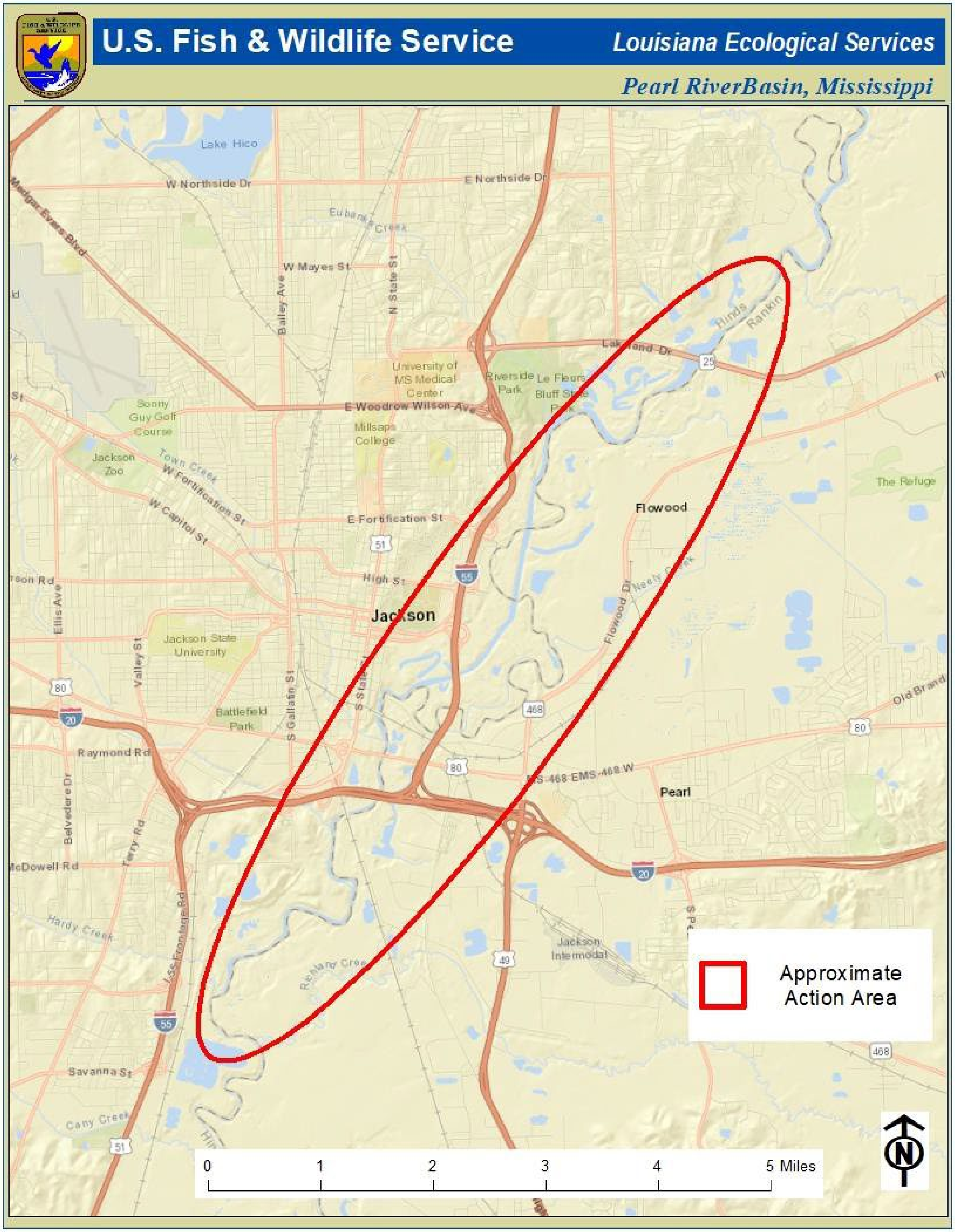
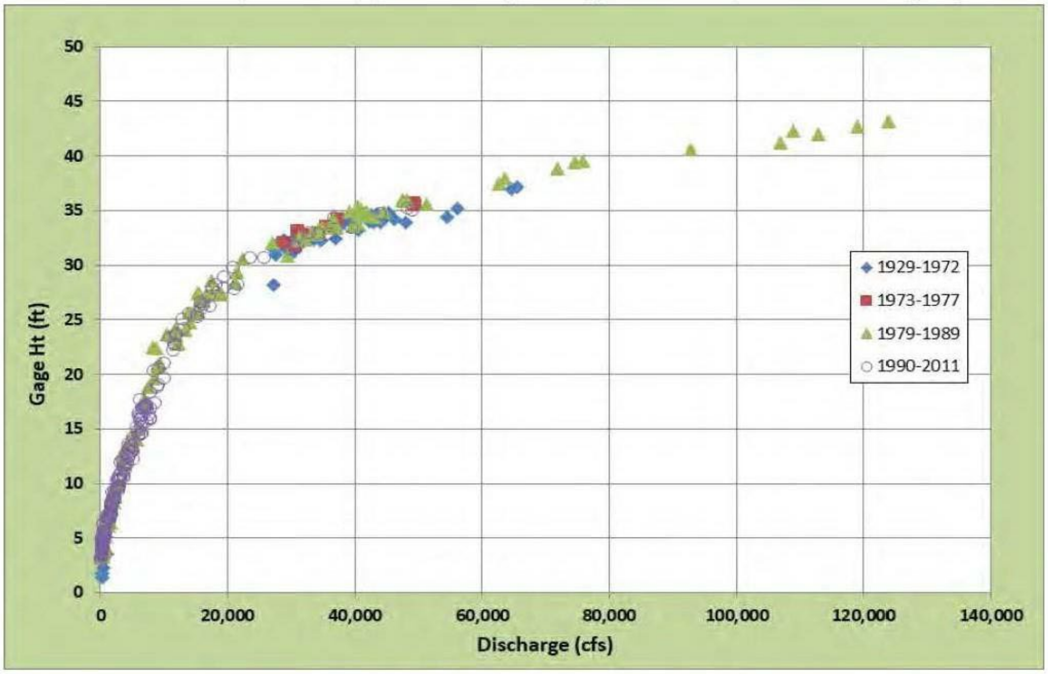


Figure 2.2 Action Area



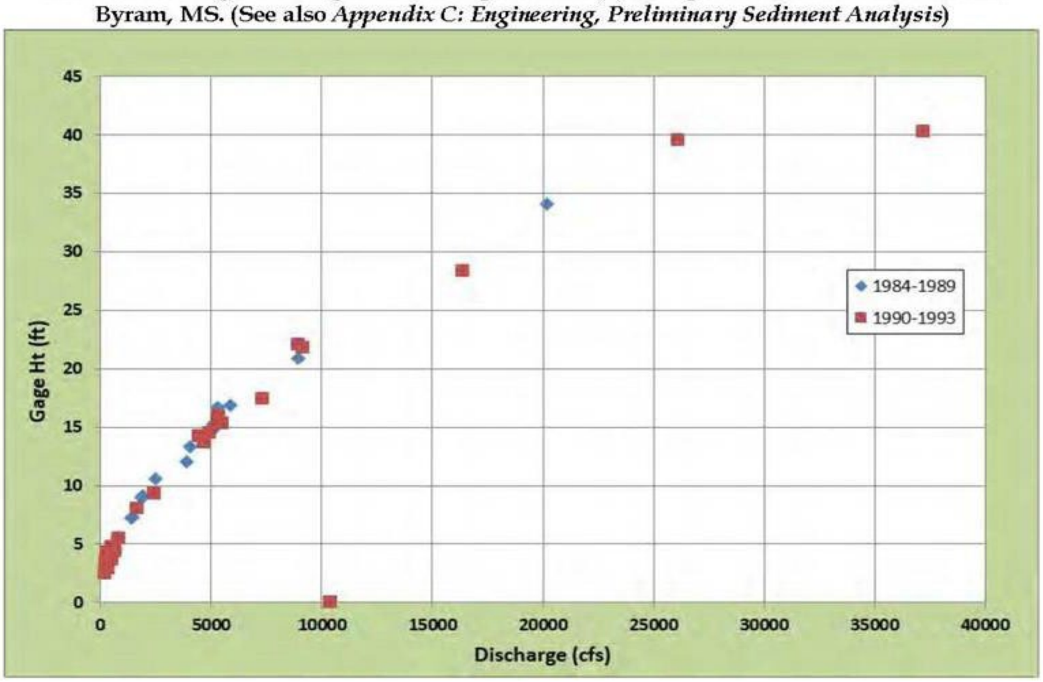
Graph 2.1

Exhibit A.1: Stage-discharge relationships for four (4) time periods for the Pearl River at Jackson, MS. (See also *Appendix C: Engineering, Preliminary Sediment Analysis*)



Graph 2.2

Exhibit A.2: Stage-discharge relationships for two (2) time periods for the Pearl River at Byram, MS. (See also *Appendix C: Engineering, Preliminary Sediment Analysis*)



Graph 2.3

Exhibit A.3: Stage-discharge relationships for four time periods for the Pearl River at Rockport, MS. (See also *Appendix C: Engineering, Preliminary Sediment Analysis*)

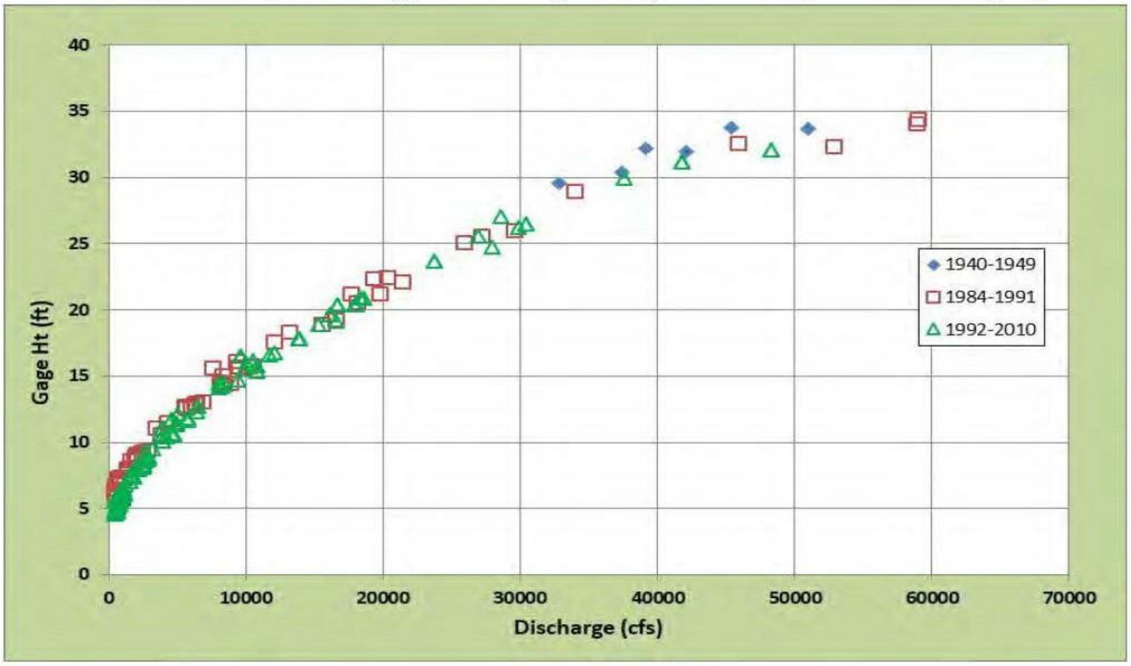


Table 2.1. Monthly average discharge (cfs), 1 Standard Deviation (STD) and minimum monthly average flow 1966-2013.

	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec
Average	8333	9303	9101	8183	4312	1562	1154	961	1140	1331	2078	5421
1 STD	5920	5875	4914	7700	4816	1734	1330	1237	1683	2313	1967	4868
Minimum	338	321	1233	412	256	183	180	197	208	195	142	298

Table 2.2. Percent of months having discharge less than the rate indicated from 1966-2013.

Discharge cfs	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec
<1000	8	2	0	8	23	52	65	65	75	77	43	11
<2000	15	4	6	21	42	81	85	92	85	85	57	26
<5000	31	29	21	42	67	92	96	98	96	94	89	57
<10,000	71	56	65	71	92	100	100	100	100	98	100	87
<20,000	96	96	96	92	98	100	100	100	100	100	100	98
<40,000	100	100	100	100	100	100	100	100	100	100	100	100

Table 2.3. Hydrologic information for just above the proposed weir

River Mile	Profile	Plan	Discharge	Water Surface Elevation	Velocity Total	Area	Top Width
			(cfs)	(ft)	(ft/s)	(sq ft)	(ft)
284.833	-	Existing	1000.00	238.43	1.17	851.32	137.08
284.833	-	Alt C	1000.00	258.40	0.05	22369.18	3182.28
284.833	-	Existing	10000.00	252.55	2.08	7497.75	1660.40
284.833	-	Alt C	10000.00	259.86	0.45	27544.17	3932.60
284.833	-	Existing	20000.00	258.10	2.09	19673.79	3698.12
284.833	-	Alt C	20000.00	260.95	0.83	32053.53	4378.40
284.833	2 YR	Existing	40000.00	264.05	1.67	55605.10	9560.92
284.833	2 YR	Alt C	40000.00	264.27	1.21	53155.65	8360.21
284.833	5 YR	Existing	50000.00	266.13	1.40	76860.01	10733.41
284.833	5 YR	Alt C	50000.00	266.32	1.18	71379.16	9239.55
284.833	10 YR	Existing	56800.00	267.20	1.35	88425.55	10858.47
284.833	10 YR	Alt C	56800.00	267.39	1.21	81291.73	9339.00
284.833	25 YR	Existing	73000.00	269.36	1.32	112014.10	10992.36
284.833	25 YR	Alt C	73000.00	269.55	1.27	101675.40	9469.48
284.833	50 YR	Existing	90000.00	271.42	1.23	134758.50	11070.64
284.833	50 YR	Alt C	90000.00	271.60	1.25	121130.50	9517.18
284.833	100 YR	Existing	106000.00	272.86	1.23	150825.60	11154.90
284.833	100 YR	Alt C	106000.00	273.06	1.28	135026.70	9594.28

Table 2.4. Hydrologic information for the area between I-20 and US 80

River Mile	Profile	Plan	Discharge	Water Surface Elevation	Velocity Total	Area	Top Width
			(cfs)	(ft)	(ft/s)	(sq ft)	(ft)
287.14	-	Existing	1000.00	240.61	1.17	853.71	261.04
287.14	-	Alt C	1000.00	258.40	0.05	19424.23	1538.63
287.14	-	Existing	10000.00	254.29	1.14	8748.41	924.12
287.14	-	Alt C	10000.00	259.89	0.46	21717.10	1549.71
287.14	-	Existing	20000.00	259.79	1.40	17500.82	2262.55
287.14	-	Alt C	20000.00	261.03	0.85	23490.22	1557.56
287.14	2 YR	Existing	40000.00	265.84	1.45	31340.73	2310.40
287.14	2 YR	Alt C	40000.00	264.44	1.39	28841.54	1580.90
287.14	5 YR	Existing	50000.00	267.95	1.53	36962.38	2923.83
287.14	5 YR	Alt C	50000.00	266.51	1.56	32175.60	1786.84
287.14	10 YR	Existing	56800.00	269.10	1.61	40784.96	3839.39
287.14	10 YR	Alt C	56800.00	267.60	1.68	34312.81	2080.83
287.14	25 YR	Existing	73000.00	271.49	1.77	50877.87	4396.79
287.14	25 YR	Alt C	73000.00	269.82	1.95	40371.36	3382.07
287.14	50 YR	Existing	90000.00	273.71	1.93	60780.36	4521.68
287.14	50 YR	Alt C	90000.00	272.07	2.18	48410.02	3690.79
287.14	100 YR	Existing	106000.00	275.31	2.05	68093.63	4626.52
287.14	100 YR	Alt C	106000.00	273.53	2.43	53838.36	3779.51

Table 2.5. Hydrologic Information for the area between E. Fortification St. and the Water Works Weir							
River Mile	Profile	Plan	Discharge	Water Surface Elevation	Velocity Total	Area	Top Width
			(cfs)	(ft)	(ft/s)	(sq ft)	(ft)
290.45	-	Existing	1000.00	245.90	0.95	1048.46	317.47
290.45	-	Alt C	1000.00	258.40	0.05	21878.94	2269.97
290.45	-	Existing	10000.00	257.62	1.76	5689.12	594.83
290.45	-	Alt C	10000.00	259.95	0.39	25419.89	2321.46
290.45	-	Existing	20000.00	262.75	1.76	13908.46	3172.01
290.45	-	Alt C	20000.00	261.22	0.70	28421.88	2518.79
290.45	2 YR	Existing	40000.00	268.86	1.14	35196.52	3674.10
290.45	2 YR	Alt C	40000.00	264.87	1.02	39105.57	3170.99
290.45	5 YR	Existing	50000.00	271.14	1.14	44007.17	3926.98
290.45	5 YR	Alt C	50000.00	267.03	1.08	46106.50	3254.74
290.45	10 YR	Existing	56800.00	272.47	1.15	49241.71	3945.49
290.45	10 YR	Alt C	56800.00	268.18	1.14	49867.96	3261.55
290.45	25 YR	Existing	73000.00	275.28	1.21	60388.65	3982.73
290.45	25 YR	Alt C	73000.00	270.57	1.27	57677.16	3275.64
290.45	50 YR	Existing	90000.00	277.87	1.27	70733.42	4017.11
290.45	50 YR	Alt C	90000.00	273.00	1.37	65663.00	3289.99
290.45	100 YR	Existing	106000.00	279.81	1.35	78652.36	4126.38
290.45	100 YR	Alt C	106000.00	274.64	1.49	71065.68	3299.67

Table 2.6. Hydrologic Information for the area just downstream of the dam							
River Mile	Profile	Plan	Discharge	Water Surface Elevation	Velocity Total	Area	Top Width
			(cfs)	(ft)	(ft/s)	(sq ft)	(ft)
302.08	-	Alt C	1000.00	260.43	0.22	4546.82	458.15
302.08	-	Existing	10000.00	271.43	0.97	29436.15	5112.16
302.08	-	Alt C	10000.00	271.41	0.97	29332.66	5102.81
302.08	-	Existing	20000.00	275.02	1.56	53490.64	9232.33
302.08	-	Alt C	20000.00	274.91	1.57	52479.26	8773.74
302.08	2 YR	Existing	40000.00	279.69	0.70	113204.30	14348.72
302.08	2 YR	Alt C	40000.00	278.87	0.86	101558.70	13996.55
302.08	5 YR	Existing	50000.00	281.29	0.63	136908.00	15164.46
302.08	5 YR	Alt C	50000.00	280.72	0.71	128334.60	14970.76
302.08	10 YR	Existing	56800.00	282.08	0.62	149088.10	15300.98
302.08	10 YR	Alt C	56800.00	281.73	0.66	143659.50	15255.45
302.08	25 YR	Existing	73000.00	283.85	0.62	176337.50	15655.02
302.08	25 YR	Alt C	73000.00	283.22	0.67	166599.30	15505.50
302.08	50 YR	Existing	90000.00	285.61	0.61	204424.70	16313.44
302.08	50 YR	Alt C	90000.00	284.56	0.69	187586.40	15840.08
302.08	100 YR	Existing	106000.00	287.28	0.61	232381.90	17105.22
302.08	100 YR	Alt C	106000.00	285.68	0.72	205577.00	16334.50

Graph 2.3. Velocities in the Action Area. Green shade represents area north of the pool.

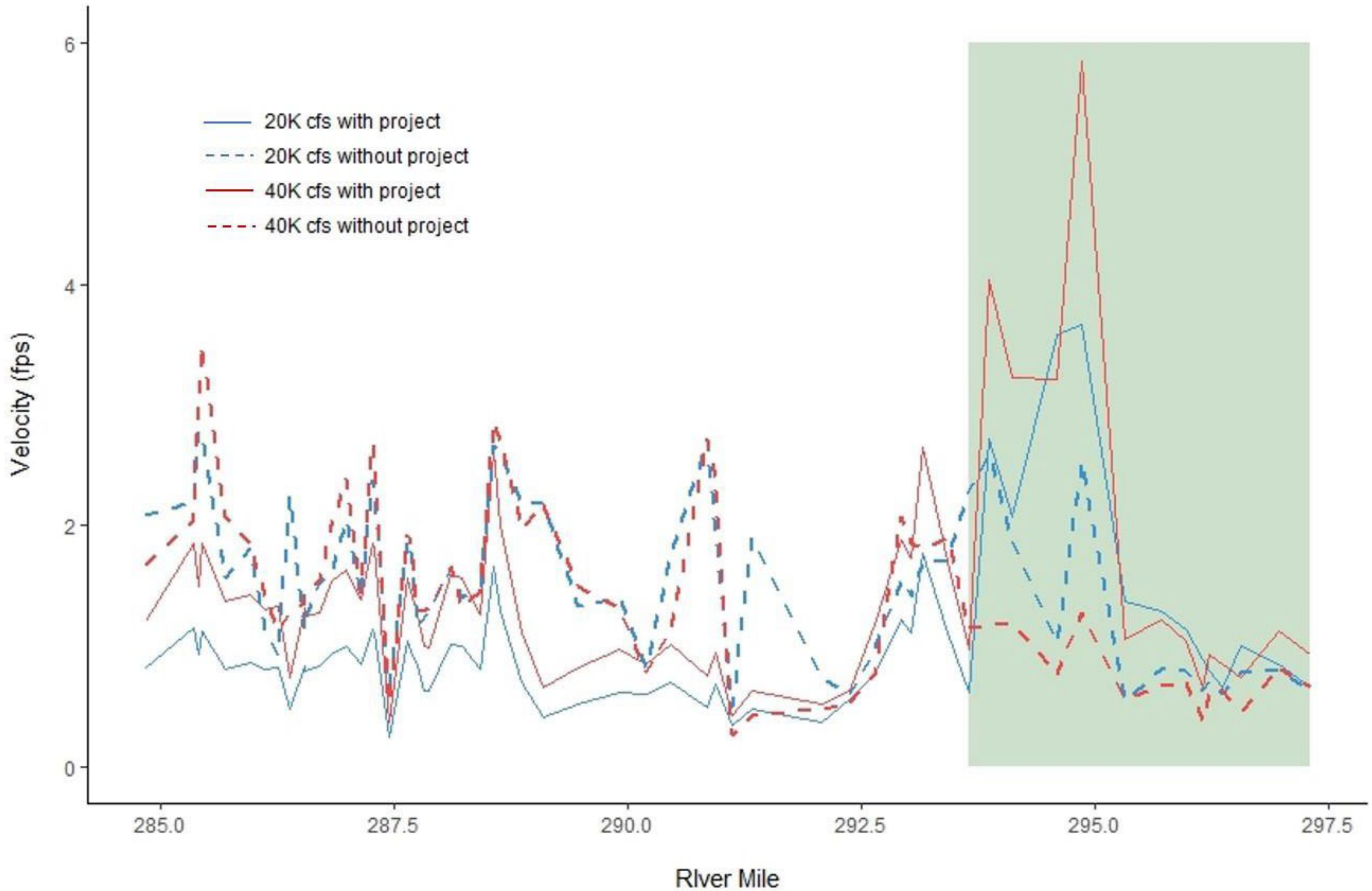


Table 2.7. Modeled Water Quality Parameters pre and post project.

	Depth (ft)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	6.7	4.8	1.3	18.6	22.1	21.9	20.3	24.0

	Flow (cfs)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	3988.0	1320.0	212.8	16268.2	3991.8	1315.4	247.5	16273.4

	Temperature (F)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	68.7	68.7	50.3	87.4	69.0	69.1	49.1	88.2

	Biochemical Oxygen Demand (mg/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	10.7	10.8	10.6	10.8	6.9	7.5	1.8	10.5

	Dissolved Oxygen (mg/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	8.4	8.1	5.8	11.4	8.4	8.2	6.2	10.9

	Total Nitrogen (mg/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	0.94	0.93	0.74	1.10	0.90	0.92	0.68	1.05

	Total Phosphorus (mg/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	0.11	0.11	0.08	0.14	0.11	0.11	0.08	0.14

	Chlorophyll a (ug/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	2.14	1.98	1.93	2.74	1.97	1.21	0.02	9.53

	Total Suspended Solids (mg/L)							
	Existing				Channel Improvements			
	Mean	Median	5%Tile	95%Tile	Mean	Median	5%Tile	95%Tile
Project Area	32.6	28.9	13.4	63.2	31.0	26.8	13.1	62.1

3. CONCURRENCE

The USACE determined that the Action is not likely to adversely affect the wood stork and the Northern long-eared bat. The Service concurs with these determinations, for reasons we explain in this section.

3.1 Wood Stork

The threatened wood stork is a large, long-legged wading bird, about 50 inches tall, with a wingspan of 60–65 inches. The plumage is white except for black primaries and secondaries and a short black tail. The head and neck are largely unfeathered and dark gray in color. Wood storks occur seasonally in Mississippi during the non-breeding season (May–October). Typical foraging sites include freshwater marshes, swales, ponds, hardwood and cypress swamps, narrow tidal creeks or shallow tidal pools, and artificial wetlands (such as stock ponds; shallow, seasonally flooded roadside or agricultural ditches; and impoundments).

Suitable habitat for this species is found within the project area and will be impacted; however, due to the amount of available habitat present in the state of Mississippi, we expect discountable and insignificant effects to the wood stork due to loss of available wetland habitat as a result of project implementation and the very low occurrence of wood storks in the area. In addition, these non-breeding adults are expected to avoid the project area during construction; therefore, the Service concurs with the USACE's determination that the proposed project may affect, but is not likely to adversely affect the wood stork.

3.2 Northern Long-eared Bat

The Northern long-eared bat (NLEB) is a small bat (3.0 to 3.7 inches in length, 9.0 to 10.0 inch wingspan) that is distinguished by its long ears compared to other *Myotis* bats. The bats are found in all or portions of 37 U.S. states, including northeastern Mississippi. A migrating species, the NLEB utilizes forested habitats in the summertime for roosting and rearing their young and hibernate in caves during winter. There is one known hibernaculum cave in Tishomingo County, Mississippi and no known maternity roost trees within the state.

White-nose syndrome, a deadly fungal infection that infects bats within hibernaculum, is the main threat to the NLEB. The fungus has spread to 28 of the 37 states within the bats range and includes locations within Mississippi. Under the NLEB final 4(d) rule, published February 16, 2016 (81 FR 1900 1922), the White-Nose Syndrome Buffer Zone was established to include all areas within 150 miles of the boundaries of U.S. counties or Canadian districts where the fungus has previously been detected. The project area falls within this buffer area but outside of the 0.25-mile (0.4 km) protected buffer zone of the known hibernaculum.

Secondary threats to the NLEB include the disturbance of roosts and hibernation areas, forest management practices, and forest habitat modifications (development, wind power development). The final 4(d) rule states availability of forested habitat does not now, nor will it likely in the future, limit the conservation of the species. The proposed project will not occur near or affect any known maternity roost trees, but will remove potential roosting and foraging habitat and could result in potential adverse effects. Under the final 4(d) rule, any incidental take resulting from forest conversion as a part of the channel excavation and levee realignment action of this project would be considered incidental take resulting from otherwise lawful activities and is not prohibited under the Endangered Species Act. Accordingly, the Service concurs with the USACE's determination that the Action may affect, but is not likely to adversely affect the NLEB.

This concurrence concludes consultation for the listed species and designated critical habitats named in this section, and these are not further addressed in this BO. The circumstances described in the Reinitiation Notice of this BO that require reinitiating consultation for the Action, except for exceeding the amount or extent of incidental take, also apply to these species and critical habitats.

4. GULF STURGEON

4.1. Status of Gulf Sturgeon

This section summarizes best available data about the biology and current condition of the Gulf sturgeon throughout its range that are relevant to formulating an opinion about the Action. The Service published its decision to list Gulf sturgeon as threatened on September 30, 1991 (56 FR 49653 49658). The Gulf Sturgeon Recovery/Management Plan was finalized September 22, 1995. The Service published its final decision designating critical habitat for Gulf sturgeon on March 19, 2003 (68 FR 13370 13495). The most recent 5-year status review of the species was completed September 22, 2009.

4.1.1. Description of Gulf Sturgeon

The Gulf sturgeon, also known as the Gulf of Mexico sturgeon, is an anadromous fish (breeding in freshwater after migrating up rivers from marine and estuarine environments), inhabiting coastal rivers from Louisiana to Florida during the warmer months and overwintering in estuaries, bays, and the Gulf of Mexico. It is a nearly cylindrical primitive fish embedded with bony plates or scutes. The head ends in a hard, extended snout; the mouth is inferior and protrusible and is preceded by four conspicuous barbels. The caudal fin (tail) is heterocercal (upper lobe is longer than the lower lobe). Adults range from 1.2 to 2.4 meters (m) (4 to 8 feet (ft)) in length, with adult females larger than males. The Gulf sturgeon is distinguished from the geographically disjunct Atlantic coast subspecies *(A. o. oxyrinchus)* by its longer head, pectoral fins, and spleen (Vladykov 1955; Wooley 1985). King et al. (2001) have documented substantial divergence between *A. o. oxyrinchus* and *A. o. desotoi* using microsatellite DNA testing.

4.1.2. Life History of Gulf Sturgeon

Like most sturgeons, the Gulf sturgeon is characterized by large size, longevity, delayed maturation, high fecundity, and far-ranging movements. Gulf sturgeon typically live for 20 to 25 years, but can reach ages of at least 42 years old (Huff 1975). Age at sexual maturity ranges from

8 to 12 years for females and 7 to 9 years for males (Huff 1975). High fecundity has been demonstrated by Chapman et al. (1993), who estimated that mature female Gulf sturgeon weighing between 29 and 51 kilograms (kg) (64 and 112 pounds (lb)) produce an average of 400,000 eggs. Long-range migrations from the open Gulf of Mexico to bays and estuaries to coastal rivers are also common. Migratory behavior of the Gulf sturgeon is likely influenced by sex and reproductive status (Fox et al. 2000), change in water temperature (Wooley and Crateau 1985; Chapman and Carr 1995; Foster and Clugston 1997), and increased river flow (Chapman and Carr 1995; Heise et al. 1999a, b; Sulak and Clugston 1999; Ross et al. 2000 and 2001b; Parauka et al. 2001).

In general, all life stages of Gulf sturgeon migrate into rivers in the spring (from late February to May), where sexually mature sturgeon spawn when the river temperatures rises to between 17 to 25 degrees Celsius (ºC) (75 degrees Fahrenheit (°F)). Similar to Atlantic sturgeon, Gulf sturgeon are believed to exhibit a long inter-spawning period, with male Gulf sturgeon capable of annual spawning, but females requiring more than one year between spawning events (Huff 1975; Fox et al. 2000) and only a small percentage of females spawn in a given year (Sulak and Clugston 1999; Pine et al. 2001). Therefore, Gulf sturgeon population viability is highly sensitive to changes in adult female mortality and abundance (Pine et al. 2001; Flowers 2008).

Spawning occurs in the upper reaches of rivers, at least 100 km (62 miles) upstream of the river mouth (Sulak et al. 2004), in habitats consisting of one or more of the following: limestone bluffs and outcroppings, cobble, limestone bedrock covered with gravel and small cobble, gravel, and sand (Marchant and Shutters 1996; Sulak and Clugston 1999; Heise et al. 1999a; Fox et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006). These hard bottom substrates are required for egg adherence and shelter for developing larvae (Sulak and Clugston 1998). Documented spawning depths range from 1.4 to 7.9 m (4.6 to 26 ft) (Fox et al. 2000; Ross et al. 2000; Craft et al. 2001; USFWS unpub. data 2005; Pine et al. 2006).

Gulf sturgeon eggs are demersal (bottom dwelling) and adhesive, and require at least 2 to 4 days to hatch (Parauka et al. 1991; Chapman et al. 1993). After hatching, larval Gulf sturgeon are particularly sensitive to water temperatures above 25°C (77°F) (Chapman and Carr 1995). Young-of-year (YOY) fish disperse widely throughout the river and remain in freshwater for 10 to 12 months after spawning occurs (Sulak and Clugston 1999). They are typically found in open sand-bottom habitat away from the shoreline and vegetated habitat.

Throughout early spring to late autumn, Gulf sturgeon of all ages remain in freshwater until fall (6 to 9 months) (Odenkirk 1989; Foster 1993; Clugston et al. 1995; Fox et al. 2000; Sulak et al. 2009). They typically occupy discrete areas either near the spawning grounds (Wooley and Crateau 1985; Ross et al. 2001b) or downstream areas referred to as summer resting or holding areas. These resting areas are often located in deep holes, and sometimes shallow areas, along straight-aways ranging from 2 to 19 m (6.6 to 62.3 ft) deep (Wooley and Crateau 1985; Morrow et al. 1998; Ross et al. 2001a, b; Craft et al. 2001; Hightower et al. 2002), and frequently near (not in) natural springs (Clugston et al. 1995; Foster and Clugston 1997; Hightower et al. 2002). The substrates consisted of mixtures of limestone and sand (Clugston et al. 1995), sand and gravel (Wooley and Crateau 1985; Morrow et al. 1998), or just sandy substrate (Hightower et al. 2002). With the exception of YOY fish, Gulf sturgeon do not typically feed during freshwater residency (Mason and Clugston 1993; Gu et al. 2001). Sulak et al. (2012) reported that the vast majority (approximately 94 percent) of juvenile, subadult, and adult Gulf sturgeon sampled from the Suwannee River exhibited complete feeding cessation for the 8 to 9-month summer residency; however, a small percentage (approximately 6 percent) of juveniles and subadults did feed in freshwater.

All non-YOY begin to migrate downstream from fresh to saltwater around September (at about 23°C [73 degrees Fahrenheit (°F)]) through November (Huff 1975; Wooley and Crateau 1985; Foster and Clugston 1997), and they spend the cool months in estuarine areas, bays, or in the Gulf of Mexico (Odenkirk 1989; Foster 1993; Clugston et al. 1995; Fox et al. 2002). During the fall migration, Gulf sturgeon may require a period of physiological acclimation to changing salinity levels, referred to as osmoregulation or staging (Wooley and Crateau 1985). This period may be short (Fox et al. 2002) as sturgeon develop an active mechanism for osmoregulation and ionic balance by age 1 (Altinok et al. 1998). Some adult Gulf sturgeon may also spawn in the fall (Randall and Sulak 2012).

Throughout fall and winter, juveniles feed in the lower salinity areas in the river mouth and estuary (Sulak and Clugston 1999; Sulak et al. 2009), while subadults and adults migrate and feed in the estuaries and nearshore Gulf of Mexico habitat (Foster 1993; Foster and Clugston 1997; Edwards et al. 2003, 2007; Parkyn et al. 2007). Some Gulf sturgeon may also forage in the open Gulf of Mexico (Edwards et al. 2003).

The Gulf sturgeon is a benthic (bottom dwelling) suction feeder: it feeds mostly upon small invertebrates in the substrate using its highly protrusible (capable of extension) tubular mouth. The type of invertebrates ingested varies by habitat but are mostly soft-bodied animals that occur in sandy substrates. YOY Gulf sturgeon feed on freshwater aquatic invertebrates, mostly insect larvae and detritus (Mason and Clugston 1993; Sulak and Clugston 1999; Sulak et al. 2009). Juveniles (less than 5 kg (11 lbs), ages 1 to 6 years) forage in lower salinity habitats near the river mouth and in the estuaries, and subadults and adults feed in the estuary and nearshore feeding grounds in the Gulf of Mexico (Foster 1993; Foster and Clugston 1997; Edwards et al. 2003, 2007; Parkyn et al. 2007). Prey in estuarine and marine habitats include amphipods, brachiopods, lancelets, polychaetes, gastropod mollusks, shrimp, isopods, bivalve mollusks, and crustaceans (Huff 1975; Mason and Clugston 1993; Carr et al. 1996; Fox et al. 2000; Fox et al. 2002). Ghost shrimp (*Lepidophthalmus louisianensis*) and haustoriid amphipods (e.g., *Lepidactylus spp.*) are strongly suspected to be important prey for adult Gulf sturgeon over 1 m (3.3 ft) in length (Heard et al. 2000; Fox et al. 2002).

Previous tagging studies indicated that Gulf sturgeon exhibit river fidelity (USFWS and GSMFC 1995). Stabile et al. (1996) identified five regional or river-specific stocks (from west to east): (1) Lake Pontchartrain and Pearl River, (2) Pascagoula River, (3) Escambia/Conecuh and Yellow Rivers, (4) Choctawhatchee River, and (5) Apalachicola, Ochlockonee, and Suwannee Rivers. Dugo et al (2004) reported that genetic structure occurs at the drainage level for the Pearl, Pascagoula, Escambia/Conecuh, Yellow, Choctawhatchee, and Apalachicola rivers (no samples were taken from the Suwannee population). Gulf sturgeon do make inter-river movements (USFWS unpubl. data 2012), and more genetic research is needed to determine if inter-stock movement is resulting in inter-stock reproduction.

4.1.3. Numbers, Reproduction, and Distribution of Gulf Sturgeon

Historically, the Gulf sturgeon occurred from the Mississippi River east to Tampa Bay. Its present range extends from Lake Pontchartrain and the Pearl River system in Louisiana and Mississippi east to the Suwannee River in Florida. Sporadic occurrences have been recorded as far west as the Rio Grande River between Texas and Mexico, and as far east and south as Florida Bay (Wooley and Crateau 1985; Reynolds 1993).

In the late 19th century and early 20th century, the Gulf sturgeon supported an important commercial fishery, providing eggs for caviar, flesh for smoked fish, and swim bladders for isinglass, which is a gelatin used in food products and glues (Huff 1975; Carr 1983). Gulf sturgeon numbers declined due to overfishing throughout most of the 20th century. The decline was exacerbated by habitat loss associated with the construction of dams and sills (low dams), mostly after 1950. In several rivers throughout the species' range, dams and sills have severely restricted sturgeon access to historic migration routes and spawning areas (Wooley and Crateau 1985; McDowall 1988).

On September 30, 1991, the Service and the National Marine Fisheries Service (NMFS) listed the Gulf sturgeon as a threatened species under the Act (56 FR 49653). Threats and potential threats identified in the listing rule included: construction of dams; modifications to habitat associated with dredging, dredged material disposal, de-snagging (removal of trees and their roots) and other navigation maintenance activities; incidental take by commercial fishermen; poor water quality associated with contamination by pesticides, heavy metals, and industrial contaminants; aquaculture and incidental or accidental introductions; and the Gulf sturgeon's long maturation and limited ability to recolonize areas from which it is extirpated.

The Service and NMFS conducted a 5-year status review in 2009 where we concluded that the following threats continue to affect the Gulf sturgeon and its habitat: impacts to habitats by dams, dredging, point and nonpoint discharges, climate change, bycatch, red tide, and collisions with boats (USFWS and NMFS 2009). Additional threats may include ship strikes and potential hybridization due to accidental release of non-native sturgeon. These threats persist to varying degrees in different portions of the species range. The juvenile stage of Gulf sturgeon life history is the least understood, and perhaps the most vulnerable as this cohort remains in the river for the first years of its life and is, therefore, exposed to most of the threats faced by the species and its habitat. Further, the species' long-lived, late-maturing, intermittent spawning characteristics make recovery a slow process.

Currently, seven rivers are known to support reproducing subpopulations of Gulf sturgeon. Table 4.1 lists these rivers and the most-recent estimates of subpopulation size. Abundance numbers indicate a roughly stable or slightly increasing population trend over the last decade in the eastern river systems (Florida), with a much stronger increasing trend in the Suwannee River and a possible decline in the Escambia/Conecuh River. Populations in the western portion of the range (Mississippi and Louisiana) have never been nearly as abundant, and their current status is unknown as comprehensive surveys have not occurred in the past ten years.

At this time, the Service characterizes the status of the species range wide as stable; however, the status of the subpopulations in the Pearl and Pascagoula rivers is uncertain. These rivers do not have current population estimates and have recently been threatened by hurricanes, the Deepwater Horizon oil spill, and a pot-liquor spill in the Pearl River. The Gulf sturgeon continues to meet the definition of a threatened species. While some riverine populations number in the thousands, abundance of most populations is in the hundreds. Loss of a single year class could be catastrophic to some riverine populations with low abundance. Further, while directed fisheries no longer occur, many threats continue and new ones are arising. Data are not yet available to determine if Gulf sturgeon recovery is limited by factors affecting recruitment (e.g., spawning habitat quantity or quality), adult survival (e.g., incidental catch in fisheries directed at other species), or the late-maturing, intermittent reproductive characteristics of the species.

4.1.4. Conservation Needs of and Threats to Gulf Sturgeon

At the time of the listing of the Gulf sturgeon, several threats were discussed as the reason for the decline of the species. These threats and potential threats include: modifications to habitat associated with dredging, dredged material disposal, de-snagging (removal of trees and their roots) and other navigation maintenance activities; incidental take by commercial fishermen; poor water quality associated with contamination by pesticides, heavy metals, and industrial contaminants; aquaculture and incidental or accidental introductions; and the Gulf sturgeon's slow growth and late maturation (56 FR 49653).

Dams restrict the gulf sturgeon's ability to use upstream areas past the dams for spawning because they are unable to pass through these dam systems (56 FR 49653). The Ross Barnett dam on the Pearl River and the Jim Woodruff Lock and Dam on the Apalachicola are two such dams that block the upstream migration of the species. While smaller dams such as the Poole's Bluff and Bogue Chitto Sills are passable at certain flow conditions, those small structures can still impede upstream migration. Not only do dams restrict upstream migration, they can cause altered flow, channel morphology changes, and water quality issues well downstream of their construction (USFWS 2009). Dredging activities have also led to habitat degradation for the Gulf sturgeon by modification of important channel features used for spawning and foraging. Dredging can also be detrimental to gulf sturgeon due to direct mortality from entrainment.

Although direct take of Gulf sturgeon is prohibited within the states in the current species range, risk from incidental bycatch due to entanglement in fishing and trawling gear still occurs (USFWS 2009). Shrimpers have continued to document Gulf sturgeon bycatch in shrimp trawls even with the inclusion of sea turtle and fish excluder devises on the trawls. The State of Florida has made it unlawful to use entangling nets (i.e., gill and trammel nets) in state waters and has also restricted the use of other types of nets (i.e., cast nets, seines, etc.). The implementation of these net bans has likely been a benefit to recovery for Gulf sturgeon; however, sturgeon continue to be caught in these nets in states without these types of bans or restrictions.

The threat of poor water quality, while not clearly understood, has been studied and some studies indicate the potential impacts to various life stages of Gulf sturgeon. A study in the Suwannee River by Sulak et al. (2004) indicated that for Gulf sturgeon to have successful egg fertilization a narrow range of pH and calcium ion concentration may be required. It has also been shown that egg and larval development can be vulnerable to various forms of pollution, temperature, and dissolved oxygen (DO) levels. The Bogalusa paper mill spill on the Pearl River in 2011 contributed to a large fish kill, resulting in the death of approximately 28 Gulf sturgeon most of which were juveniles (Slack et al. 2014).

Hurricanes and collisions with boats are also ongoing threats to the species. Hurricanes such as Ivan in 2004, Katrina in 2005, and most recently Michael in 2018 have shown to cause mortality of Gulf sturgeon. While the impacts of the population in the Pearl River from Hurricane Katrina are generally unknown, reports from the first few days after the storm counted at least eight dead Gulf sturgeon (Mike Beiser, MSDEQ, personal communication). After Hurricane Michael, dozens of dead Gulf sturgeon were documented by Florida Fish and Wildlife Conservation Commission (FWC) biologists (Kaeser 2019). Gulf sturgeon have been seen jumping out of the water, possibly as a form of group communication to maintain group cohesion (Sulak et al. 2002). Collisions with boats has been attributed to this jumping behavior resulting in mortality to the species, as well as posing a safety issue to boaters (USFWS 2009).

The most recent 5-year review (2009) confirmed that these threats continue to be ongoing for the Gulf sturgeon.

4.1.5. Tables and Figures for Status of Gulf Sturgeon

Table 4.1. Estimated size of known reproducing subpopulations of Gulf sturgeon. In some cases, multiple estimates are presented based on differences in population estimation models used. All estimates apply to a proportion of the population exceeding a minimum size, which varies by researchers according to the sampling method used. CI = confidence interval. NR =not reported.

River	Year of data collection	Abundance Estimate	Lower Bound 95% CI	Upper Bound 95% CI	Source
Pearl	2001	430	323	605	Rogillio et al. 2001
Pascagoula	2000	181	38	323	Ross et al. 2001
Pascagoula	2000	206	120	403	Ross et al. 2001
Pascagoula	2000	216	124	429	Ross et al. 2001
Escambia/Conecuh	2006	451	338	656	USFWS 2007
Yellow	2011	1,036	724	1,348	USFWS 2012 unpub. data
Choctawhatchee	2008	3,314	NR	NR	USFWS 2009
Apalachicola	2005	2,000	NR	NR	Pine and Martell 2009a
Apalachicola	2010	1,292	616	1,968	USFWS 2010 unpub. data
Suwannee	2004	10,000	NR	NR	Pine and Martell 2009a
Suwannee	2006	9,728	6,487	14,664	Randall 2008
Suwannee	2007	14,000	NR	NR	Sulak 2008

4.2. Environmental Baseline for Gulf Sturgeon

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the Gulf sturgeon, its habitat, and ecosystem within the Action Area. The environmental baseline refers to the condition of the listed species or its designated critical habitat in the Action Area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the Action Area, the anticipated impacts of all proposed Federal projects in the Action Area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR §402.02).

4.2.1. Action Area Numbers, Reproduction, and Distribution of Gulf Sturgeon

Recent studies for the Gulf sturgeon have not been conducted in this reach of the Pearl River and survey data from this area is not prevalent; however, there are unconfirmed sightings of Gulf sturgeon as far upstream as the City of Jackson, Mississippi, in Hinds County which is within the Action Area (Morrow et. al. 1996; Lorio 2000; Slack, pers. comm. 2002). Just north of the Action Area at RM 301.77 is the Ross Barnett Reservoir, which presents a total barrier to migration to Gulf sturgeon (56 FR 49653). Prior to the construction of the Ross Barnett, there are records of Gulf sturgeon found in the vicinity of the dam and reservoir site as well as further upstream along the Pearl River; however, since its completion no sturgeon have been captured upstream of the dam (Morrow et al 1998; Sulak et al 2016). In 1915, the City of Jackson built a low weir at approximately RM 290.7 to provide the water supply for the city, which continues to provide a large portion of the city's water supply. This weir potentially restricts the Gulf sturgeon's access to the Action Area in low flow periods.

To address flooding in the Jackson area, the 1960 Flood Control Act authorized construction of the Jackson (i.e., Fairgrounds) and East Jackson levees which was completed in 1968. An extension of the Jackson levee was completed in 1984. This flood control project consists of those two earthen levees on either side of the river totaling 13.2 miles. There is also channel work associated with the levees which includes 9.3 miles of enlargement and realignment of the main river channel through the town of Jackson (approximately 5 miles of cutoffs). Maintenance includes any necessary periodic removal of vegetation along a 650-foot-wide cleared strip of floodplain along the river and complete clearing downstream of that; a total of 346 acres of the floodplain are maintained in some form of cleared or partially cleared floodplain. About 80 percent of the Action Area has been affected by these past flood control activities. Due to these activities, the river in this area has relatively shallow base flows except for a short time after rain events where the river will have a fairly fast, deep flow.

Most of the Gulf sturgeon surveys in the Pearl River Basin have been focused in the Lower Pearl River and the Bogue Chitto River. The Poole's Bluff and Bogue Chitto Sills in the lower part of the river system have limited the Gulf sturgeon's migration access to the reaches of the river north of these sills during low water periods; however, surveys have shown that Gulf sturgeon can and do swim past the both sills during high water periods (USFWS BRFWCO 2018). A study conducted by the Baton Rouge Fish and Wildlife Conservation Office (BRFWCO) from 2013 to 2016 assessed the Gulf sturgeon's ability to move upstream of the two sills. It was found that of the sturgeon that made the attempt to pass over the sills, 72 percent successfully passed the Poole's Bluff sill and 21 percent successfully made it over the Bogue Chitto Sill. It is uncertain if these sturgeons are navigating over or around the structures (Kohl 2003).

Scientific surveys conducted over the past three decades have collected early juveniles which demonstrates that the Pearl River still supports a spawning population, although the exact spawning locations are yet to be discovered (Sulak et al. 2016). Results from sturgeon captures in the Pearl River between 1992 and 2001 suggest a stable subpopulation of 430 fish, with approximately 300 hundred adults (Rogillio et al. 2001). With the presence of juvenile sturgeon captured during survey efforts, it leads to the indication that successful spawning takes place at some location in the Pearl River (USFWS 2003). Survey activities have primarily been focused on the lower Pearl River and the Bogue Chitto River; therefore, the data for gulf sturgeon in this reach of the river, approximately 19.37 miles, in the Action Area are minimal and typically consist of sporadic captures by commercial fishermen (Table 4.2). Although these records are not from scientific surveys, the commercial data indicate that sturgeon are migrating north of the Poole's Bluff Sill, into the upper reaches of the Pearl River, approximately every 3.4 years, due to water levels at the sill occurring at passible levels for the sturgeon. As scientific surveys have not been conducted in this reach of the river, the sporadic captures from Table 4.2 do not give a good indication of sturgeon density in this reach of the river; the density is at this time not fully known. As shown in Table 4.2, there have been 24 Gulf sturgeon captured by commercial fishermen, eight of which being captured within the Action Area and the most recent of those captures occurring, a juvenile, in 2008. Adult sturgeon have also been captured by commercial fisherman downstream of the Action Area near the Strong River's confluence with the Pearl River during spawning season leading to the possibility that adults that make it past the Poole's Bluff Sill come at least as far as the Strong River to spawn (BRFWC 2019 pers. comm.). The Service suspects that the only true suitable spawning habitat is found north of both sills on the Pearl River and the Bogue Chitto River, but the habitat is only accessible during high flow periods (USFWS 2003). That same area of the river, north of the sill, is also thought to have the gravel substrate necessary for spawning in the Pearl River.

4.2.2. Action Area Conservation Needs of and Threats to Gulf Sturgeon

The Action Area consists of approximately 231 acres of riverine habitat that would be impacted by the project. This impacted area of the river consists of habitat that Gulf sturgeon could use to either spawn as adults or feed as juveniles if they migrate to that reach of the river; however, suitable spawning habitat is believed to be further south of the Action Area. The status of the Gulf sturgeon in the Action Area has been influenced by past channelization and a 200–foot-wide weir structure that supplies the City of Jackson with water. The past channelization and levee construction isolate 5.34 miles of Pearl River meanders. These previous actions have reduced the amount of river habitat available for the species, including reductions in foraging and spawning areas. The degraded water quality from pollution and storm water runoff into the

Pearl River in the Action Area could also have impacted the Gulf sturgeon populations in this area. The section of river north of the water supply weir is accessible in high water events; however, in times of low water the species are not able to migrate past the structure. Downstream of the 200–foot-wide weir is a ring levee around the Savannah Street Wastewater Treatment Plant. The levee surrounding this plant has, during high water events, overtopped and stormwater has spilled into the river; however,but the plant's containment system would be upgraded during the Action to prevent this from occurring in the future. In the area just north of the Action Area, the Ross Barnett Reservoir has resulted in an obstruction to Gulf sturgeon migration further up the Pearl River. Suitable spawning habitat for this species is thought to be south of the Action Area on the Pearl River near its confluence with the Strong River due to the substrate of the river bottom at that location.

In May of 2019, the MDEQ issued a water quality advisory for the Pearl River in Jackson due to ongoing sanitary sewer overflows around the City of Jackson discharging wastewater into various waterbodies that flow into the river (MDEQ 2019). There are two former landfills (Gallatin Street and Jefferson Street) and the former Gulf States Creosote Plant that are located within the proposed project area. The 62-acre Gallatin Street landfill contains urban and industrial trash. Leachates from within the site contain cadmium, lead, and nickel above the regulatory standards. Debris from this landfill is reported to be washing into the river. The 45-acre Jefferson Street (or Lafleurs Landing) landfill also has debris that can be eroded during high river stages. The EPA's PSA/SI done in 2003 found barium, cobalt, managanese, and zinc, as well as creosote residuals consisting of a variety of polycyclic aromatic hydrocarbons (PAHs).

According to the 2009 5-year review, the population of Gulf sturgeon in the Pearl River is stable (USFWS 2009). Most of the survey data comes from the lower reach of the river, but studies show that there is recruitment in the river. However, this information does not provide population or capture data for the Action Area. The evidence of presence for the species in the Action Area is based on commercial data from fishermen capturing the species in hoop nets as discussed in Section 4.2.1. Scientific surveys would be needed to accurately quantify population numbers in the Action Area.

4.2.3. Tables and Figures for Environmental Baseline for Gulf Sturgeon

Table 4.2. Historic Gulf sturgeon captures north of the Poole's Bluff Sill. Note: Records in *Italics* are in the Action Area, while records from below the Action Area are not italicized.

Year	Location	Age Class	Capture Method	Captured By	Source
1939	*Upper Pearl River, above Pools Bluff Sill, Pearl River, north of Jackson MS*	*Adult*	*Unknown*	*Unknown*	*Morrow et al. 1996*
1940	*Pearl River N. Jackson MS*	*Unkown*	*Unknown*	*Unknown*	*Lafayette Printed Database*

1940	Upper Pearl River, above Pools Bluff Sill, Rockport, MS	Unknown	Unknown	Unknown	Morrow et al. 1996
1942	*Pearl River N. Jackson MS*	*Unknown*	*Unknown*	*Unknown*	*Lafayette Printed Database*
1942	*Upper Pearl River, above Pools Bluff Sill, Pearl River, north of Jackson MS*	*Unknown*	*Unknown*	*Unknown*	*Morrow et al. 1996*
1953	Strong River near Caney Creek	Adult	Unknown	Fisherman	www.fffmag.com; August 2001
1971	*Upper Pearl River, above Pools Bluff Sill, Below Spillway of Ross Barnett Res.*	*Adult*	*Unknown*	*Unknown*	*Morrow et al. 1996*
1976	*Pearl River-below Ross Barnett Reservoir spillway*	*Adult*	*Unknown*	*Commercial fisherman*	*Gulf Sturgeon Recovery Plan 1995*
1979	*Pearl River below spillway of Ross Barnet Res.*	*Unknown*	*Unknown*	*Unknown*	*Lafayette Printed Database*
1982	Pearl River Monticello MS	Unknown	Unknown	Unknown	Lafayette Printed Database
1982	Upper Pearl River, above Pools Bluff Sill, Pearl River at Monticello MS	Adult	Unknown	Unknown	Morrow et al. 1996

1982	Upper Pearl River, above Pools Bluff Sill, Pearl River at Monticello MS	Juvenile	Unknown	Unknown	Morrow et al. 1996
1982	Upper Pearl River, above Pools Bluff Sill, Pearl River at Monticello MS	juvenile	unknown	unknown	Morrow et al. 1996
1983	Pearl River Monticello MS	Unknown	Unknown	Unknown	Lafayette Printed Database
1984	Upper Pearl River, above Pools Bluff Sill, Pearl River at Byram, MS	Unknown	Unknown	Unknown	Morrow et al. 1996
1984	Upper Pearl River, above Pools Bluff Sill, Pearl River at Byram, MS	Unknown	Unknown	Unknown	Lafayette Printed Database
1985	Pearl River between Wanilla and Rockport	Unknown	Hoop net	Unknown	Slack pers. Comm.
1993	Upper Pearl River, above Pools Bluff Sill, Strong River, MS	Adult	Unknown	Unknown	Morrow et al. 1996
1996	Pearl River south of Georgetown, MS	Adult	Hoop net	Fisherman	Knight 1996
2000	Pearl River near Georgetown, MS	Unknown	Hoop net	Unknown	Slack pers. Comm.

2002	Red Bluff Creek, North of Morgantown, MS	Unknown	Hoop net	Unknown	Slack pers. Comm.
2008	*Below Ross Barnett Reseroir*	*Juvenile*	*Rod and reel*	*Fisherman*	*Slack pers. Comm.*
2018	Pearl River between Wanilla and Rockport	2 Adults	Hoop net	Commercial fisherman	Mann pers. Comm.

4.3. Effects of the Action on Gulf Sturgeon

This section analyzes the effects of the Action on the Gulf Sturgeon. Effects of the Action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the Action may occur later in time and may include consequences occurring outside the immediate area involved in the Action (50 CFR §402.02).

Our analyses are organized according to the description of the Action and the defined Action Area in section 2 of this BO.

4.3.1. Effects of Channel Excavation and Levee Relocation on Gulf Sturgeon

Approximately 207.7 acres of open water would be impacted by the channel excavation and levee relocation. Channel excavation and levee relocation would occur on the outer banks with approximately 100 feet of buffer area between the bank excavation and the river to retain some bank stability. The excavation is projected to occur during the low water periods of the year; however, while the excavation and levee relocation construction activities are being conducted, the disturbance to the sediment would increase the turbidity in the river. During the construction period and until a vegetative cover is established on the levees, the levees and all disturbed areas would be subject to erosion. This eroded material would be carried into small tributary streams and into the Pearl River system. The turbidity would be additive to any downstream riverbank erosion resulting from sediments being trapped behind the weir after its construction. Increased sediment and turbidity can result in decreased light penetration and decreased photosynthesis. High levels of sediment can settle on fish spawning areas and smother fish eggs and larvae. Production of benthic organisms also can be reduced by high levels of sediment. Further, sediments can settle on respiratory surfaces of fish and aquatic organisms and interfere with respiration.

Sulak et al (2016) found it difficult to quantify indirect mortality impacts from natal river habitat alterations including channelization, dredging, impoundments, and bulk heading. This uncorrected and/or uncontrolled alteration of Gulf sturgeon habitat could limit the success of a promising year-class at various stages in the Gulf sturgeon life cycle. Kynard and Parker (2004) found that while juveniles are mostly bottom feeders, they also spent an unusual amount of time in a holding pattern in the water column suggesting that when benthic foraging in the river is scarce, juvenile fish have evolved to drift feed. If this assumption is correct, should the water quality affect the benthic macroinvertebrates in the Action Area, the foraging juveniles could move up in the water column to drift feed. Areas that have high concentrations of suspended sediments show a decrease in macroinvertebrate diversity, especially the more sensitive species (Sawyer et al 2004). Studies from other regions indicate that sedimentation decreases available spawning habitat, increases egg and larvae mortality, and can decrease feeding success of species that rely on visual search strategies (Sawyer et al. 2004; Berman and Rabeni 1987; Henley et al. 2000). The increased sedimentation and turbidity in the river from the channel excavation and levee relocation would have impacts on the macroinvertebrate prey for any juvenile Gulf sturgeon that would be temporarily feeding in the Action Area.

4.3.2. Effects of Weir Construction and Impoundment on Gulf Sturgeon

With the construction of the 1,500-foot-wide weir structure and resulting impoundment from the weir, changes to the velocity and water surface elevation would occur within the Action Area. The weir has been designed to match the current discharge of the river; therefore, there should not be significant change in discharge after the target area has filled to the top of the weir. Low-head dams, such as weirs, impede migratory pathways of fish including the Gulf sturgeon. As a way for fish to move past the weir, a fish passage channel would be constructed east of the weir and low flow structure.

During the construction of the weir, low flow structure, and fish passage channel, there would be similar impacts to Gulf sturgeon as those associated with the channel excavation and levee relocation. These impacts include increased sedimentation, increased turbidity, and bank destabilization. Excavation of the area for the weir site, low flow structure, and fish passage channel is necessary, but would potentially cause excess sediment to flow downstream approximately 1.6 miles south of the construction area and erosion could be exacerbated in that area until the riverbank has stabilized. See Section 4.3.1 for more information regarding the effects of sedimentation and turbidity.

Dams or impoundments are thought to be one of the main obstacles to Gulf sturgeon recovery in the Pearl River (Sulak et al. 2016). Low-head barrier dams such as the proposed weir structure have consistent influences on stream-fish assemblages which have shown longitudinal declines in species richness from below to above barriers and result in altered population dynamics of a species (Porto et al. 1999; Pringle 1997). Impoundments confine spawning and YOY nursery/feeding habitat to the unimpounded reaches of the river below the dams. In the lower Pearl River, the Poole's Bluff Sill blocks access to the river north of the structure during low river stages; however, when the water is high Gulf sturgeon are able to move past the structure (BRWFCO 2018). The migratory blockage caused by the weir structure could impact the sturgeon's ability to swim north of the structure unless there are high water events; however, a fish passage channel has been included as part of the project design to minimize the impacts on aquatic species migration.

Impoundments/dams generally have adverse impacts to riverine fish communities by interrupting migratory movements. With the addition of the fish passage channel in the design of the project, impacts may be minimized to Gulf sturgeon migration providing that flow conditions would meet the needs of the species to be able to navigate the passage. These conditions include water velocity that does not exceed the sturgeon's swim speed and enough water flow levels for the species to be able to swim through it. Studies have shown that Gulf sturgeon cannot swim against currents greater than 1 to 2 meters per second (mps) (3 to 6 fps); however, they can swim up to 2 to 2.5 body lengths per second (Boyd Kynard, pers.comm. 2003; Kohl 2003; Wakefield 2001). Studies on fish passage attraction speed flow has shown that the recommended flow should be between 2 and 4 fps with sustained swim speed ranges for sturgeon to be in the range of 3 to 4 fps (Cheong et al. 2006; White and Mefford 2002). The swimming capabilities of juvenile sturgeon has been tested, and it was documented that juveniles less than 6 inches long can swim at velocities up to 1 mps (3 fps) (Kohl 2003). At this time, there is only a conceptual model of the fish passage channel, approximately 1.4 miles long of a curving channel, with the possible velocities ranging anywhere from 1 to 7 fps. The maximum velocity of 7 fps would push the limits of adult Gulf sturgeon's ability to swim against the current and any juveniles attempting to migrate through the passage would be unable to swim through it. However, depending on where in the channel the velocities would occur at that speed the fish may be able to migrate successfully through it; therefore, the proposed passage feature should be monitored for water velocity and water level conditions.

Water velocity is an important factor in the life-cycle of Gulf sturgeon. During spawning, flows that are too strong would prevent eggs from settling on and adhering to suitable substrate, while flows that are too low could cause clumping of the eggs and lead to increased mortality from fungal infection and asphyxiation (Wooley and Crateau 1985; USFWS 2003). A study performed by Flowers et al. (2009) on the Apalachicola River presented evidence that flow regime and water velocity influence Gulf sturgeon spawning by stimulating the adults to move to spawning grounds. These flow regimes and water velocities also structure and modify substrate to create suitable areas for egg attachment and provide adequate oxygenation for egg survival (Auer 1996; Fox et al. 2000). Optimal spawning site flows generally are high and have a continuous rate of current flow, approximately > 4.9 fps. There are no spawning sites documented in the Action Area, and although recruitment occurs in the Pearl River, the actual site(s) where spawning occurrs is unknown at this time.

According to the 1995 Gulf Sturgeon Recovery Plan, Wooley and Crateau (1985) reported that sturgeon in the Apalachicola River around the Jim Woodruff Lock and Dam have been found in depths of 19.7 to 39.4 ft and at velocities ranging from 2 to 3 fps during the summer months. The low flow structure is designed to meet the same required discharge as the Ross Barnett dam, which would allow for no change in the current discharge due to the project. Table 2.1 shows the average monthly discharges from 1966 to 2013, indicating that June through October typically have the lowest discharges and December through April typically have the highest discharges. While the velocities in the Action Area will be modified due to the project, discharges are anticipated to remain the same because the project design matches the flow of the

Ross Barnett dam. Because the current flows would remain the same according to project design and sturgeon have been captured by commercial fishermen in the Action Area previously at the similar flow regime, impacts to sturgeon in the area would be minimal.

An examination of low-head dams determined that the major issue resulting from such structures is alterations in water temperature caused by anthropogenic influences impacting water quality within the created water body (Cummings 2004). In water, temperature influences other water quality factors such as DO and pH in the water column. In freshwater, when the temperature increases the pH decreases (Kishinhi et al. 2006). Kishinhi et al. (2006) studied the water quality in the Pearl River around Jackson, Mississippi, and in the Ross Barnett Reservoir and found that the mean pH values at all of the sampling sites were within the State's recommended range of 6.0 to 9.0. The optimal pH for sturgeon eggs generally lies in the range of nearly neutral (pH=7.0) to slightly alkaline (pH<8.0) (Sulak et al. 2016). Although water quality parameters were modeled for pre- and post-project, it did not include a model specifically for pH. However, temperature was modeled, and although there were slight differences, none were significantly different from current ranges (Table 2.7). Therefore, we can infer that the pH would not have significant changes post project.

Dissolved oxygen levels are also important water quality aspects for feeding and the survival of juvenile sturgeon. Secor and Gunderson (1997) studied the effects of temperature and DO on juvenile Atlantic sturgeon (*Acipenser oxyrinchus*). According to this sturdy, reduced oxygen levels resulted in a threefold reduction in growth rate and a reduction in routine respiration rates. Juveniles were more vulnerable to low DO levels and high temperatures; however, in spite of reduced respiration and survival, they continued to feed and grow through reduced activity to allocate more energy to growth. Although specific DO tolerance levels have not been established for the Gulf sturgeon, hypoxia for other *Acipenser* species have been documented to start at 4 milligrams per liter (mg/L) (Cech et al. 1984; Jenkins et al. 1993; Kahn and Mohead 2010; Secor and Gunderson 1998). The DO levels for the Pearl River and Ross Barnett Reservoir were monitored by Kishinhi et al. (2006); the lowest levels of DO occurred in August at 5 mg/L and the highest levels occurred in December at 20 mg/L. The mean concentrations of DO for that study were normally above the minimum of 5 mg/L recommended by the MDEQ for the protection of aquatic life. As with the pre- and post-project water quality modeling for temperature, DO was modeled with slight but not significant differences from pre–project conditions. High temperatures and lower DO levels would be likely to occur during the summer months when juvenile Gulf sturgeon would use the area for feeding, but DO levels for this area are projected to be minimally changed post-project and the levels should not drop below the tolerance for juveniles (Table 2.7). Should the DO levels drop below 4 mg/L, the juveniles would be stressed from the lower levels, however, it is possible they would continue to feed but reduce activity to have the ability to continue to grow.

4.3.3. Effects of Non-Federal Activities caused by the Federal Action on Gulf Sturgeon

With the improved channel, recreational water sports (e.g., fishing and boating) will be expected to increase as a result of the improved access to the Action Area. This increase in fishing could lead to more incidental captures of Gulf sturgeon in hoop nets or with rods and reels if sturgeon migrate past the weir through the fish passage channel. Because the Service does not know the degree to which recreational uses will increase, we are unable to estimate the number of sturgeon that could be impacted due to the increase in fishing.

An increase in development adjacent to the improved channel could also lead to a decrease in water quality which would, in turn, impact prey sources and juvenile growth. Although changes in water quality could be measured, estimating the amount or extent of those changes to prey resources and juvenile growth are difficult to predict or anticipate. The relocation or retrofitting of existing infrastructure within the Action Area (i.e., roads, bridges, pipelines, transmission lines) would also lead to a decrease in water quality during construction of such actions, although such impacts would be temporary. These impacts to water quality would include increased sedimentation and turbidity from any excavation in or around the river. This increased sedimentation and turbidity would have the same impacts to sturgeon as discussed in Section 4.3.1.

4.3.4 Summary of Effects of the Action on Gulf Sturgeon

The Poole's Bluff Sill hinders sturgeon migration upriver and sturgeon can only pass that sill during high water events. Based on best available data and the probability of high water events coinciding with northward riverine migration, we estimate that only 0.9 percent of the Pearl River population can access river habitat north of the sill on any given year. Individually, the separate activities of the Action will likely not harm sturgeon in the Action Area; however, collectively the compounding effects of all the activities are likely to rise to the level of harm. Thus, we estimate that the collective activities of the entire Action would disturb a maximum of 4 sturgeon per year to the level of harm. Given that construction would last up to 5 years, the maximum number of sturgeon affected by the Action would be 20 fish (4.6 percent of the Pearl River population).

4.4. Cumulative Effects on Gulf Sturgeon

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. At this time the Service is unaware of any future state, tribal, local, or private non-Federal unrelated to the proposed action that are reasonably certain to occur in the Action Area. Therefore, cumulative effects are not relevant to formulating our opinion for the Action.

4.5. Conclusion for Gulf Sturgeon

In this section, we summarize and interpret the findings of the previous sections for the Gulf sturgeon (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"*Jeopardize the continued existence*" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

The Action would alter 9.5 river miles of Gulf sturgeon habitat in the Pearl River. Increased sedimentation and turbidity from the construction of the weir and fish passage channel, as well as erosion during the excavation phase of the approximately 5-year project would decrease the macroinvertebrates in the area. This decrease in food sources could lead any juveniles in the area to possibly leave in search of sustenance. The increased turbidity and sedimentation caused by all of the construction actions including the retrofitting or relocation of existing infrastructure would be temporary; therefore, as Gulf sturgeon are highly mobile and can avoid these areas, any effects on their overall health would be minimal. After construction has been completed, it is probable that sturgeon could return to the area as long as it is a year when water flow is high enough to migrate past the Poole's Bluff Sill that occurs downstream.

The anticipated changes in DO from the impoundment would impact any juveniles foraging in the area as well as their prey base. The reduction in water quality from lower DO levels would impact any foraging sturgeon in the area, but they are known to reduce activity to conserve energy to feed and grow in periods of low DO.

The weir structure will possibly cause migration issues for the sturgeon; however, a fish passage feature has been designed for just downstream of the weir. The construction of the fish passage channel would increase the possibility of sturgeon having the ability to return to the area should they migrate into that reach of the river.

The various stressors and forms of disturbance from the Action, considered separately, are not likely to cause harm of sturgeon found in the Action Area. However, considered collectively, the combined level of stressors and disturbances could result in harm to a maximum of 20 sturgeon (4.6 percent of the Pearl River population) utilizing the Action Area. The status of the subpopulation of Gulf sturgeon in the Pearl River has been shown to be stable. Our analysis indicates that while the Action would have a negative effect on those 20 sturgeon, such effects to 4.6 percent of that subpopulation would not be appreciable for the survival and recovery of the Gulf sturgeon.

After reviewing the current status of the species, the environmental baseline for the Action Area, the effects of the Action and the cumulative effects, it is the Service's biological opinion that the Action is not likely to jeopardize the continued existence of the **GULF STURGEON**.

5. CRITICAL HABITAT FOR GULF STURGEON

5.1. Status of Gulf Sturgeon Critical Habitat

This section summarizes best available data about the current condition of all designated units of critical habitat for Gulf sturgeon that are relevant to formulating an opinion about the Action. The Service published its decision to designate critical habitat for Gulf sturgeon on March 19, 2003 (68 FR 13370 13495). The most recent 5-year status review of the species was completed September 22, 2009.

5.1.1. Description of Gulf Sturgeon Critical Habitat

The Service and NOAA Fisheries jointly designated Gulf sturgeon critical habitat on April 18, 2003 (68 FR 13370, March 19, 2003). Gulf sturgeon critical habitat includes areas within the major river systems that support the seven currently reproducing subpopulations and associated estuarine and marine habitats. Gulf sturgeon use rivers for spawning, larval and juvenile feeding, adult resting and staging, and moving between the areas that support these life history components. Gulf sturgeon use the lower riverine, estuarine, and marine environment during winter months primarily for feeding and, more rarely, for inter-river movements.

Fourteen areas (units) are designated as Gulf sturgeon critical habitat (Figure 5.1). Critical habitat units encompass approximately 2,783 km (1,729 mi) of riverine habitats and 6,042 square km (km^2) (2,333 square miles) of estuarine and marine habitats, and include portions of the following Gulf of Mexico rivers, tributaries, estuarine and marine areas:

Unit 1	Pearl and Bogue Chitto Rivers in Louisiana and Mississippi;
Unit 2	Pascagoula, Leaf, Bowie, Big Black Creek and Chickasawhay Rivers in Mississippi;
Unit 3	Escambia, Conecuh, and Sepulga Rivers in Alabama and Florida;
Unit 4	Yellow, Blackwater, and Shoal Rivers in Alabama and Florida;
Unit 5	Choctawhatchee and Pea Rivers in Florida and Alabama;
Unit 6	Apalachicola and Brothers Rivers in Florida;
Unit 7	Suwannee and Withlacoochee River in Florida;
Unit 8	Lake Pontchartrain (east of causeway), Lake Catherine, Little Lake, the Rigolets, Lake Borgne, Pascagoula Bay and Mississippi Sound systems in Louisiana and Mississippi, and sections of the state waters within the Gulf of Mexico;
Unit 9	Pensacola Bay system in Florida;
Unit 10	Santa Rosa Sound in Florida;
Unit 11	Nearshore Gulf of Mexico in Florida;
Unit 12	Choctawhatchee Bay system in Florida;
Unit 13	Apalachicola Bay system in Florida; and
Unit 14	Suwannee Sound in Florida.

Critical habitat designation for the Gulf sturgeon used the term "primary constituent elements" (PCEs) to identify the key components of critical habitat that are essential to its conservation and may require special management considerations or protection. Revisions to the critical habitat regulations in 2016 (81 FR 7214, 50 CFR §4.24) discontinue use of the term PCEs, and rely exclusively the term “physical and biological features” (PBFs) to refer to these key components, because the latter term is the one used in the statute. This shift in terminology does not change how the Service conducts a “destruction or adverse modification” analysis. In this BO, we use the term PBFs to label the key components of critical habitat that provide for the conservation of the Gulf sturgeon that were identified in its critical habitat designation rule as PCEs. The PBFs of Gulf sturgeon critical habitat are (68 FR 13370 13495):

- Abundant food items, such as detritus, aquatic insects, worms, and/or mollusks, within riverine habitats for larval and juvenile life stages; and abundant prey items, such as amphipods, lancelets, polychaetes, gastropods, ghost shrimp, isopods, mollusks and/or crustaceans, within estuarine and marine habitats and substrates for subadult and adult life stages;
- Riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone, or hard clay;
- Riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during freshwater residency and possibly for osmoregulatory functions;
- A flow regime (*i.e.*, the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) necessary for normal behavior, growth, and survival of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging, and for maintaining spawning sites in suitable condition for egg attachment, egg sheltering, resting, and larval staging;
- Water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages;
- Sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
- Safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g., an unobstructed river or a dammed river that still allows for passage).

The following types of Federal actions, among others, may destroy or adversely modify critical habitat:

- Actions that would appreciably reduce the abundance of riverine prey for larval and juvenile sturgeon, or of estuarine and marine prey for juvenile and adult Gulf sturgeon, within a designated critical habitat unit, such as dredging; dredged material disposal; channelization; in-stream mining; and land uses that cause excessive turbidity or sedimentation;
- Actions that would appreciably reduce the suitability of Gulf sturgeon spawning sites for egg deposition and development within a designated critical habitat unit, such as impoundment; hard-bottom removal for navigation channel deepening; dredged material disposal; in-stream mining; and land uses that cause excessive sedimentation;
- Actions that would appreciably reduce the suitability of Gulf sturgeon riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, believed necessary for minimizing energy expenditures and possibly for osmoregulatory functions, such as dredged material disposal upstream or directly within such areas; and other land uses that cause excessive sedimentation;
- Actions that would alter the flow regime (the magnitude, frequency, duration, seasonality, and rate-of -change of fresh water discharge over time) of a riverine critical

habitat unit such that it is appreciably impaired for the purposes of Gulf sturgeon migration, resting, staging, breeding site selection, courtship, egg fertilization, egg deposition, and egg development, such as impoundment; water diversion; and dam operations;

- Actions that would alter water quality within a designated critical habitat unit: including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, such that it is appreciably impaired for normal Gulf sturgeon behavior, reproduction, growth, or viability, such as dredging; dredged material disposal; channelization; impoundment; in-stream mining; water diversion; dam operations; land uses that cause excessive turbidity; and release of chemicals, biological pollutants, or heated effluents into surface water or connected groundwater via point sources or dispersed non-point sources;
- Actions that would alter sediment quality within a designated critical habitat unit such that it is appreciably impaired for normal Gulf sturgeon behavior, reproduction, growth, or viability, such as dredged material disposal; channelization; impoundment; instream mining; land uses that cause excessive sedimentation; and release of chemical or biological pollutants that accumulate in sediments;
- Actions that would obstruct migratory pathways within and between adjacent riverine, estuarine, and marine critical habitat units, such as dams, dredging, point-source-pollutant discharges, and other physical or chemical alterations of channels and passes that restrict Gulf sturgeon movement (68 FR 13399).

5.1.2. Conservation Value of Gulf Sturgeon Critical Habitat

The 14 riverine and estuarine/marine habitats were included in the designation because it is believed that with proper management and protection, they collectively represent the habitat that is necessary for the conservation of the species (68 FR 13370, March 19, 2003). These selected units were chosen to be designated because they are areas that contain one or more of the PBFs essential to the species. The analysis of this Biological Opinion focuses on the riverine units of critical habitat, therefore, the 7 estuarine/marine units will not be discussed further.

Unit 1

The Pearl River distributaries are used for migration to spawning grounds, summer resting holes, and juvenile feeding (68 FR 13370, March 19, 2003). The presence of juvenile sturgeon in the river system indicates successful spawning at some location in the river system. The only suitable spawning habitat believed to occur in the Pearl River system occurs north of the sills on the Pearl River and Bogue Chitto River with access to these areas limited only to periods of high flows (Morrow et al. 1996; Morrow et al. 1998). The typical bedrock and limestone outcroppings preferred for spawning in other river systems do not occur in the Pearl River system; however, sturgeon spawning areas in the Pearl drainage likely include soapstone, hard clay, gravel and rubble areas, and undercut banks adjacent to these substrates (W. Slack pers. comm. 2001). Even though upstream movement is blocked by Poole's Bluff Sill during periods of low water, potential spawning sites have been identified upstream of the sill at various locations between Monticello, Lawrence County, Mississippi, and the Ross Barnett Dam spillway, Hinds and Rankin Counties, Mississippi (F. Parauka, pers. comm. 2002) and sturgeon have been reported as far upstream as Jackson, Hinds County, Mississippi (Morrow et al., 1996;, Lorio 2000; W. Slack pers. comm. 2002). Suitable spawning habitat occurs within the Bogue Chitto River upstream of the Bogue Chitto Sill (W. Slack pers. comm. 2001; W. Granger, FWS, pers. comm. 2002; F. Parauka pers. comm. 2002) and juvenile, adult, and subadult sturgeon have been documented on the Bogue Chitto as far upriver as 1 mile north of Quinn Bridge (Mississippi State Highway 44), McComb, Pike County, Mississippi (W. Slack pers. comm. 2001; D. Oge, Louisiana Department of Environmental Quality, pers. comm. 2002; F. Parauka, pers. comm. 2002); therefore, the main stem of the Bogue Chitto River upstream of the Quinn Bridge to Mississippi State Highway 570 has been included in this unit.

Unit 2

The subpopulation of the Pascagoula River, based on captures in summer holding areas, ranges between 162 and 216 sturgeon; however, these estimates are primarily based on large fish and do not account for juvenile or subadult fish (Heise et al. 1999; Ross et al. 2001; S. Ross USM pers. comm. 2001). The only confirmed spawning area in the Pascagoula River drainage occurs on the Bouie River and was confirmed via egg collection in 1999 (Slack et al. 1999; Heise et al. 2004; Sulak et al. 2016). Gulf sturgeon have been documented using the downstream area of the Bouie River as a summer holding area (Ross et al. 2001). The documented sightings of sturgeon and identified suitable spawning habitat upstream to Mississippi Highway 588 (Reynolds 1993; W. Slack pers. comm. 2002; F. Parauka pers. comm. 2002), confirmed use as a migratory corridor, and confirmed use by juvenile sturgeon are the reasons for the inclusion of the Leaf River in this unit. The Chickasawhay River has had documented sightings of sturgeon, presence of suitable spawning habitat, and migratory movement of sturgeon (Miranda and Jackson 1987; Reynolds 1993; Ross et al. 2001). The West and East distributaries of the Pascagoula River are used by Gulf sturgeon during spring and fall migrations. Big Black Creek and the Pascagoula River have had documented summer resting areas.

Unit 3

Larval sightings and suitable spawning habitat have been reported on the Conecuh River and spawning confirmed between River Mile 100 and 105.6 (Parauka and Giorgianni 2002; N. Craft, Florida Department of Environmental Protection pers. comm. 2001). At five sites along the Escambia River, between rkms 161-170 (RM 100-105), eggs have been collected (Craft et al. 2001; Sulak et al. 2016). The Sepulga River has been described as having smooth rock walls, and long pools with stretches of rocky shoals and sandbars which makes for suitable spawning habitat for the Gulf sturgeon (Estes et al. 1991). Scour holes in the lower Escambia River have been found as holding areas for Gulf sturgeon (Stewart et al. 2012; Sulak et al. 2016). It is believed that Gulf sturgeon likely use the Escambia River main stem and all the distributaries for exiting and entering the Escambia/Conecuh River as the use of distributaries in other systems for this purposes has been documented.

Unit 4

Multiple areas of limestone outcrops have been documented as possible spawning sites on the Yellow River because YOY sturgeon are observed near these types of riverine features, which also confirms that reproduction is occurring in this subpopulation (Parauka and Giogianni 2002; Craft et al. 2001). Potential summer resting areas along the main stem of the Yellow River have also been identified. Shoal River summer resting habitats have been confirmed (Lorio 2002), as well as summer resting and staging sites on the Blackwater River main stem and between the Wright and Cooper Basins (Reynolds 1993; Craft et al. 2001).

Unit 5

Suitable spawning habitat has been identified from the Elba Dam to the Pea River with one confirmed spawning location; however, the Elba Dam blocks sturgeon migration further upstream at all flow conditions (Parauka and Girgianni 2002; Hightower et al. in press). The lower reaches of this river system have often been used for summer resting (Fox et al. 2000). The main stem of the Choctawhatchee River has had several spawning sites and resting area identified with male Gulf sturgeon in spawning condition found near these areas (Parauka and Giorgianni 2000; H. Blalock-Herod, FWS pers comm 2002; Hightower et al. in press). With the capture of sturgeon in the Indian River, Cypress River, and Bells Leg during March and April, it is likely that sturgeon are using these tributaries as migratory corridors to and from the Choctawhatchee River main stem.

Unit 6

With the construction of the Jim Woodruff Lock and Dam in the 1950s, the Gulf sturgeon was restricted to the portion of the Apalachicola River downstream of the dam. Resting aggregations and successful spawning has been confirmed at the base and just downstream of the dam (Sulak et al. 2016; Pine et al. 2006; Scollan and Parauka 2008; Parauka and Giogianni 2002; Wooley et al. 1982). The Brothers River has been documented to have sturgeon use the area as a resting and possible osmoregulation area before migrating into estuarine and marine habitats for winter feeding (Wooley and Crateau 1985).

Unit 7

Spawning sites within the river have been confirmed with the collection of eggs on artificial substrate (Marchant and Shutter 1996; Sulak and Clugston 1999) with YOY sturgeon having been documented in the river system (Carr et al. 1996; Sulak and Clugston 1999; K. Sulak, pers. comm. 2002; Clugston, pers. comm. 2002). Multiple resting areas throughout the Suwannee River have been discovered as well (Foster and Clugston 1997). Gulf sturgeon adults use the East Pass and West Pass for emigration and immigration (Mason and Clugston 1993; Edwards et al., in prep.). Telemetry data for the Suwannee River found that male Gulf sturgeon enter the river in late January to mid-February and rapidly swim to the staging areas just below the upriver spawning grounds (USGS-WARC, unpublished telemetry database; Sulak et al. 2016). For all of these reasons these areas were included in this unit.

5.1.3. Tables and Figures for Status of Gulf Sturgeon Critical Habitat

Figure 5.1. Designated critical habitat and historical range of Gulf sturgeon.

5.2. Environmental Baseline for Gulf Sturgeon Critical Habitat

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of designated critical habitat for Gulf sturgeon within the Action Area. The environmental baseline refers to the condition of the listed species or its designated critical habitat in the Action Area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the Action Area, the anticipated impacts of all proposed Federal projects in the Action Area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR §402.02).

5.2.1. Action Area Conservation Value of Gulf Sturgeon Critical Habitat

Two of the seven PBFs identified for Gulf sturgeon critical habitat (see section 5.1.1) do not occur in the Action Area: riverine spawning sites and riverine aggregation (resting) areas. Spawning sites and aggregation areas are thought to be downstream of the Action Area (see section 4.2.1). The PBFs found in the Action Area are food, flow regime, water quality, sediment quality, and migratory pathways.

The Action Area occurs on the Pearl River around Jackson, MS. The Pearl River is included in Critical Habitat Unit 1, the Pearl and Bogue Chitto Rivers in Louisiana and Mississippi, which is currently known to support a reproducing subpopulation of Gulf sturgeon. Unit 1 consists of a total of 494 miles. The Action Area occurs at the top extent of this Critical Habitat Unit. This section of the river has been previously altered throughout the 20th century by channelization and dredging of the river, levee systems, and a weir for the water supply of the City of Jackson. As discussed in Section 4, the Ross Barnett Reservoir prevents Gulf sturgeon migration north of the reservoir. The City of Jackson water supply weir can impede sturgeon migration up to the Ross Barnett dam except in high flow events. On the Lower Pearl River, the Poole's Bluff Sill, a low-head dam, also serves as an impediment to sturgeon migration to the upper reaches of the river except in high water events.

While adult sturgeon do not usually feed in freshwater, juveniles forage extensively in rivers on aquatic insects, worms, and mollusks (Mason and Clugston 1993; Huff 1975; Sulak and Clugston 1999). A specific study of the macroinvertebrates (i.e., detritus, aquatic insects, worms, and/or mollusks) has not been conducted; however, with the varying aquatic species within the Action Area that feed on those types of prey it can be assumed that the area does contain enough of these prey items to support the populations of species that inhabit the area.

This area of the Pearl River has been altered in the past by dredging and channelization, losing 5.34 miles of meanders. Suitable spawning substrate within the Pearl River likely includes soapstone, hard clay, gravel, and rubble areas and undercut banks adjacent to these substrates (W. Slack, pers. comm. 2001). Specific surveys have not been conducted on the substrate of the river within the Action Area; however, grab samples were taken as part of the Wetland Delineation conducted for the EIS/Feasibility Study that did not exhibit the suitable substrates necessary for sturgeon spawning in the Pearl River. Critical habitat was designated up to the Ross Barnett dam on the Pearl River due to the potential of spawning sites being identified between Monicello, Mississippi, and the Ross Barnett Reservoir (F. Parauka, pers. comm. 2002); however, migration past the Jackson water supply weir to any potential spawning ground upstream towards the reservoir is impeded unless there is a high water event.

As discussed in section 4.2.1, the reach of the Pearl River in the Action Area can have fairly fast, deep flows during rain events but has shallow baseline flows and can exhibit shallow flows during certain parts of the year. Gulf sturgeon depend on flow regimes in the riverine environment for all life stages including migration, breeding site selection, courtship, egg fertilization, resting and staging, and for maintaining spawning sites in the suitable condition needed for egg attachment, sheltering, resting, and larval staging. Based on average flow rates from 1966 to 2013, this area of the river currently has high flows during the springtime with flows decreasing significantly during the summer (Table 2.1).

Water quality in the Action Area was discussed in section 4.2.1; however, a brief summary is provided here as well. In 2019, a water advisory was issued for the Pearl River in Jackson due to continued discharges of sanitary sewer overflows into the river. In the Action Area, there is a former creosote plant as well as two former landfills from which debris periodically washes into the river. Leachates from these landfills were found to contain heavy metals above the regulatory standards. In 2003, the EPA also found barium, cobalt, zinc, and other contaminates in the river in the Action Area.

5.3. Effects of the Action on Gulf Sturgeon Critical Habitat

This section analyzes the direct and indirect effects of the Action on critical habitat for Gulf sturgeon. Effects of the Action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the Action may occur later in time and may include consequences occurring outside the immediate area involved in the Action (50 CFR §402.02). Our analyses are organized according to the description of the Action in section 2 of this BO.

5.3.1. Effects of Channel Excavation and Levee Relocation on Gulf Sturgeon Critical Habitat

The PBFs of flow regime, sediment quality, and migratory pathways would not be impacted by the construction of the channel excavation and levee relocation; therefore, this section will discuss the effects of the excavation and relocation on the PBFs of food and water quality.

As previously discussed in section 4.3.1, the channel excavation and levee relocation would occur during low water periods on the outer banks and an approximately 100-foot buffer along the riverbank would be maintained during excavation to retain some bank stability. Although the excavation and relocation would occur during low water periods, the Action Area would still be subject to increased sedimentation and turbidity should a heavy rainfall occur during construction and before vegetation cover could be reestablished. Increased turbidity when rainfall is the highest is a normal part of variations in turbidity following seasonal patterns of rainfall (Kishinh et al. 2006); however, the increase in turbidity would be additive to the normal turbidity surge due to the excess amount of loosened sediment during construction. Important contributors to the decline of aquatic assemblages are habitat degradation, sedimentation, and turbidity (Sawyer et al. 2004; Stewart and Swinford 1995; Henley et al. 2000). The increased turbidity and sedimentation would lead to impacts on water quality, which then leads to impacts on the prey base for juvenile sturgeon. See section 4.3.1 for more information on the effects of sedimentation and turbidity on the juvenile sturgeon food supply. These impacts on water quality would be temporary and would be reduced through erosion control measures.

5.3.2. Effects of Weir Construction and Impoundment on Gulf Sturgeon Critical Habitat

With the establishment of the 1,500-foot wide impoundment from the construction of the weir, changes to water velocity and water surface elevation in the Action Area would be anticipated. The weir has been designed to mimic the existing discharge of the river, and any changes in river discharge should be minimal once the pool area has been filled to the top of the weir. The impacts of flow are discussed in more detail in section 4.3.2.

Dams such as the proposed weir present an obstacle to sturgeon migration and are thought to be the main hindrance to Gulf sturgeon recovery in the Pearl River (Sulak et al. 2016). As a way to offset the effects of the weir on sturgeon migration, the construction of a fish passage channel is part of the proposed action. Kohl (2003) evaluated the opportunity to design a proposed bypass at the Poole's Bluff Sill to assist Gulf sturgeon migration north of the sill. That evaluation determined that a bypass channel could assist in sturgeon migration as long as the feature was designed to accommodate sturgeon swim speeds and other factors such as flow. Thus, if the fish passage channel were designed properly, it should provide for sturgeon migration past the weir structure. The impacts that the proposed weir and fish passage channel would have on sturgeon migration are discussed in detail in section 4.3.2.

Gulf sturgeon critical habitat in the Action Area is likely to experience reduced water quality during the construction of the weir and fish passage channel as a result of increased sedimentation and turbidity. As discussed in sections 5.3.2 and 4.3.2, this temporary effect would influence the macroinvertebrate community upon which juvenile sturgeon feed. Pools created by impoundments generally consist of fewer taxa of macroinvertebrates than free-flowing river reaches which tend to support a more diverse macroinvertebrate community (Santucci et al. 2005). However, Dean et al. (2002) found that while impounded areas lacked species diversity, the abundance of individuals was similar to that of free-flowing river reaches. Water quality was also modeled for the impoundment, and the results indicated that water quality would not significantly decline from the current condition. Accordingly, we could assume that the project would have minimal effects on the macroinvertebrate community because of the lack of changes to baseline water quality conditions. As discussed in section 5.2.1, suitable spawning substrate has not been indicated in this reach of the river; therefore, it is unlikely that spawning occurs in the area. However, sediment quality is important for more than just spawning. Sediment quality is also necessary for the macroinvertebrate food sources of juvenile sturgeon. The impacts to sediment quality on the macroinvertebrate community would be similar to impacts to water quality as described above.

As discussed in section 4.3.2, DO and temperature are other important water quality factors for sturgeon. As temperature increases, DO levels decrease which can affect the growth and respiration rates of juvenile sturgeon. Water quality modeling conducted for temperature and DO indicated post-project levels would have a slight but not significant difference from the pre-project levels. Thus, we can reasonably assume that while sturgeon may be stressed from any slightly lower levels, they would still continue to feed and grow.

5.3.3. Effects of Non-Federal Activities caused by the Federal Action on Gulf Sturgeon Critical Habitat

The relocation or retrofitting of existing infrastructure within the Action Area would lead to a reduction in sediment and water quality from the increased sedimentation and turbidity involved with implementing those actions. The increase in development adjacent to the improved channel could also lead to a reduction in water and sediment quality in the Action Area. Effects to water quality would be the same as mentioned above. The effects from decreased reduction in water quality are discussed in sections 4.3.2 and 5.3.3.

5.3.4 Summary of Effects of the Action on Gulf Sturgeon Critical Habitat

The Action Area encompasses a total of 19.37 miles of critical habitat that would be affected due to the Action. Thus, we estimate that approximately 3.9 percent of critical habitat Unit 1 and 1.1 percent of the total riverine critical habitat units would be impacted by the Action. Based on the best available data, the collective activities from the Action would affect the PBFs including food, flow regime, water quality, sediment quality, and migratory pathways, however, these impacts would either be temporary or offset by activities such as the construction of a fish passage for migratory purposes.

5.4. Cumulative Effects on Gulf Sturgeon Critical Habitat

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. At this time, the Service is unaware of any future state, tribal, local, or private non-Federal actions planned or scheduled that would occur in the Action Area. Therefore, cumulative effects are not relevant to formulating our pinion for the Action.

5.5. Conclusion for Gulf Sturgeon Critical Habitat

In this section, we summarize and interpret the findings of the previous sections for Gulf Sturgeon critical habitat (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:

a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.

"*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species" (50 CFR §402.02).

The Action Area occurs at the northernmost extent of Critical Habitat Unit 1. As discussed in Section 4, the Action Area encompasses the Pearl River and its adjacent lands from the Ross Barnett dam south to 1.6 miles south of the weir structure. A total of 19.37 miles of critical habitat would be affected equaling approximately 3.9 percent of Unit 1 and 1.1 percent of the total riverine critical habitat units being impacted. (Please note that this assessment does not include the estuarine and marine units of critical habitat because none would be affected and because we are specifically addressing changes to the riverine portions of Gulf sturgeon critical habitat due to the PBFs being affected.) The PBFs impacted by this Action are food, flow regime, water quality, sediment quality, and migratory pathways.

Water and sediment quality go hand in hand when it comes to effects on food resources. The analyses of water and sediment quality impacts from implementing the Action indicate that impacts would be either temporary or insignificant, which infers that impacts to food sources for foraging sturgeon would be minimal. A reduction in water and sediment quality from sedimentation and turbidity from the indirect actions (e.g., relocation of and retrofitting existing infrastructure) would also be temporary, because water quality would return to similar conditions once such actions are completed. The modeling of water quality parameters, specifically temperature and DO, pre- and post-project does not indicate a significant difference in those parameters; thus, water quality in the Action Area would not be permanently degraded to such a degree that sturgeon would not be able to use the area.

The weir structure would impact the migratory pathway to sturgeon movement into this reach of the river. To offset the effects to sturgeon migration from the weir, a fish passage structure has been designed for just downstream of the weir. There is no documentation of sturgeon using fish passage structures, but studies show that certain designs would make migration through a fish passage more successful. Specifically, swim speed should be considered in the design of the fish passage feature in order to maintain sturgeon migration into the Action Area post-construction.

Our analysis indicates that while the Action would have negative effects to 3.9 percent of Critical Habitat Unit 1 and 1.1 percent of the riverine units as a whole, it is not likely to appreciably diminish ability of Unit 1 to provide the intended conservation value to the Gulf sturgeon and would not result in an adverse modification to Critical Habitat Unit 1.

6. RINGED MAP TURTLE

6.1. Status of the Ringed Map Turtle

This section summarizes best available data about the biology and current condition of the Ringed map turtle (*Graptemys oculifera*) throughout its range that are relevant to formulating an opinion about the Action. The Service published its decision to list Ringed map turtle as threatened on December 23, 1986 (51 FR 45907 45910). The most recent published 5-year review was completed August 17, 2010. A new 5-year review was requested to be initiated May 7, 2018 (83 FR 20092 20094).

6.1.1. Description of the Ringed Map Turtle

For a thorough description of the ringed map turtle see Jones & Selman (2009); all information in this section can be found in that description unless otherwise cited. The ringed map turtle is a small turtle. Each shield of its upper shell (carapace) has a yellow ring bordered inside and outside with dark olive-brown; its undershell (plastron) is yellow. The head has a large yellow spot behind the eye, two yellow stripes from the orbit backwards, and a characteristic yellow stripe covering the whole lower jaw. Males grow on average to 3.5 inches (89 millimeters) and females to 6 inches (156 millimeters) in plastron length.

6.1.2. Life History of the Ringed Map Turtle

The ringed map turtle's habitat is typically riverine with a moderate current and numerous basking structures. Using data from five studied populations in Mississippi, river conditions have been described as:

- width from 67 to 361 feet (20 to 110 meters);
- mean stream flow rates from 3,000 to 15,000 cfs; and
- river bottom composed of clay, sand or gravel.

This species has also been observed in oxbow lakes that are connected or disconnected from the main river system. It is assumed that turtles observed in disconnected lakes arrived due to flooding and remained or were isolated during construction of the levees. Individuals have been reported from the Ross Barnett Reservoir, although there is no evidence of a breeding population there or in any disconnected lakes (Selman 2018). Basking structures vary from deadwood to man-made structures (e.g., culverts, shopping carts, etc.). The turtles are found in rivers that must be wide enough to allow sun penetration for several hours. Turtles prefer basking sites which are partially submerged in areas with the deepest water and swiftest current. The occurrence of downed trees within the river has been strongly associated with the presence of *Graptemys* (Killebrew et al. 2002; Linderman 1997, 1998, 1999). However, ringed map turtles have also been found in areas that are predominately shallow with few deep areas (Selman and Smith 2017).

The preferred velocity of the ringed map turtle has not been determined; however, Killebrew et al. (2002) determined that the Cagles map turtle (*Graptemys caglei*) preferred velocities from 0.5 to 2.5 fps. Shealy (1976) stated that the Alabama map turtle (*Graptemys pulchar*) was found in velocities typically ranging from 0.9 fps to 2.7 fps. To aid in the impact determination the Service examined computer modeled without project velocities for that reach of the river where the stable Lakeland population is found; typically velocities ranged from 0.4 to 2.0 fps. Because those velocities are mean cross-sectional velocities the Service used that information and the Cagle's map turtle velocities to hypothesize that suitable velocities for the ringed map turtle would likely occur between 0.5 and 2.5 fps.

Nesting habitat consists of large, high sand bars adjacent to the river. Sandbars range in size from 430 square feet (40 square meters) to over 2.2 acres (8,900 square meters) and are generally composed of 39 percent open sand, 38 percent herbaceous vegetation, and 23 percent woody vegetation (Jones 2006). Nesting has also been reported to be attempted in shell road beds and mowed grassy areas adjacent to the river. Nesting occurs during daylight hours from mid-May through mid-July. Nest sites are usually located, on average, 59 feet (18 meters) from water and within 3.3 feet (1 meter) of vegetation, with an average canopy cover of approximately 37 percent.

The diet of the ringed map turtle consists primarily of insects (caddisflies, diptera, mayflies, and beetles) and mollusks. Some observational data have also pointed to carrion as a food source. Selman and Linderman (2018) postulated that ringed map turtles may also consume freshwater sponges as do other *Graptemys*. The presence of wood in diet samples of the ringed map turtle indicate that sponges and vegetative prey items (e.g., filamentous algae) may occur along with animal prey on deadwood substrate.

Jones (2006) found a minimum of approximately 60 percent of the females reproducing annually, but some females may skip a year between nesting. Nesting was found to only occur during daylight hours and primarily before noon. Nesting is initiated in May and ends in August with multiple (2 to 3) clutches per year being common (annual clutch frequency ranged from 0.96 to 1.42). Clutch size averaged approximately 3.6 eggs per nest. Final nesting attempts usually ended towards the end of July. Eggs incubate for approximately 64 days (Jones 2006) before pipping and then hatchlings emerge approximately 12 days after pipping (total time in the nest is approximately 76 days).

Mean annual survivorship estimates for males, females, and juveniles were 0.88, 0.93, and 0.69, respectively. Maximum longevity estimates were 48.8 years for males and 76.4 years for females. Average longevity estimates were 13.9 for females and 8.5 years for males. The sex ratio of captured turtles was male-biased before 2000 but unbiased after 2000. Time to maturity varies between male and female turtles. Males mature at about 4.6 years of age while females mature about 9.1 years of age (Jones 2017).

6.1.3. Numbers, Reproduction, and Distribution of the Ringed Map Turtle

The ringed map turtle is restricted to the main channels of the Pearl, Strong, and Bogue Chitto Rivers in Mississippi and Louisiana (Figure 6.1). It occurs in most reaches of the Pearl River from near the coastal salt water influence in St. Tammany Parish, Louisiana, upstream to Neshoba County, Mississippi. It only occupies the lower approximately 4.7 miles of the Strong River in Simpson County. In the Bogue Chitto River it is found upstream to Warnerton, Louisiana. Occupied river miles are estimated to be 488.5 miles.

Using 25 years of data at 5 sites along the Pearl River in Mississippi, Jones (2017) provides the most recent information on long-term trends for the ringed map turtle in the mid- and upper-Pearl River. While the population trend as a whole remained stable over the 25 years of the study, one site showed decline (Carthage), three sites showed the initial stages of decline (Ratliff Ferry, Monticello, Columbia), and one site (Lakeland) is relatively stable (Table 6.1).

In 2012, Landry conducted survey on the Pearl River near Bogalusa. Landry & Gregory (2010) conducted the most recent survey of the Bogue Chitto River following up on a 1999 survey of the same reach (Shively 1999). Between 6.51 to 114.7 turtles/kilometer (km) were estimated between the confluence of the river and Warnerton, Louisiana. Turtle concentrations were higher in the downstream reaches, potentially due to acclimation to human disturbance. Ringed map turtle numbers were down from the previous survey.

Recent surveys on the Pearl River below the Bogue Chitto River are limited. Dickerson and Reine (1996; summarized in Selman and Jones 2017) surveyed between Pools Bluff and Hwy 90 and found between 15.7 (Bogue Chitto Sill) to 1.4 (Pools Bluff) turtles/km. Along the East Pearl River at Stennis Western Maneuver Area, basking surveys conducted from 2012 to 2015. Abundance was estimated between 1.2 and 6.8 turtles/km (Buhlmann 2017). Over all, there are 12 relatively recent separate surveyed areas across the ringed map turtle range. These surveyed reaches represent 1.37 percent of the species' range, but there are river reaches longer than approximately 80 miles that have neither recent nor any survey information. The survey reports for six areas did not mention a trend in the abundance. Of the remaining six, one was declining, three were in the initial stages of decline, another one was declining but the stage of decline was not stated, and only one was stable.

The use of basking surveys to obtain a relatively good indication of the abundance level has been suggested by Jones and Hartfield (1998) and Killebrew et al. (2002). To determine the overall abundance the Service estimated the occupied river miles within the species' range based on literature. The U.S. Geological Survey's (USGS) stream reporter (txpub.usgs.gov) was used to determine river miles; adjustments to those mileages were done in ArcGIS using the National Hydrologic Database. The mean number of basking turtles observed for all surveyed reaches was extrapolated to the unsurveyed reaches to estimate a range wide abundance. The Service assumed an even distribution across the range; however, Killebrew et al. (2002) and Lechowicz (undated) found *Graptemys* were not always evenly distributed. The range in abundance displayed in Jones (2017) indicates an unequal distribution for the ringed map turtle; however, the Service could not find literature indicating a better method for determining *Graptemys* distribution and abundance. The Service used an average of all surveyed reaches to calculate an average abundance per river mile of the Pearl River (26.6 turtles/km) and used the average number of turtles from Landry and Gregory (2010) to calculate the abundance in the Bogue Chitto; the Service estimates approximately 17,916 turtles occur across the species' range (Table 6.1).

6.1.4. Conservation Needs of and Threats to the Ringed Map Turtle

Several threats were identified at the time of listing of the ringed map turtle (1986):

- habitat modification (desnagging, channelization, impoundment, and erosion),
- water quality degradation (pollution & siltation),
- over-utilization (collection for the pet trade and shooting of basking turtles for recreation);
- disturbance of nesting and basking (due to recreation and boating); and,
- The subsequent recovery plan (1988) identified predation as an additional threat.

At the time of listing, 21 percent of the ringed map turtle's range had been modified by channelization or impoundment and an additional 28 percent of that range had construction projects planned or authorized. This includes the Ross Barnett Reservoir and a channelized section within the Action Area. While many of the projects have not been constructed, and some are no longer under consideration, some are still authorized and may be initiated if funding becomes available. The ringed map turtle is not found within the approximately 16-mile-long Ross Barnett Reservoir which creates a barrier to turtle movement, though a small remnant population is found in Pelahatchie Creek near the dam. It has been stated that operations of the reservoir have created downstream impacts to habitat including channel filling and widening due to collapse of waterlogged banks from sudden water releases to maintain pool elevations (Selman and Jones 2017) and channel instability resulting from captured sediment in the Ross Barnett Reservoir (Hasse 2006; Kennedy and Hasse 2009; Tipton et al. 2004). Killebrew et al. (2002) stated that populations of Cagle's map turtle (*Graptemys caglei*) found downstream of a dam were decreased due to rapid changes in the water level associated with dam releases. Those dam releases were implicated in the flooding of nests and reduced food availability. Richards and Seigle (no date) stated that fluctuating water levels downstream of a dam altered habitats, reduced turtle movements, and resulted in loss of basking habitat for the northern map turtle (*Graptemys geographica*).

The recovery plan recognized that to reduce the threat of habitat modification, habitat protection was needed. Criteria 1 of the plan identified that protection of a total of 150 miles of the turtle's habitat in two reaches of the Pearl River was needed to delist the species. The reaches must be on opposite ends of the Ross Barnett Reservoir, and there must be a minimum of 30 miles in either reach. Currently there is only one protected reach north of the Ross Barnett Reservoir, an approximately 11.8–mile-long ringed map turtle sanctuary. Just south of the reservoir one of Mississippi Department of Transportation's mitigation banks protects 21,491 linear feet (approximately 4 miles) of the east bank of the Pearl River. Approximately 73 miles of at least one bank in the lower Pearl River is within state or federally protected and managed lands, but this area has some of the lowest population densities. Thus, additional protected areas are needed to meet this recovery goal. However, placing lands within a protected status may not be sufficient to preclude the decline of a turtle species (Browne and Hecnar 2007); additional management actions may also be required.

Agricultural, municipal, and industrial effluents may have historically impaired water quality in the lower Pearl River (McCoy and Vogt 1980). Direct effects of water quality on ringed map turtles has not been researched, but negative effects to their primary food sources has (Stewart et al. 2005). Decreases in other *Graptemys* species have been attributed to reduced water quality downstream of development (Killebrew et al. 2002). Selman and Jones (2017) cited studies that pointed to the decrease or loss of *Graptemys* species due to poor water quality in the Pearl River; recovery of those Pearl River populations due to improved water quality was also noted.

Predation of nests by raccoons, armadillos, and fish crows is well documented with most nests being predated within 14 days (Jones 2006). Predator numbers have increased and may be subsidized by humans which could have an impact on recruitment (Bulhmann 2017; Jones 2006). Jones and Selman (2009) suggested that those predators could have a significant impact to recruitment in the future. A recent increase in American alligators (*Alligator mississippiensis*) was also postulated to have possibly resulted in a decline in adult males and juveniles (Jones 2017). Predation is estimated to destroy approximately 86 percent of nests, and invertebrates (i.e., ants and fly larvae) kill an additional 24 percent of fertilized eggs within nests.

The impact of human disturbance, primarily recreating (e.g., camping, picnicking, boating) to nesting turtles and/or nests has been pointed to as another source of decline in the population (Jones 2006; Jones 2017; Selman and Jones 2017). Horne et al., (2003) found that even their observation blind reduced *Graptemys flavimaculata* nesting attempts by three times more than without that disturbance. Direct mortality associated with recreational and commercial fishing and recreational boating has been identified as another impact to *Graptemys* populations (Bluté et al. 2010; Selman et al. 2013; Smith et al. 2018). Jones (2017) expressed a concern about those same activities impacting the ringed map turtle. Blute et al., (2010) found that impacts to the northern map turtle (*Graptemys geographica*) from boat strike could lead to an increased risk of localized extinction. The potential for reduced vigor because of disturbed basking has been found in other *Graptemys* populations as well as the ringed map turtle (Heppard and Buchholz

2018; Selman and Qualls 2011; Selman et al., 2013). Based on basking surveys it is apparent that *Graptemys* species including the ringed map turtle may habituate to humans, the amount of time required for such habituation is not known and there is some uncertainty as to the degree of habituation that will occur. (Jones and Hartfield 1995; Landry and Gregory 2010; Lechowicz 2013; Selman and Jones 2017, Selman and Qualls 2011; Selman et al., 2013).

Listing of the ringed map turtle as federally threatened may have reduced impacts of the pet trade that trade is still apparently operating within the Pearl River Basin (Jones 2017; Selman and Jones 2017).

The most recent 5-year review (USFWS 2010) confirmed that all of the threats continue.

6.1.5. Tables and Figures for Status of the Ringed Map Turtle

See following pages.

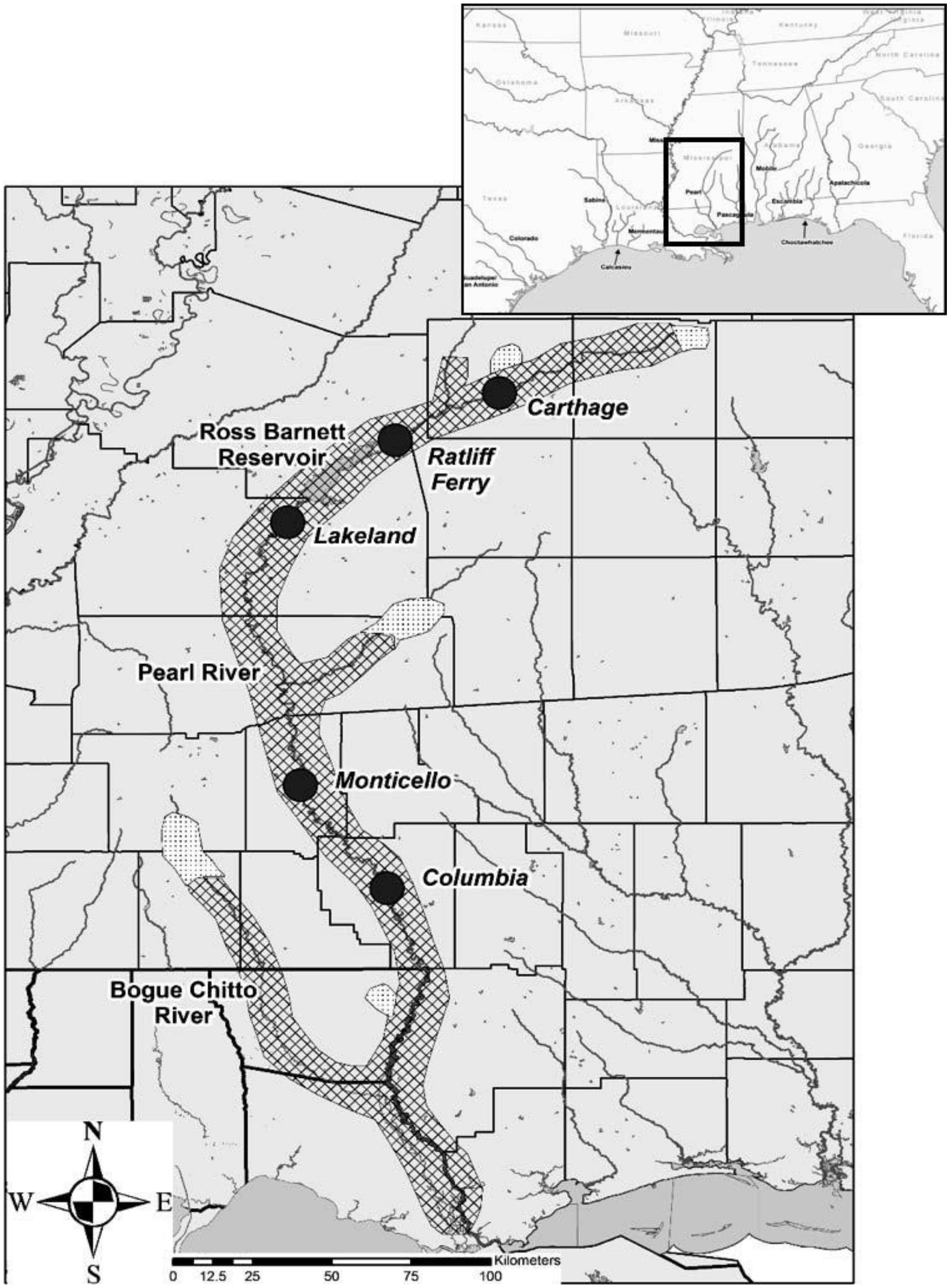


FIGURE. 6.1. The geographic location of the Pearl River in the southeastern United States (top inset) and map of sample sites in central Mississippi (bottom). Cross-hatching represents areas where *Graptemys oculifera* and *Graptemys pearlensis* co-occur; whereas stippling represents upstream areas only occupied by *G. pearlensis* (based on maps by Lindeman 2013) and new records of Lindeman (2014a, b). Taken from Selman and Jones 2017.

Table 6.1. Abundance estimates based on basking survyes1 and percent of species range.

Source	Location	Variation in number of turtles per kilometer (km)	Mean number of Turtles Observed	Total Number of Turtles Observed	Length of survey (km)	Turtles per km	Turtle Abundance (length of survey x turtles per km)	% of occupied river each survey area represents with channelized areas combined[3]	Surveyed reaches estimated percentage of abundance based on average abundance applied to non-surveyed reaches (26.6 turtles/km)[4]
Jones 2017	Pear River: Carthage	SD± 15.5	62		4.8	13	62.4	0.6	0.3
Jones 2017	Pearl River: Ratliff Ferry	SD± 51.1	188		3.2	59	188.8	0.4	1.1
Selman 2018	Pearl River: Jackson Reach S1[1]	173-389; SD±98	279.5		5.3	52.5	278.3	0.7	1.6
Selman 2018	**Pearl River: Jackson Reach S2**	**149-295; SD±63**	**220.6**		**5.3**	**41.5**	**220.0**		
Selman 2018	**Pearl River: Jackson Reach S3**	**42-77; SD±15**	**62.6**		**5.3**	**11.7**	**62.0**	**2.0**	**2.2**
Selman 2018	**Pearl River: Jackson Reach S4**	**59-177; SD±49**	**109.6**		**5.3**	**20.6**	**109.2**		
Selman 2018	Pearl River: Jackson Reach 5	166-291;SD±47	240.4		5.3	45.2	239.6	0.7	1.3
Jones 2017	Pearl River: Monticello	SD± 33.5	96		4.8	20	96.0	0.6	0.5

Table 6.1. Abundance estimates based on basking survyes1 and percent of species range – continued.

Source	Location	Variation in number of turtles per kilometer (km)	Mean number of Turtles Observed	Total Number of Turtles Observed	Length of survey (km)	Turtles per km	Turtle Abundance (length of survey x turtles per km)	% of occupied river each survey area represents with channelized areas combined[3]	Surveyed reaches estimated percentage of abundance based on average abundance applied to non-surveyed reaches (26.6 turtles/km)[4]
Jones 2017	Pearl River: Columbia	SD± 17.7	62		4.8	13	62.4	0.6	0.3
Landry and Gregory 2010	Bogue Chitto River	6.51 – 114.7		208	43.9	4.7	208.0	5.6	1.2
Landry 2012	Pearl River: West Pearl			121	10	12.1	121.0	1.3	0.7
Bulhammn 2017[2]	Pearl River: East Pearl and Mike's River			43	10	4.3	43.0	1.3	0.2
Total					108	26.6	1690.6	13.7	

[1]Basking surveys are used because not all surveys included trapping or mark/recapture thus to assess project impacts to the species range wide the most consistent/predominant method of surveying was used.

[1]This reach overlaps Jones 2017 Lakeland population, because Selman's data is more recent, Jones 2017 Lakeland population information is not included in the analysis.

[2] Mean for the three year sampling period was used because individual years included unidentified turtles.

[3] To determine the total length of river occupied by ringed map turtle the Service started at Hwy 15 Pearl River crossing south of Burnside subtracted the Ross Barnett Reservoir (16 miles) and the river mileage below Interstate 10 (14 miles); the East Pearl path was measured below the river bifurcation (total distance 602.4 km [380.4 miles]). The Bogue Chitto was measured starting at the confluence of McGee Creek in Mississippi, down the West Pearl to Interstate 10 (total distance 167.4 km [104 miles]). The lower 7.6 km (4.7 miles) of the Strong River. All miles total equals an estimated 786 km (488.5 miles).

[4] 26.6 turtles per km was multiplied by the potential total occupied river miles of the Pearl and Strong River (610 km) minus the Pearl River sampled reaches (64.1 km). This produces an estimate (16,226) of the overall abundance of turtles within the Pearl River range outside of the sampled reaches. The Bogue Chitto total occupied habitat was determined the same way but the turtles per km for that river were used (i.e., 167.4 km - 43.9 km = 123.5 km x 4.7 turtles/km = 580.5 turtles). Within the sampled reaches an abundance of approximately 1690.2 turtles and range wide is estimated at 17,916.

6.2. Environmental Baseline for the Ringed Map Turtle

This section is an analysis of the effects of past and ongoing human and natural factors leading to the current status of the Ringed Map Turtle, its habitat, and ecosystem within the Action Area. The environmental baseline refers to the condition of the listed species or its designated critical habitat in the Action Area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the Action Area, the anticipated impacts of all proposed Federal projects in the Action Area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 CFR §402.02).

6.2.1. Action Area Numbers, Reproduction, and Distribution of the Ringed Map Turtle

Recent surveys by Selman (2018) provide a current estimate of the number and status of ringed map turtles in the Action Area (Table 6.1 above). Ringed map turtles were also observed in oxbow lakes and sloughs adjacent to the Pearl River within the Action Area; however, not all oxbow and slough habitat will be altered by the proposed impoundments. Only a portion of Crystal Lake will be impacted by proposed levee realignment (set-back) and impoundment. Ringed map turtles are found throughout all reaches of the Pearl River within the Action Area, with lower numbers in the channelized sections of the River (just south of RM 293 to approximately RM 287). Approximately 40 percent of the proposed excavation area has little or no riparian habitat and little to no natural basking and feeding habitat, especially within the channelized portion. Selman (2018) found a greater concentration of turtles within forested riparian sites along this portion of the river. He also documented nest sites, turtle nesting crawls, and juvenile turtles all indicative of successful recruitment occurring in all stretches of the Action Area, including the area with reduced riparian habitat. A greater abundance of juveniles (10 to 20 percent) was found within the northern channelized section than in other sites outside of the Action Area. Selman (2018) postulated that the increased juvenile production may result from the use of narrower sandbanks along the channelized sections as opposed to sand bars, thus reducing predation success. Approximately 31.4 acres of accretion (e.g., sand bars, sand banks) were determined by the FCDCD to exist within the Action Area based on 2010 National Agriculture Imagery Program color photography; this acreage was spread over approximately 20 sites throughout the Action Area. Selman (2018) documented 20 sandbars within the project area and noted 102 potential nesting crawls. Of the 20 sand bars, 11 were not surveyed but two of those not surveyed were noted as having crawls but the number of crawls were not counted. Two surveyed sand bars had no nests or crawls.

Based on basking survey data, the Action Area represents 2 percent of the turtle's range having approximately 2 percent of the turtle's range wide abundance (Table 6.1 above). Jones (2006) used nesting survey data from upstream of the Action Area to develop a relationship between sandbar size and number of nests. Based on that relationship the approximate number of turtle nests found on the 31.4 acres of sandbars within the excavated area was calculated to be 1,177. With each nest having approximately 4 eggs per nest (rounded up from 3.6) this corresponds to approximately 4,326 eggs within the excavated area. However, once the 86 percent predation rate of nests and the 24 percent predation rate of eggs by insects are applied to the number of eggs only approximately 451 eggs are likely to hatch.

Not included in the above abundance estimate are the small isolated populations at Cypress, Crystal, and East and West Maye's Lakes within the project area. There has been no evidence of these populations reproducing (Selman 2018). Turtles at Crystal Lake were isolated from the river following levee construction. Due to the lack of riverine created habitat, especially nesting habitat, these populations are expected to eventually disappear. Selman (2018) counted 11 and 9 ringed map turtles at Cypress and Crystal Lakes, respectively. East and West Maye's Lakes were found to have 24 and 4 turtles, respectively; however, unlike Cypress Lake, the other lakes connect to the river during large flood events. Selman (2018) believed the population in both Mayes Lakes were supported by immigration only but were not viable. There are no construction activities proposed in the immediate vicinity of both Mayes Lakes and the adjacent Cypress Lake. Other ringed map turtle studies have typically not surveyed oxbows or lakes within the floodplain, though their presence was noted in downstream lakes.

Selman (2018) used basking density surveys along with basking frequency data from two *Graptemys* species found in the Pascagoula River to estimate population size within the Action Area (Selman and Qualls 2011; Selman and Lindeman 2015). Selman (2018) used correction factors of 20 and 30 percent of the basking population observed to estimate the potential range of turtles missed by such surveys. Killebrew et al. (2002) used a level of conspicuousness (between 33 and 36 percent) to estimate undetected turtles from basking surveys to predict population levels. The Service used the mid-point between Selman's ranges (i.e., 25 percent) as a reasonable method to estimate potential numbers in the Action Area. The Service also used survey results from Selman (2018) to determine the number of turtles within the channelized area and upstream and downstream of that area to which we applied the correction factor. Based on those calculations, the Service estimated that 2,196 turtles potentially exist in the area that will be inundated by the project. Upstream and downstream of the project area, we estimate approximately 1,556 and 1,164 turtles, respectively (2,720 total) with the later number representing the Lakeland population found north of the excavated area. In addition, we estimate that approximately 192 ringed map turtles inhabit Crystal, Cypress, and East and West Mayes Lake based on the 25 percent correction factor. In summary, we estimate a total of approximately 5,108 turtles occur in the Action Area. The estimated number of turtles includes juveniles as these are not separated in our analysis below.

Selman (2018) used information from two *Graptemys* species to develop the potential number of turtles in the Action Area. The use of surrogate species is common in conservation biology, particularly when implementing the ESA where needed data may not be available or is difficult to collect. Selman's use of a "correction factor" to determine population size within the project area is based off the peer-reviewed Selman and Qualls (2011) basking behavior study of an ecologically similar species (*Graptemys flavamaculata*, Yellow-blotched map turtle) within the Pascagoula River. This species is as equally imperiled as the ringed map turtle and suffers from similar threats. When lacking actual data for a species, we will use the best available information often from a surrogate species (Murphy & Weiland 2014; Caro 2010); Service policy on using surrogate species can be found in Final Rule 80 FR 90 26832-26845. Throughout this BO the Service has relied upon existing information about the ringed map turtle as much as possible; however, when information is deficient or absent the Service first examined literature regarding other *Graptemys* species, and then other aquatic turtle species to provide the best possible assessment of impacts to the subject species and its habitat.

6.2.2. Action Area Conservation Needs of and Threats to the Ringed Map Turtle

The current status of ringed map turtles in the Action Area has been heavily influenced by previous flood control actions and urbanization. Portions of the Pearl River within the Action Area have been channelized, desnagged, and contain a cleared floodway where woody vegetation is controlled via herbicide and/or mowing. These actions have reduced the amount of habitat available for this species, including reductions in basking material, potential foraging areas, and nesting sandbars. Relatively few deep areas are also found within this section. Degraded water quality through nutrient and pollution input through this urbanized section of the Pearl River may also be impacting the ringed map turtle populations. Even with these impediments the ringed map turtle manages to persist within the Action Area. Finally, the construction of the Ross Barnett Reservoir just north of the Action Area has resulted in a barrier to ringed map turtle migration from and into the Action Area. The significantly decreased water velocity within the reservoir, the lack of basking material, nesting habitat and increased development and recreational activities has resulted in the elimination of a viable population of ringed map turtles for the length of the reservoir (approximately 16 miles of the Pearl River, i.e., RM 302 to approximately RM 328). An isolated non-reproducing population is found in the Pelahatchie Creek area just north of the dam in the reservoir.

The Action Area contains one of the reaches selected in the 1988 Recovery Plan for long term population monitoring because of its perceived low population (Stewart 1988). This population, referred to as the Lakeland population, is 3 miles long and is found between the Ross Barnett Reservoir and Lakeland Drive (northern portion of the Action Area). The Service's most recent five-year status report states that the population at this location represents the healthiest population south of the Ross Barnett Reservoir. Approximately 30 percent of river within the Lakeland population area will be directly impacted by the project. Selman and Jones (2017) concluded that the Lakeland population is the only stable population they surveyed. There has been no long-term monitoring of the population south of Lakeland drive within the Action Area. Selman's recent surveys (2017, 2018) were the first efforts to document population status within this area.

The Ratliff Ferry population and populations to the north became isolated from populations south of the Ross Barnett Reservoir with the construction of that reservoir in 1960 (17 years short of the estimated longevity of a female). The ringed map turtle populations north of the reservoir are beginning to experience a decline. Predation and disturbance of nesting areas are believed to possibly be the greatest factors causing the decline along with sedimentation in the upper portion of its range (Jones 2017).

After studying the Ratliff Ferry and Lakeland populations, Heppard and Buchholz (2019) suggested that increases in boat traffic can be mitigated to some extent by providing greater basking perch abundance and by reducing boat speed and the interwake interval of passing boats. They recommended that no wake zones be placed around areas set aside for ringed mapped turtle conservation and that basking refugia be established by restricting boater access. Turtles basked for longer times in no wake zones. The proposed measures would be done to improve the health, survival and reproduction of the ringed mapped turtle and reduce the likelihood of boats striking adults. For the Lakeland and the three other populations studied in the northern part of the range, Selman and Jones (2017) attributed some of the population declines to direct mortality from boat strikes.

6.3. Effects of the Action on the Ringed Map Turtle

This section analyzes the direct and indirect effects of the Action on the Ringed Map Turtle. Effects of the Action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the Action may occur later in time and may include consequences occurring outside the immediate area involved in the Action (50 CFR §402.02). Our analyses are organized according to the description of the Action in section 2 of this BO.

6.3.1. Effects of channel excavation and levee relocation on the Ringed Map Turtle

To decrease sedimentation from the construction site and allow most excavation to occur in a dry environment, excavation of areas away from the river bank would occur first. This would leave the riverbank and an additional adjacent area separating the river from the excavation area undisturbed. During the final construction phase the buffer (i.e., river bank area) would be removed and then the area river would be closed and the area would flood. Prior to that, the buffer area would reduce the likelihood of disturbing basking turtles or turtles attempting to nest and would reduce the potential of having nesting sites located within ongoing work areas. Therefore, we anticipate a very small percentage of turtles will be killed due to ground disturbance activities away from the river

Disturbance from excavating 25 million cubic yards of material from approximately 1,901 acres within and adjacent to the river over approximately two years could result in death of individuals if they are unable to flee the construction work area. Most of the top bank of the river will be disturbed through the direct removal of vegetation, sand and dirt as well as through other associated ground disturbing activities such as stockpiling dirt, machinery egress and ingress, etc. Aquatic turtle research that focused on disturbances associated with construction found that aquatic turtles within a construction area would move up or downstream from the construction activity (Chen and Leu 2009; Plummer and Mills 2008). Therefore, it is reasonable to assume that many turtles currently found in the proposed impounded area will slowly move away from construction activities. As construction progresses upstream from the weir location it is assumed that most turtles will migrate upstream and will encounter the Lakeland population where the river will not be directly altered. At the downstream end of the project some turtles are likely to move downstream encountering turtles south of the weirs location. All turtles in the construction area (estimated at 2,196) are expected to be disturbed in some form of alteration of normal feeding, basking and nesting activities while channel excavation activities are taking place and they are displaced from the construction site.

A modest decline in the softshell turtle (*Apalone spinifera*) population in a small stream was noted by Plummer et al., (2008) following the excavation of that area. That population recovered within four years (Plummer and Mill 2008). They postulated that having areas (up or down stream) to escape construction activities was important in avoiding a population impact from construction. Eskew et al., (2010) also found pre-impact population levels four years following disturbance to ponds inhabited by painted turtles (*Chrysemys pictato*) however, they did cite literature that pointed to potential long term declines following similar disturbance. Review of information in Chen and Leu (2009) indicates a population decrease of approximately 14 percent following excavation and concrete lining. Therefore, we believe that construction activities could result in the death of approximately 14 percent of the turtles (281 turtles or 0.4 percent of the population) within the channelized area as a result of construction activities. If excavation along the river occurs during the fall when turtles are less active the ability of turtles to escape may be reduced resulting in a slightly higher number of turtles being killed, regardless, the Service does not anticipate that a large number of turtles will be killed by excavation activities.

To offset the loss of 31.4 acres of nesting habitat due to excavation and submergence an equal or greater number of sandbars would be recreated in areas identified as having velocities suitable for ringed map turtles during higher flow periods. *Graptemys* and other aquatic turtles have been found to successfully use artificially created nesting habitat (Dobie 1992; Goodwin 2002; Patterson et al., 2013; Seigel et al., 2016). The greatest problem with created nesting habitat is the high predation rate and disturbance by humans. Reducing either or both of these factors would increase nesting success offsetting some project impacts.

The project would also include the creation of islands from approximately RM 289.5 to RM 292.0 in addition to previously mentioned sandbars. These areas would have public access restrictions, placement of snags and no wake zones. The proposed islands and sandbars within the new impoundment would include in their design nesting and basking habitat features for turtles that remain in the excavated portion of the river. Typically, sandy areas within the area encourage beach use by recreational boaters. The FCDCD has indicated that they will have law enforcement authority to restrict access to conservation features and will also use signage to prevent use of sandbars, islands, and sandbanks by the public. Without adequate enforcement of no-human disturbance and vegetative maintenance these features would be ineffective (Godwin 2002). The low nest survival rate due to predation may further reduce the success of created nesting habitat on islands therefore, monitoring of predation rates would be undertaken to determine the need to reduce land based predators (e.g., raccoons, armadillos) and improve hatchling success. To help ensure the nesting and basking areas provided are suitable habitat, areas with higher modeled velocities within the improved channel were identified and targeted for the creation of those habitats. Basking habitat would be recreated through the placement of trees, root wads and crowns adjacent to the sandbars. No-wake zones would be established to reduce disturbance to basking turtles and shelfing of nesting sites. Because the proposed location of some of the sandbars are in areas that would expose turtles to disturbances, such as road noise, the degree of success cannot be fully estimated however currently ringed map turtles are found near highway crossings in the area (Selman and Smith 2017). Locating sand bars adjacent to highway ROWs could reduce the potential recreational boat usage of such sites and adjacent basking habitat and aid in the enforcement of no public access. There is insufficient information for us to estimate the positive benefits of no wake zones on nesting habitat, turtle health and reduced direct mortality from boat strikes, even though literature recommends this measure to reduce all of those factors. Constructing approximately 20 acres of sand bars on the islands and implementation of predator controls to limit predation to 73 percent produces an estimated 357 additional hatchlings to the population each year.

During the year that the river banks would be excavated, sand bars in the area would be surveyed every two day at the start of the nesting season. Any nests found would be relocated north of the excavated area (specific locations would be coordinated with the Service and MSDFWP). Relocated nests would have predator guards placed over them and would be monitored for nesting success. Typically, turtle nests with predator guards have a higher chance of hatching with the percent of successful nests varying from 78 to 100 percent (Horne et al., 2003; Jones 2006). It is estimated that the 1,177 potential nests on the 31.4 acres of sand bar in the excavation area could have approximately 4,236 eggs (3.6 eggs per nest). If approximately half of those nest are found prior to predation and are successfully transferred approximately 2,118 eggs could potentially hatch. Predation by insects could further reduce that number to 1,609 hatchlings. This represents an increase above the determined predation rates by Jones (2006) that estimates those nests would produce only about 451 hatchlings. Selman (2018) postulated that the higher number of juveniles found in the channelized section that will be excavated results from a higher hatching success rate. The success of relocated nests has been documented (Burke et al., 1998; Kornaraki 2006; McElroy 2006; Wyneken 1988) with higher nesting success rates at times resulting from the relocation to better nesting sites (e.g., farther from possible flooding, etc.). Mortality resulting from moving eggs has been documented but the reported best times during incubation to move eggs has yet to be defined (Ahles 2009; Bonach et al., 2003; McElroy 2006).

Approximately 10 miles of river bank would also be preserved and protected through either fee-title purchase or restrictive easements assigned to the land. Such restrictions would prevent the development of habitat adjacent to river thus providing a barrier against disturbance and loss of habitat. This action and implementation of no public access and no wake zones would aid in ensuring greater nesting success and an increase in less disturbed basking periods which can help maintain the health of the turtles (Heppard and Buchholz 2019). Based on information presented in Jones (2006) and Selman (2018) we determined that there are on average approximately 2 sand bars per river kilometer. If 5 miles of both sides of the river are purchased (total of 10 miles of riverbank) then approximately 13 sand bars would be protected. Jones (2006) documented the size of 11 sand bars in approximately 4 river kilometers. Sand bars varied in size from 38 to 9085 square meters and averaged 2,486 square meters. If public access conditions are enforced we estimate that the disturbance to nesting attempts measured by Horne et al. (2003) might be reduced by half. This would result in one average sized sand bar producing an additional 8 hatchlings per year (based on Jones 2006 equation relating sand bar size to number of nests multiplied by the average clutch size). If protection could be applied to 13 sand bars an additional 99 hatchlings would be produced each year based on all sand bars being average in size. Applying the same methodology to the sand bars presented in Jones 2006 it is estimated that approximately 1,226 more hatchlings could be produced per year.

The influence of increased hatchlings survival on the perpetuation of a population has been investigated; hatchling and juvenile mortality is often high enough that reduction of adult mortality is believed to be a better option to sustain the species. However, in populations with little recruitment increasing the survival of hatchlings (Knoerr 2018) and adults (Heppell et al. 1996; Spencer 2017) is viewed as being a better means to ensure survival.

Elimination of basking habitat, disturbance during basking by construction activities and the reduction of food sources due to increased turbidity and removal of structure can result in the decreased health of turtles (Chen and Lue 2009; Heppard and Buchholz 2019). All turtles in the Action Area (2,196) would likely experience these effects especially during the final construction phase. As turtles move from the construction area into areas already inhabited, the potential for crowding with concurrent increased stress and competition for food and habitat could affect their health, survival and reproduction (Chen and Lue 2009). These effects would be felt by all turtles in the Action Area (2,196) as well as those in the Lakeland population (1,556) and those downstream of the weir (1,164). Currently, of the 9.5 miles of river bank in the excavated area, over approximately 4 miles (approximately 40%) have no or limited wooded bank line, thus a portion of the population within the excavated area is persisting in an area with little riparian buffer. The placement and maintenance of basking habitat would offset the loss off existing basking habitat.

The excavated material will be used to upgrade existing levees in the Action Area as well as used to create new levees and 971 acres of elevated fill for future economic development and parks. This will result in the removal of any forested riparian habitat which is the main source of basking and feeding habitat and escape cover used by *Graptemys* (Lechowicz 2013; Lindeman 1997; Lindeman 1998; Lindeman 1999; Killebrew et al., 2002). Reforestation of the lake perimeter is not planned so naturally occurring basking and feeding habitat will largely be eliminated and not replaced. The loss of this habitat would be reflected in the decreased health, survival and reproduction; these adverse effects would be felt by all turtles that remain or return to the channelized area following construction. However, these adverse effects should not result in the lethal take of any turtle and the project includes placement and maintenance of basking and foraging material at the created sandbars thus reducing the impacts from the loss of those areas that have this habitat.

Relocation of the levee near Cypress Lake is expected to disturb those turtles living in that lake, however, trapping and relocation of an estimated 20 of the 36 turtles back into the Pearl River in the northern part of the Lakeland area is planned prior to construction to reduce the potential for direct mortality as these turtles cannot move up or downstream to avoid construction activities.. We expect the remaining turtles to be able to avoid the construction area and not be directly harmed by the activity.

Relocation of turtles, especially aquatic turtles has had varied success (Attum et al., 2013; Attum and Cutshall 2015; Bogossian 2010; Sealy 1976). Soft releases (i.e., including a period of acclimation prior to full release) of turtles has reduce the movement of turtles away from the point of release (Attum and Cutshall 2015); some increased success has also been noted in the relocation events that occur prior to estivation and with greater distances moved. Differential movement between the sexes of mature adults has been noted but relocated juveniles tend to move less than relocated adults. Time till return has taken up to three years (Sealy 1976). Because the relocation site would be separated from the capture site by a levee, the return to that site is improbable, thus increasing the chance of successful relocation, but dispersal from the relocation area may occur. The potential to capture and release individuals from areas where they would never contribute to the population and possibly be affected by construction and relocate them so that they may contribute to the population is the goal of this action. Tracking of the released turtles would aid in the knowledge needed to ensure the continued survival of the species. It is estimated that no more than 20 turtles will be captured and translocated. While a positive conservation action this would result in the harassment of approximately 20 turtles.

An adaptive management and monitoring plan will be developed in conjunction with the Service and the Mississippi Department of Wildlife, Fisheries and Parks (MDWFP) which would provide ongoing monitoring, long-term management, and habitat protection benefits for the listed turtle. Based on the number of turtles handled and/or observed by Jones (2017) we anticipate up to 1,600 turtles over 15 years would be taken in the form of harassment due to being trapped, tagged, data collected, tracked, observed, and monitored for population and movement studies.

6.3.2. Effects of Weir Construction and Impoundment on the Ringed Map Turtle

The establishment of a 1,500-acre impoundment from weir construction will result in changes in the velocity and water surface elevation within the project area. Because the weir has been designed to match the current discharge of the river there should not be a significant change in discharge once flows begin overtopping the weir.

The lake conditions of the Ross Barnett Reservoir has precluded the ringed map turtle from persisting once the reservoir was filled. Killebrew et al., (2002) found Cagles map turtle was absent from five impoundments and attributed that absence to the lack of river type habitat including shoreline for nesting (including sandbars), shoreline vegetation for food and shelter (fallen trees or undercut banks exposing roots), and basking structure. Increase turtle abundance in small riverine lakes was attributed to the relatively unaltered shoreline, lack of development along the shoreline (any development was not close to the shoreline), and the small size of the lakes (a few hundred yards in length). Lakes small enough to still maintain lotic conditions were observed to have a greater abundance of turtles if they also possessed the previously mentioned habitat characteristics (Killebrew 2002). A decrease in Cagles map turtle populations occurred after repairs to a dam that was no longer retaining flows but was followed by an eventual increase once sandbars, riparian habitat, and snag habitat returned. However, if these habitat features did not return and/or if shoreline development occurred the turtle populations did not fully recover or were extirpated. Linderman (1998) stated that habitat characteristics, (deadwood and current) and shoreline development could explain the difference in *Graptemys* abundance in reservoirs. Sealy (1976) stated that the Alabama map turtle (*Graptemys pluchra*) could persist within a lake type environment but stated its degree of success in a lentic versus a lotic habitat has not been determined. Selman and Qualls (2009) did not observe any *Graptemys* in non-flowing lake like conditions created by gravel mines in the Pascagoula River. Selmans's (2018) survey of lakes within the Action Area determined that turtles were present but the populations were predominated by males and only one juvenile was observed. He characterized both Mayes Lakes as being ecological sinks with their populations being supported only by immigration and that of Crystal Lake as not having a long-term viability. Killebrew et al., (2002) found that Cagle's map turtle would be found between approximately 0.6 and 2.6 fps with an optimum velocity around 2.5 fps. Examination of modeled mean cross sectional velocities estimated at an average of approximately every 1,100 feet for the river miles where the Lakeland population is found indicates that the turtle is found in velocities between 0.4 and 2.3 fps. We hypothesize that this range represents the suitable velocities for the ringed map turtle. Because those velocities are mean cross-sectional velocities the actual suitable velocities may vary from those values, however, since all velocities for the project are mean cross-sectional values their application to the impact assessment is appropriate.

Flows of 20,000 cfs with the project constructed would experience velocities within the 0.4 to 2.3 fps range over approximately 83 percent of the range where they would be experienced without the project. At 40,000 cfs there would be an increase in the area experiencing those same velocities. However, once discharges decrease below 10,000 cfs the improved channel's velocities would significantly decrease (<0.4 fps) and lake like conditions would occur. Average monthly flows exceeding 10,000 cfs occur less than 13 percent of the time with that velocity rarely being exceeded from June through November or about half the year (Table 6.3). While this is the normal discharge pattern, the improved channel would experience average cross-sectional velocities that would not be within 0.4 to 2.3 fps during that time. At 10,000 cfs with the project constructed suitable velocities would be found in approximately 92.7 percent of the channel that normally would have those velocities. As discharges decrease the amount of area having suitable velocities would also decline; at 1,000, and 2,000 cfs there would no longer be any area having suitable velocities when prior to the project approximately 87 percent of the area would have had suitable velocities. At 5,000 cfs with the project constructed there would only be approximately 6 percent of the area within the estimated suitable velocities. Mean monthly velocities in the 1,000 cfs range typically occur from July through October. Velocities associated with those conditions will be similar to conditions found at the Ross Barnett Reservoir, where generalist turtle species such as the red-eared slider, common musk turtle, and common snapping turtle increased while specialist riverine turtle species such as the ringed map turtle decreased.

However, between RM 293 and 294 (approximately 0.2 percent of the species range) there would be a significant increase in velocities (>5 fps) that would make these portions of the river less favorable for the *Graptemys* resulting in an additional loss of suitable habitat during normal flow conditions (Killebrew et al., 2002). At higher discharge events (equal to or greater than a five year event) there would be a decrease in velocities which would be allow this area to temporally provide habitat to the turtle on an infrequent basis.

While velocity is not the only habitat factor determining *Graptemys* use of lake like areas, it has been identified as an important one. Its importance is often linked to the need for erosional forces that create tree falls and sandbars. While almost all of the channelized area would not experience those type of velocities at discharges less than 5,000 cfs the creation and maintenance of these habitats would offset the need for velocities to create such habitat. Other habitat characteristics identified as important to the persistence of *Graptemys* within reservoirs include a riparian zone and little lakeside development. The riparian zone will be almost eliminated and development is planned for most of the 971 acres of fill surrounding the improved channel.

Placement of trees as basking habitat would reduce one of the needs for a riparian zone to provide fallen trees for basking and shelter. Velocities used in the Service's analysis are means for the entire cross section of the river. Because the velocities are averages there will be areas throughout the area that will be faster and slower than those presented. Proposed monitoring of the turtle population and created habitat within the Action Area would aid in determining the effectiveness of the created habitat features.

Turtles downstream of the proposed weir are likely to experience short-term impacts associated with increased sediment/siltation on sandbars and basking material during construction. Effectively controlling downstream sediment run-off, especially during high rain events, will be very difficult. However, once sediment runoff issues have dissipated due to high streamflow events, we expect habitat immediately downstream of the weir to remain suitable for the ringed map turtle. We would expect such effects to last less than two years after project completion. Once construction is complete and water pools behind the weir, the mean water depth will increase from approximately 6.7 feet to approximately 21 feet, approximately 14 feet above the existing water surface. If this occurs during nesting season (May to October) it could flood existing nests reducing recruitment from that year's nests. However, filling of the area would likely occur during the higher flow periods, December through May, thus avoiding their nesting time. If filling took place in May it could impact approximately 40 percent of the nests. Details of how the filling will be undertaken have not been finalized but would be coordinated with the Service.

Santucci et al. (2005) studied the impacts of weirs to macroinvertebrates and discovered that species distribution was truncated. Free-flowing river reaches supported a higher quality macroinvertebrate community while pool communities consisted of relatively few taxa dominated by oligochaetes and chrinomid larvae that are more tolerant of poorer water quality. Gangloff (2011) observed that mussel populations upstream of dams had a greater number of historical mussel species. Conversely, Dean et al. (2002) found fewer species but similar abundance upstream and within the influence of the weir resulting from deeper water, slower velocity and silty substrates. Potential upstream impacts to mussels and fish could also result due to changes in tributary velocities upstream of the pool (Roghair et al., 2016). The response of mussels to weirs varies according to individual species tolerance to changes resulting from the weir, including changes in sedimentation rates, suspended sediments, and water quality (Early 2006; Tiemann et al., 2016). It is reasonable to assume that the proposed pool would experience similar changes in macroinvertebrate and mussel communities. Recolonization rates within the channel improvement area would likely occur quickly for invertebrates that could drift downstream and those that disperse aerially. Invertebrates that do not easily disperse (e.g., snail and mussels) would require a longer time period until they fully recolonized the area. Until recolonization is complete the competition for food resources within the channelized area would impact all ringed map turtles within the impoundment.

Cummings (2004) examination of low head dams determined that the biggest issue is anthropogenic influences impacting water quality within the created water body including temperature. Butts and Evans (1978) found that channel dams resulted in lower dissolved oxygen (DO) levels within the pool and the downstream design of the weir influenced the amount of oxygen reintroduced to the water column. Ramped weirs had less re-aeration than water falling over vertical weirs but the greatest influences on DO levels were the water velocity over the dam and the distance water fell. The proposed weir is a vertical weir. Data within the study displays that DO levels within the pool may exhibit wider DO fluctuations typically associated with ponds. Helms et al. (2011) found no physiochemical changes associated with mill dams, and Smith et al. (2017) found that dams did not impact local abiotic factors. Gangloff et al. (2011) found that streams with weirs had lower nitrogen concentrations but observed few statistical differences between habitat variables measured in streams with intact, breached, and relict low-head dams. Santucci et al. (2005) observed that DO and pH levels in pools experienced wide daily fluctuations and at times did not meet state water quality standards. Within the proposed channel improvement area there are eight streams draining approximately 61 square miles of predominately urban areas. Drainage from urban areas typically has increased nutrient loadings and concentrations of pesticides, herbicides, and various hydrocarbon products. High nutrient levels could result in eutrophication of the proposed waterbody. Fluctuations and stratifications in the water quality (e.g., DO) similar to what occurs in the Ross Barnett Reservoir (larger but similar in depth) could be expected. Killebrew et al., (2002) found that even though areas downstream of development were meeting water quality standards there were decreases or localized extirpation of Cagles map turtle. Selman and Jones (2017) cited reports that indicated that prior to improved water quality standards local populations near developed areas were extirpated. Jones and Hartfield (1999) also cited a study that found decreased turtle body size downstream of Jackson. This was attributed to poorer and/or reduced food sources because of decreased water quality and the potential influence of contaminants. Modeling of the project area indicates that water quality should not significantly decline and ringed map turtles are currently persisting in the area with the ongoing discharges. Therefore we believe that while some water quality changes may occur they would not have an adverse effect.

The fish -passage channel would provide approximately 1 mile (0.2 percent of the species range) of flowing water during low flow periods when the channelized area would experience low velocities. Depending on the width and velocities of this feature it could provide additional habitat for the ringed map turtle and would prevent isolation of the populations up and down stream of the weir.

Sediment transport modeling indicates there would be some loss of sediment within the improved channel. Literature regarding the impact of weirs on sediment transport supports that analysis. The loss of sediment will not be comparable to that experienced with large dams but could result in some instability within a limited area downstream of the weir. The Service anticipates approximately 1.6 miles, approximately 0.3 percent of the species range, downstream of the weir would experience some degree of instability that would occur over several years with the capture of small amounts of sediment. Impacts from this would result primarily from an increase in turbidity decreasing potential food sources. The degree of instability and time over which this will occur is unknown but monitoring of this area would be conducted. Eventually, a state of equilibrium would be reached and the impacts would no longer affect the turtle.

Monitoring of the populations within the Action Area would include trapping, tagging and observing, all of which would have some level of disturbance to the turtles. Previous populations studies (i.e., Jones 2017) resulted in the handling of over 1,600 turtles with no known mortality.

6.3.3. Effects of Non-Federal Activities caused by the Federal Action on Ringed Map Turtle

Recreational water sports (e.g., fishing, boating) will likely increase within the improved channel, as well as the Lakeland area, as a result of improved access to the Action Area. This could lead to greater disturbance in basking and nesting behaviors with resulting declines in health and nesting success (Heppard and Buchholz 2019; Selman et al. 2013). Boat wakes can cause shelving of sandbars resulting in turtles nesting in areas closer to the water surface (Selman et al. 2013) which in turn could result in the flooding of turtle eggs and mortality. Because the Service does not know the rate and degree to which recreation will increase, we are unable to estimate the number of nests and individuals potentially impacted. Mortality resulting from boat strikes could also impact the population (Carriere and Blouin-Demers 2010; Selman et al. 2013; Smith et al. 2018). Because larger turtles use deeper water habitats and are typically females, they have an increased potential of being killed, thus reducing their future contribution to the population (Selman et al. 2013; Smith et al. 2018).

No-wake zones would be established around sandbars to reduce the potential impact of both boat strikes and sand bar shelving. No public access would be allowed on the created sandbars thus reducing disturbance to newly created basking habitat and nesting and feeding habitats.

Increased development adjacent to the improved channel could also lead to a decrease in water quality impacting food resources in the improved channel, again the Service is unable to estimate the rate and degree to which this will occur.

Activities that would not occur but for the proposed Federal action include relocation or retrofitting of existing infrastructure within the action area (i.e. roads, bridges, pipelines, powerlines), riverfront access and development. Effects resulting from these activities would include the temporary disturbance to basking, foraging and nesting activities. In addition, temporary and localized increase in turbidity and sedimentation impacts to forage species.

6.3.4. Summary of the Effects of the Action on the Ringed Map Turtle

All the various forms of disturbance (e.g., crowding, displacement) are individually not likely to result in the harm of turtles but collectively they could result in the loss of a portion of the population; this loss is estimated at 1,306 turtles or 2 percent of the entire population across its range. To determine this amount the Service used the mean number of turtles within the two surveyed reaches in the current channelized area and determined what percent of the adjacent population (i.e., more natural areas) they represented. The mid-point between the two average percentages for those two areas was judged to represent a reasonable estimate of the population that could be supported by the proposed channelized area.

The population is expected to undergo an initial decline (from construction mortality) and then a slow decline in the pooled area but would eventually stabilize. Increased survival of adults and hatchlings would occur with the implementation of the above offsetting measures (e.g., no wake zone) resulting in a long-term increase in the population.

6.3.5. Tables and Figures for Effects of the Action on Ringed Map Turtle

Table 6.2. Monthly average discharge (cfs), 1 Standard Deviation (STD) and minimum monthly discharge 1966-2013.

	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec
Average	8333	9303	9101	8183	4312	1562	1154	961	1140	1331	2078	5421
+1 STD	14253	15178	14015	15883	9128	3296	2483	2198	2823	3644	4045	10289
-1 STD	2413	3428	4187	484	-504	-172	-176	-277	-544	-981	111	553
Minimum	338	321	1233	412	256	183	180	197	208	195	142	298

6.4. Cumulative Effects on the Ringed Map Turtle

For purposes of consultation under ESA §7, cumulative effects are those caused by future state, tribal, local, or private actions that are reasonably certain to occur in the Action Area. Future Federal actions that are unrelated to the proposed action are not considered, because they require separate consultation under §7 of the ESA. At this time the Service is unaware of any future state, tribal, local, or private non-Federal actions scheduled to occur in the Action Area. Therefore, cumulative effects are not relevant to formulating our pinion for the Action.

6.5. Conclusion for the Ringed Map Turtle

In this section, we summarize and interpret the findings of the previous sections for the ringed map turtle (status, baseline, effects, and cumulative effects) relative to the purpose of a BO under §7(a)(2) of the ESA, which is to determine whether a Federal action is likely to:
a) jeopardize the continued existence of species listed as endangered or threatened; or
b) result in the destruction or adverse modification of designated critical habitat.
"*Jeopardize the continued existence*" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02).

The proposed project would affect approximately 19.4 miles of the Pearl River from RM 301.77 to RM 282.4 (i.e., the Action Area) resulting in increased stress to all turtles (approximately 4,960 individuals [7 percent of the total population]) within the Action Area because of a decrease in food sources, basking habitat, and nesting habitat, which in turn increases competition for those resources. Of that 19.4 miles, approximately 9.5 river miles (roughly 2 percent of the species' range) of ringed map turtle habitat would be altered from lotic to lentic habitat for approximately 6 months each year as a result of channel modifications and installation of the weir. Construction of the project and the above habitat alterations would decrease water velocities and temporarily increase turbidity, and would affect turtles as follows:

- a temporary loss of food, basking habitat, and nesting habitat for all turtles (approximately 2,010 individuals) remaining in the channelized area until the pool area is flooded and newly created habitat becomes available;

- a temporary decrease in food resources for approximately 3,360 turtles (roughly 5 percent of the total population) as a result of increased turbidity within and downstream of the construction area; and,
- a direct loss of approximately 281 turtles (roughly 0.4 percent of the total population) due to construction.

To offset or reduce direct losses of turtles due to construction, up to 2,018 eggs would be relocated outside the construction area and protected from predators and 20 individuals (0.03 percent of the total population) would be relocated from Cypress Lake to the Pearl River. In addition, up to 1,600 individuals (1 percent of the total population) would be trapped, tagged, data collected, tracked, observed, and monitored in the Action Area population. While these activities are a form of harassment, no turtles are expected to die from such activities.

In summary, all the various stressors and forms of disturbance, considered separately, are not likely to result in the harm of turtles. However, considered collectively, the combined level of stressors and disturbances could result in the loss of a portion of the population due to harm, estimated at 1,306 turtles. As mentioned above, we also estimate the death of approximately 281 turtles directly from construction activities. Thus, the total estimated take of turtles is 1,588 individuals (approximately 2 percent of the total population).

Additional offsets to turtle losses that would be implemented as part of the Action include: (1) the creation and protection of 31.4 acres of nesting habitat (estimated to produce at least 1,176 nests) and adjacent basking habitat and predator control; (2) the establishment and enforcement of no-wake zones to reduce boat strikes and disturbance during basking; (3) the placement of public access conditions to reduce disturbances to basking and nesting behaviors and habitats (4) the creation of an approximately 1 mile fish by-pass, and (5) the protection of 10 miles of riverbank that would prevent the development and destruction of riparian habitat utilized by the turtle and also reduce nesting and basking disturbances. In total the above offsetting measures have the potential to contribute approximately 2,118 hatchlings following construction (from the relocation of eggs) and 474 hatchlings per year thereafter. Provided that the USACE fully implements those conservation features, the Action is not likely to appreciably reduce the likelihood of the survival and recovery of the ringed map turtle.

After reviewing the current status of the ringed map turtle, the environmental baseline for the Action Area, and the effects of the Action (both detrimental and beneficial activities proposed), it is the Service's biological opinion that implementation of the Action is not likely to jeopardize the continued existence of the ringed map turtle. No critical habitat has been designated for this species; therefore, none will be affected.

7. INCIDENTAL TAKE STATEMENT

ESA §9(a)(1) and regulations issued under §4(d) prohibit the take of endangered and threatened fish and wildlife species without special exemption. The term "take" in the ESA means "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (ESA §3). In regulations at 50 CFR §17.3, the Service further defines:

- "harass" as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering;"
- "harm" as "an act which actually kills or injures wildlife. Such act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding or sheltering;" and
- "incidental take" as "any taking otherwise prohibited, if such taking is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity."

Under the terms of ESA §7(b)(4) and §7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered prohibited, provided that such taking is in compliance with the terms and conditions of an incidental take statement (ITS).

The Action considered in this BO includes a conservation measure to relocate turtles from Crystal Lake within the construction area to the Lakeland population area and monitor the population in the Action Area through the sampling, including but not limited to the capturing, tagging, tracking, observing and taking measurements, of individuals. Through this statement, the Service authorizes this conservation measure as an exception to the prohibitions against trapping, capturing, or collecting listed species. This conservation measure is identified as a Reasonable and Prudent Measure below, and we provide Terms and Conditions for its implementation.

For the exemption in ESA §7(o)(2) to apply to the Action considered in this BO, USACE must undertake the non-discretionary measures described in this ITS, and these measures must become binding conditions of any permit, contract, or grant issued for implementing the Action. USACE has a continuing duty to regulate the activity covered by this ITS. The protective coverage of §7(o)(2) may lapse if USACE fails to:

- assume and implement the terms and conditions; or
- require a permittee, contractor, or grantee to adhere to the terms and conditions of the ITS through enforceable terms that are added to the permit, contract, or grant document.

In order to monitor the impact of incidental take, USACE must report the progress of the Action and its impact on the species to the Service as specified in this ITS.

7.1. Amount or Extent of Take

This section specifies the amount or extent of take of listed wildlife species that the Action is reasonably certain to cause, which we estimated in the "Effects of the Action" section(s) of this BO. We reference, but do not repeat, these analyses here.

7.1.1. Gulf Sturgeon

The Service anticipates that the Action is reasonably certain to cause incidental take of Gulf sturgeon consistent with the definition of harm resulting from channel excavation, levee relocation, and construction of the weir and fish passage channel that would result in impoundment of the Pearl River.

The maximum number of fish, over the five year construction period, that is anticipated to be affected to the level of harm is approximately 20 Gulf sturgeon (4.6 percent of the Pearl River population) due to temporary disturbance to foraging during construction and effects to the decrease in water quality of foraging in the impoundment (see Sections 4.3 and 4.5).

Anticipated Take of Gulf Sturgeon

Amount or Extent	Life Stage	Form of Take
20 fish	Juveniles/Adults	Harm

7.1.2. Ringed Map Turtle

The Service anticipates that the Action is reasonably certain to cause incidental take of individual ringed map turtles consistent with the definition of harm resulting from channel excavation and levee relocation (see section 6.3.1).

The following turtle numbers represents the number of turtles affected by each form of non-lethal harm out of the estimated population within the Action Area, 5,108 turtles; these numbers are not additive.

- We anticipate up to 281 turtles (0.4 percent of the population) may be taken in the form of harm as a result of being killed by machinery during construction.
- We anticipate as many as 2,196 turtles (3 percent of the population) found within the construction limits may be temporary harmed due to construction disturbance of basking, foraging, and nesting activities and fleeing during construction.
- We anticipate as many as 4,366 turtles (6 percent of the population) found within the Action Area may be taken in the form of harm due to the temporary competition for reduced basking, foraging, and nesting habitat as turtles are displaced into other areas.
- We anticipate as many as 3,360 turtles (5 percent of population) found in the improved channel and downstream to be harmed due to reduced forage because of temporary increased turbidity and sedimentation.
- We anticipate that collectively the various forms of harassment (e.g., crowding, displacement) are individually not likely to result in the harm of turtles but collectively they could result in the loss of a portion of the population; this loss is estimated at 1,306 turtles or 2 percent of the entire population.
- We anticipate up to 20 turtles would be taken in the form of harassment due to trapping and relocation from Crystal Lake into the Pearl River.
- We estimate that approximately half of the 1,177 potential nests on the 31.4 acres of sand bar in the excavation area are successfully transferred resulting in the relocation of approximately 2,118 eggs.
- We anticipate harming approximately 1,600 turtles through the trapping, tagging collecting data, tracking, observing and monitoring in the Action Area population

The Service anticipates that the Action is reasonably certain to cause incidental take of individual ringed map turtle consistent with the definition of harass resulting from weir construction and impoundment (see section 6.3.2).

- We anticipate harm from the temporary loss of 9.5 miles of riverine habitat (2 percent of the total range) as velocities would fall below those viewed as suitable for 6 months of the year.
- We anticipate harm due to the approximately 1.6 miles of habitat (0.3 percent of the species range) that would experience instability from loss of sediment transport resulting in increased sedimentation.
- We anticipate harm from the loss of 1 mile of riverine habitat (0.2 percent of the total range) as velocities would exceed those viewed as suitable for most of the year.
- We anticipate up to 1,600 turtles over 15 years would be taken in the form of harassment due to trapping, tagging, tracking and observing for population and movement studies.

Anticipated Take of Ringed Map Turtle

Adverse Action and Associated Take	Amount or Extent*	Life Stage	Form of Take
Trapping, tagging, tracking, and observing; temporary disturbance and stress	1,600 individuals	Adults and juveniles	Harass
Trapping and relocation; temporary disturbance and stress	20 individuals	Adults and juveniles	Harass
Construction; temporary disturbance of basking, foraging, and nesting activities and fleeing	2,196 individuals	Adults and juveniles	Harm
Construction; temporary competition for reduced basking, foraging, and nesting habitat	4,366 individuals	Adults and juveniles	Harm
Construction causing a temporary increased turbidity and sedimentation; decreased forage	3,360 individuals	Adults and juveniles	Harm
Construction machinery impacts; mortality	281 individuals	Adults and juveniles	Harm
Displacement, competition, stress, and reduced habitat quality; mortality	1,306 individuals	Adults and juveniles	Harm

*Numbers for harm do not represent cumulative numbers but portions of the Action Area population impacted by multiple stressor.

7.2. Reasonable and Prudent Measures

The Service believes the following reasonable and prudent measures (RPMs) are necessary or appropriate to minimize the impact of incidental take caused by the Action on listed wildlife species. RPMs are described for each listed wildlife species in the subsections below.

7.2.1. Gulf Sturgeon

RPM 1. The USACE will coordinate with the Service to ensure that completed project plans and updates to specific erosion control and off-site stormwater compensation are implemented and include comprehensive monitoring and reporting.

RPM 2. The USACE will coordinate with the Service on a monitoring and adaptive management plan for the fish passage channel to assess the use of the structure by Gulf sturgeon and other aquatic species.

RPM 3. Water quality assessment plans would be coordinated with the Service.

RPM 4: Ensure that the terms and conditions are accomplished and completed as detailed in this incidental take statement including the completion of reporting requirements.

7.2.2. Ringed Map Turtle

RPM 1 – The USACE will coordinate with the Service on the acquisition, protection, or restoration of riverine habitat for ringed map turtle.

RPM 2 – The USACE will coordinate with the Service on a plan to reduce disturbances and predation in recreated nesting and basking areas.

RPM 3 – The USACE will coordinate with the Service on a filling plan to reduce impacts to nesting areas.

RPM 3 – The USACE will coordinate with the Service on the development of a capture, relocation and monitoring plan for ringed map turtles in Crystal Lake.

RPM 4 – The USACE will coordinate with the Service on the development of a survey and nest relocation plan.

RPM 5 – The USACE shall ensure that all appropriate Project personnel (*e.g.*, inspectors, contractors, equipment operators) are fully aware of the reasonable and prudent measures and the terms and conditions in this ITS, the conservation recommendations which follow this ITS, and of the protection afforded the ringed map turtle under the Endangered Species Act.

RPM 6 – Work with the USACE to determine the feasibility of reforesting the top bank of the fish passage.

RPM 7 – The USACE will develop a plan to reduce take associated with erosion control measures and excavation activities.

RPM 8 – See RPM 1 for the Gulf sturgeon.

RPM 9 – See RPM 3 for the Gulf sturgeon.

7.3. Terms and Conditions

In order for the exemption from the take prohibitions of §9(a)(1) and of regulations issued under §4(d) of the ESA to apply to the Action, the USACE must comply with the terms and conditions (T&Cs) of this statement, provided below, which carry out the RPMs described in the previous section. These T&Cs are mandatory. As necessary and appropriate to fulfill this responsibility, the USACE must require any permittee, contractor, or grantee to implement these T&Cs through enforceable terms that are added to the permit, contract, or grant document.

7.3.1. Gulf Sturgeon

T&C 1. RPM 1. A MDEQ approved erosion and sediment control plan will be submitted and reviewed by the Service prior to start of construction to assure that potential impacts to Gulf sturgeon habitat from sedimentation and turbidity are avoided and minimized to the extent practicable. The Service will be contacted immediately if failures occur in erosion and sediment control measures occur.

T&C 2. RPM 2. Monitoring of the area where the weir and fish passage channel would be constructed pre- and post–construction for usage by aquatic species, in particular the Gulf sturgeon and ringed map turtle.

T&C 3. RPM 2. Annual post-construction water velocity monitoring would be conducted in and around the approaches of the fish passage channel. This assessment would be to evaluate if the velocities exceed swim speed of Gulf sturgeon and submitted at year 1, 2, 4, 6, 8, and 10.

T&C 4. RPM 2. An adaptive management plan would be provided to the Service for the fish passage channel in the event that monitoring of the passage shows that it is not functioning in the manner it was designed to function.

T&C 5. RPM 3. Basic water quality monitoring would be conducted in the project area and downstream of the weir to assess the temperature, turbidity, dissolved oxygen levels, and water velocities, and will be submitted at years 1, 2, 4, 6, 8, and 10.

T&C 6. RPM 4. Upon locating a dead, injured, or sick individual of an endangered or threatened species, notification must be made to the Fish and Wildlife Service Law Enforcement Office, Jackson, Mississippi at (601) 965-4699 within 24 hours. Additional notification to the Fish and Wildlife Service's Field Office at Jackson, Mississippi at (601) 965-4900 within 48 hours will be provided by the USACE. Care should be taken in handling sick or injured individuals and in the preservation of specimens in the best possible state for later analysis of cause of death or injury.

T&C 7. RPM 4. A report describing the actions taken to implement the terms and conditions of this ITS shall be submitted to the Project Leader, U.S. Fish and Wildlife Service, 6578 Dogwood View Parkway, Suite A, Jackson, MS 39213-7856, within 60 days of the completion of the project. This report shall include the dates of work, assessment, and actions taken to address impacts to the ringed map turtle and the Gulf sturgeon, if they occurred.

7.3.2. Ringed Map Turtle

T&C 1. RPM 1. A proposed land acquisition and management plan will be submitted to the Jackson Mississippi Ecological Services Office before construction begins outlining areas to be protected for ringed map turtles, how land will be restored if required, identifying potential threats to turtle habitat and how such threats will be controlled (i.e. public use, predator control, wake zones, etc.), who will oversee land management actions, and how lands will be managed in perpetuity. A minimum of 10 river miles would be protected. Land acquisition will be prioritized accordingly:

- Priority 1 – Protect via fee title or conservation easement or similar encumbrance privately held lands adjacent to the Pearl River in the upstream portion of the Action Area.
- Priority 2 – Protect via fee title or conservation easement or similar encumbrance riverbank (both sides) in that portion of the turtles range north of the Ross Barnett Reservoir.
- Priority 3 - Protect via fee title or conservation easement or similar encumbrance riverbank (both sides) south of the weir.

T&C 2. RPM 2. Monitoring nesting success per Jones 2017; if predation on islands exceeds 73 percent develop in coordination with the Service a plan to reduce predation. Sufficiently mark no wake and no public access areas to ensure compliance, note such areas on project maps, kiosks and pamphlets of the area. Monthly enforcement reporting (number of visits, verbal warnings, and citations) on restricted areas would be provided to the Service.

T&C 3. RPM 3. Develop a filling plan in coordination with the Service that would reduce the chance of flooding during nesting season.

T&C 4. RPM 4. Capture per Jones 2017 ringed map turtles from Crystal Lake prior to construction. PIT and telemetry tag turtles and track for 3 years to further define habitats used and movements throughout the year

T&C 5. RPM 5. Sandbar surveys inside of the planned construction area(s) every 2-days during the nesting season (May 1 – October 3). Surveyed areas would extend 110 feet from the top of the Pearl River bank. The purpose of monitoring during construction is to locate newly formed nests within the construction area(s) and relocate them to sandbars outside the construction area (e.g., Lakeland population) within 36-hours of eggs being laid which will significantly reduce the likelihood of take and predation. Predator guards will be placed on the nests and the nests will be monitored.

T&C 6. RPM 6 and RPM 7. Workers will be given information identifying ringed map turtles and stating the need to avoid injury or death to the turtles, their protected status under the ESA, and contact information for personnel that would respond to any turtles imperiled.

T&C 7. RPM 7. During detailed planning determine the feasibility of replanting trees on the top bank of the fish passage to improve conditions for ringed back turtles. If feasible such restoration would be implemented.

T&C 8. RPM 7. Upon locating a dead, injured, or sick individual of an endangered or threatened species, notification must be made to the Fish and Wildlife Service Law Enforcement Office, Jackson, Mississippi at (601) 965-4699 within 24 hours. Additional notification to the Fish and Wildlife Service's Field Office at Jackson, Mississippi at (601) 965-4900 within 48 hours will be provided by the U.S. Army Corps of Engineers. Care should be taken in handling sick or injured individuals and in the preservation of specimens in the best possible state for later analysis of cause of death or injury.

T&C 9. RPM 7. A report describing the actions taken to implement the terms and conditions of this incidental take statement shall be submitted to the Project Leader, U.S. Fish and Wildlife Service, 6578 Dogwood View Parkway, Suite A, Jackson, MS 39213-7856, within 60 days of the completion of the project. This report shall include the dates of work, assessment, and actions taken to address impacts to the ringed map turtle and the Gulf sturgeon, if they occurred.

T&C 10. RPM 8. Basic water quality monitoring would be conducted in the project area and downstream of the weir to assess the temperature, turbidity, dissolved oxygen levels, water velocities, pH, conductivity, redox, turbidity, nitrates, phosphorous, and chlorophyll will be submitted at years 1, 2, 4, 6, 8, and 10.

7.4. Monitoring and Reporting Requirements

In order to monitor the impacts of incidental take, USACE must report the progress of the Action and its impact on the species to the Service as specified in the ITS (50 CFR §402.14(i)(3)). This section provides the specific instructions for such monitoring and reporting (M&R). As necessary and appropriate to fulfill this responsibility, USACE must require any permittee, contractor, or grantee to accomplish the monitoring and reporting through enforceable terms that are added to the permit, contract, or grant document. Such enforceable terms must include a requirement to immediately notify USACE and the Service if the amount or extent of incidental take specified in this ITS is exceeded during Action implementation.

M&R 1- The USACE will conduct a river morphology monitoring plan for the area upstream of pool to RM 295 and 1.6 miles downstream of weir and submit to the Service.

M&R 2- A report will be submitted once the construction phase is finalized which will include:

- Amount of sand bar habitat created (number, acreage and dimensions of each),

- How sand bars will be maintained (i.e. vegetation monitoring and management),
- How sand bars will be protected (i.e. public use, predator control, wake zones), and
- Amount of basking material remaining, added and location and maintenance plan.

M&R 2- In coordination with the Service and MDFWP develop a monitoring and analysis plan per techniques in Jones 2017 and telemetry for the Channel Improvement Area, Lakeland Population, translocation area and downstream of weir; years 1, 2 and 3 post construction and then every 5 years for 15 years and determine turtle movements

8. CONSERVATION RECOMMENDATIONS

§7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by conducting conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary activities that an action agency may undertake to avoid or minimize the adverse effects of a proposed action, implement recovery plans, or develop information that is useful for the conservation of listed species. The Service offers the following recommendations that are relevant to the listed species addressed in this BO and that we believe are consistent with the authorities of the USACE.

1) Support the future monitoring research efforts for Gulf sturgeon that will be funded through the NRDA Deepwater Horizon Ocean Open Trustee Implementation Group (TIG) through assisting with the monitoring efforts.
2) Funding or supporting research/monitoring efforts for Gulf sturgeon around the weir and fish passage channel. Place monitoring stations in this area to evaluate whether tagged individuals are migrating through the area.
3) Conduct immediate watershed assessment for future impacts.
4) Examine operation of the low flow gate to help aid the downstream flow of sediment.
5) The FDCD will provide an annual operation log of the low flow gate.
6) The FDCD would work with local governments to restrict water craft access to Hanging Moss Creek, Purple Creek, Eubanks Creek, and Town Creek.
7) FDCD would work with Mississippi Department of Wildlife, Fisheries and Parks to restrict the use of hoop nets near nesting beaches.

9. REINITIATION NOTICE

Formal consultation for the Action considered in this BO is concluded. Reinitiating consultation is required if the USACE retains discretionary involvement or control over the Action (or is authorized by law) when:

a. the amount or extent of incidental take is exceeded;
b. new information reveals that the Action may affect listed species or designated critical habitat in a manner or to an extent not considered in this BO;
c. the Action is modified in a manner that causes effects to listed species or designated critical habitat not considered in this BO; or
d. a new species is listed or critical habitat designated that the Action may affect.

In instances where the amount or extent of incidental take is exceeded, the USACE is required to immediately request a reinitiation of formal consultation.

10. LITERATURE CITED

Ahles, N. 2009. Effects of mid-incubation egg movement on loggerhead turtle (*Caretta caretta*) turtle hatch success and embryo development. Florida Atlantic University, Masters Thesis.

Altinok, I., S.M. Galli, and F.A. Chapman. 1998. Ionic and osmotic regulation capabilities of juvenile Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi.* Comparative Biochemistry and Physiology 120:609-616.

Attum, O. and C. Cutshall. 2015. Movement of translocated turtles according to translocated method and habitat structure. Restoration Ecology, 23(5).

Attum, O., C. Cutshall, K. Eberly, H. Day, and B. Tietjen. 2013. Is there really no place like home? Movement, site fidelity and survival probability of translocated and resident turtles. Biodiversity Conservation 22.

Auer, N.A. 1996. Importance of habitat and migration to sturgeons with emphasis on lake sturgeon. Can J Fish Aquat Sci.

Bednar, G.A. 1980. Quality of Water in the Pearl River, Jackson to Byram, Mississippi, September 21-22, 1976. U.S. Department of Interior, Geological Survey. Jackson, MS.

Berkman, H. E. and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. Environ. Biol. Fishes. 8(4)

Blute, G., M. Carrie and G. Blouin-Demers. 2010. Impact of recreational power boating on two populations of northern map turtles (*Graptemys geographica*). Aquatic Conservation Marine and Freshwater Ecosystems. 20.

Bogosian III, V. 2010. Natural History of Resident and Translocated Alligator Snapping Turtles (*Macrochelys temminckii*) in Louisiana. Southeastern Naturalist 9(4).

Bonach, K., M. Miranda-Vilela, M. Alves, and L. Verdade. 2003. Effect of Translocation on Egg Viability of the Giant Amazon River Turtle, *Podocnemis expansa.* Chelonian Conservation and biology 4(3).

Browne, C. and S. Hecnar. 2007. Species loss and shifting population structure of freshwater turtles despite habitat protection. Biological Conservation 138(3-4). Abstract only.

Bulhmann, K. 2017. Results of Wildlife Camera Trapping on Natural Sandbars and Navy Target Areas and Considerations for Turtle Nest Site Protection, Population Enhancement, and Habitat Management to Benefit Ringed Map Turtles (*Graptemys* *oculifera*) U.S. Navy's Stennis Western Maneuver Area (WMA) East Pearl and Mike's Rivers, Mississippi Final Report.

Burke, R. 2015. Head-starting Turtles: Learning from Experience. Herpetological Conservation and Biology 10(Symposium).

Butts, T.A., and R. L. Evans. 1978. Effects of channel dams on dissolved oxygen concentrations in northeastern Illinois streams. Illinois State Water Survey; Circular 132. ISWS/CIR-132/78.

Caro, T. 2010. Conservation by Proxy: Indicator, Umbrella, Keystone, Flagship, and Other Surrogate Species. Island Press, Washington D.C.

Carr, A. 1983. All the way down upon the Suwannee River. Audubon Magazine.

Carr, S.H, F. Tatman, and F.A. Chapman. 1996. Observations on the natural history of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*, Vladykov 1955) in the Suwannee River, southeastern United States. Ecology of Freshwater Fisheries.

Cech, J.J. Jr., S.J. Mitchell and T.E. Wragg. 1984. Comparative growth of juvenile white sturgeon and striped bass: effects of temperature and hypoxia. Estuaries.

Chapman, F.A., S.F. O'Keefe, and D.E. Campton. 1993. Establishment of parameters critical for the culture and commercialization of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi.* Fisheries and Aquatic Sciences Dept., Food Science and Human Nutrition Dept., University of Florida, Gainesville, FL. Project Final Report. NOAA No. NA27FD0066-National Marine Fisheries Service. St. Petersburg, FL.

Cheong, T. S., M. L. Kavvas, and E. K. Anderson. 2006. Evaluation of Adult White Sturgeon Swimming Capabilities and Applications to Fishway Design. Envrion. Biol Fish.

Chen, Tien-Hsi and Kuang-Yang Lue. 2009. Changes in the Population Structure and Diet of the Chinese Stripe-Necked Turtle (*Mauremys sinensis*) Inhabiting a Disturbed River in Northern Taiwan. Zoological Studies 48(1).

Clugston, J.P., Foster, A. M. and S. H. Carr. 1995. Gulf sturgeon, *Acipenser oxyrinchus desotoi* in the Suwannee River, Florida, USA. pp. In: A. D. Gershanovich and T. I. J. Smith (eds.) Proceedings of the International Symposium on Sturgeons, VNIRO Publishing, Moscow.

Craft, N.M., B. Russell, and S. Travis. 2001. Identification of Gulf sturgeon spawning habitats and migratory patterns in the Yellow and Escambia River systems. Final Report to the Florida Marine Research Institute, Fish and Wildlife Conservation Commission.

Csiki, S.J.C. 2014. The impact of run-of-river dams on channel morphology and sedimentation. Master's Dissertation. University of Illinois at Urbana-Champaign, Urbana Illinois.

Csiki, S. and B. Rhoads. Hydraulic and geomorphological effects of run-of-river dams. Progress in Physical Geography, 34(6)

Crider, A.F. 1906. Geology and mineral resources of Mississippi. U.S. Department of Interior, U.S. Geological Survey.

Cummings, G.S. 2004. The impact of low-head dams on fish-species richness in Wisconsin, USA. Ecological Applications 14(5). https://www.jstor.org/stable/20204015

Dean, J.D., Eds. D. Gillette, J. Howard, S. Sherraden and J. Tiemann. 2002. Effects of low-head dams on freshwater mussels in the Nesho River, Kansas. Transactions of the Kansas Academy of Science 106(3-4).

Dobie, J. 1992. Clutch survival of Alabama red-bellied turtles on Gravine Island. Alabama Non-game Program. Project E-1.

Dugo, M.A., B.R. Kreiser, S.T. Ross, W.T. Slack, R.J. Heise, and B.R. Bowen. 2004. Conservation and management implications of fine scale genetic structure of Gulf sturgeon in the Pascagoula River, Mississippi. Journal of Applied Ichthyology.

Early, T.M. 2016. Modeling the variable effects of low-head dams on freshwater mussel assemblages. Honors Thesis Appalachian State University. .

Edwards, R.E., K.J. Sulak, M.T. Randall, and C.B. Grimes. 2003. Movements of Gulf sturgeon *(Acipenser oxyrinchus desotoi)* in nearshore habitat as determined by acoustic telemetry. Gulf of Mexico Science 21(1):59-70.

Edwards, R.E., R.E., Parauka, F.M. and K.J. Sulak. 2007. New insights into marine migration and winter habitat of Gulf sturgeon. in: J. Munro, D. Hatin, J. Hightower, K. Sulak, and F. Caron (eds.). Proceedings of the Symposium on Anadromous Sturgeons. American Fisheries Society, Symposium, Bethesda, Maryland.

Environmental Protection Agency (EPA). 1975. Report on Ross Barnett Reservoir, Jackson, Madison, and Rankin Counties, Mississippi. Region IV, Working Paper 362. .

Eskew, E., S. Price, and M. Dorcas. 2010. Survivorship and Population Densities of Painted Turtles (*Chrysemys picta*) in Recently Modified Suburban Landscapes. Chelonian Conservation and Biology, 9(2) Abstract only.

Fencl, J.S., M.E. Mather, J.M. Smith, and S.M. Hitchman. 2017. The blind men and the elephant examine biodiversity at low-head dams: are we all dealing with the same dam reality? Ecosphere, 8(11). www.esajournals.org.

Flowers, H.J. 2008. Age-structured population model for evaluating Gulf Sturgeon recovery on the Apalachicola River, Florida. M.S. Thesis, University of Florida, 2008.

Flowers, H.J., W.E. Pine III, A.C. Dutterer, K.G. Johnson, J.W. Ziewitz, M.S. Allen, and F.M. Parauka. 2009. Spawning site selection and potential implications of modified flow regimes on viability of Gulf sturgeon populations. Transactions of the American Fisheries Society 138:1266-1284.

Foster, A.M. 1993. Movement of Gulf sturgeon, Acipenser oxyrinchus desotoi in the Suwannee River, Florida. Master Thesis, University of Florida, Gainesville, FL.

Foster, A.M. and J.P. Clugston. 1997. Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2000. Gulf sturgeon spawning migration and habitat in the Choctawhatchee river system. Alabama-Florida. Transactions of the American Fisheries Society129:811-826.

Fox, D.A., J.E. Hightower, and F.M. Parauka. 2002. Estuarine and nearshore marine habitat use by Gulf sturgeon from the Choctawhatchee River system, Florida., Pages 111-126 in W. Van Winkle, P.J. Anders, D.H. Secor, and D.A. Dixon, editors, Biology, protection, and management of North American sturgeon. American Fisheries Society, Symposium 28, Bethesda, Maryland.

Gangloff, M.M., E.E. Hartfield, D.C. Werneke, and J.W. Feminella. 2011. Associations between dams and mollusk assemblages in Alabama streams. Journal of North American Benthological Society 304(4).

Gu, B., D. M. Schell, T. Frazer, M. Hoyer, and F. A. Chapman. 2001. Stable carbon isotope evidence for reduced feeding of Gulf of Mexico sturgeon during their prolonged river residence period. Estuarine, Coastal, and Shelf Science 53:275-280.

Godwin, J. 2002. Turtle Nest Success on Gravine Island with Emphasis on the Alabama Red-bellied Turtle (*Pseudemys a/abamensis*) and Delta Map Turtle (*Graptemys nigrinoda delticola*). Alabama Natural Heritage Programs, The Nature Conservancy

Hasse, C.S. 2006. Hydrologic analysis of the Pearl River between Edinburg, MS, and Bogalusa, LA. The Nature Conservancy, Southern United States Region.

Heard, R. W., J.L. McLelland, and J.M. Foster. 2000. Benthic invertebrate community analysis of Choctawhatchee bay in relation to Gulf sturgeon foraging: an overview of year 1. Interim report to the Florida Fish and Wildlife Conservation Commission, St. Petersburg.

Heise, R.J., S.T. Ross, M.F. Cashner, and W.T. Slack. 1999a. Movement and habitat use of the Gulf sturgeon *(Acipenser oxyrinchus desotoi)* in the Pascagoula drainage of Mississippi: year 3. Museum Technical Report No. 74. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 14.

Heise, R. J., S. T. Ross, M. F. Cashner, and W. T. Slack. 1999b. Gulf sturgeon *(Acipenser oxyrinchus desotoi*) in the Pascagoula Bay and Mississippi Sound. Museum Technical Report No. 76.

Heise, R. J., W. T. Slack, S. T. Ross, and M. A. Dugo. 2004. Spawning and associated movement patterns of Gulf Sturgeon in the Pascagoula River drainage, Mississippi. Transactions of the American Fisheries Society.

Helms, B.S., D.C. Werneke, M.M. Gangloff, E.E. Hartfield, and J.W. Feminella. 2011. The influence of low-head dams on fish assemblages in streams across Alabama. Journal of North American Benthological Society 304(4). http://www.bioone.org/doi/full/10.1899/10-093.1

Henley, W. F., M. A. Patterson, R. J. Neves, and A. D. Lemly. 2000. Effects of sedimentation and turbidity on lotic food webs: A concise review for natural resource managers. Rev. Fish. Sci. 8(2).

Hightower, J.E., K.P. Zehfuss, D.A. Fox, and F.M. Parauka. 2002. Summer habitat use by Gulf sturgeon in the Choctawhatchee River, Florida. Journal of Applied Ichthyology.

Heppard J. and R. Buchholz. 2019. Impact of human disturbance on the thermoregulatory behavior of the endangered ringed sawback turtle (*Graptemys oculifera*). Aquatic Conservation of Marine and Freshwater Ecosystems;1–12.

Heppell, S., L. Crowder and D. Crouse. 1996. Models to evaluate head starting as a management tool for long-lived turtles. Ecological Applications, 6(2).

Horne, B., R. Brauman, M. Moore, and R. Seigel. 2003. Reproductive and Nesting Ecology of the Yellow-Blotched Map Turtle, *Graptemys flavimaculata*: Implications for Conservation and Management. Copeia (4).

Huff, J.A. 1975. Life History of the Gulf of Mexico Sturgeon, *Acipenser oxyrhynchus desotoi* in Suwannee River, Florida. Mar. Res. Publ. No. 16.

Jenkins, W. E.; Smith, T. I. J.; Heyward, L. D.; Knott, D. M., 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum,* juveniles to different salinity and dissolved oxygen concentrations. Proc. Ann. Conf. Southeast. Fish Wildl. Agen.

Jones, R. 2006. Reproduction and Nesting of the Endangered Ringed Map Turtle, *Graptemys oculifera*, in Mississippi. Chelonian Conservation and Biology, 2006, 5(2).

Jones, R. 2017. Long-Term Trends in Ringed Sawback (*Graptemys oculifera*) Growth, Survivorship, Sex Ratios, and Population Sizes in the Pearl River, Mississippi. Chelonian Conservation and Biology, 2017, 16(2)

Jones, R. and P. Hartfield. 1995. Population Size and Growth in the Turtle *Graptemys oculifera*. Journal of Herpetology, 29(3).

Jones, R. and W. Selman. 2009. *Graptemys oculifera* (Baur 1890) - Ringed Map Turtle, Ringed Sawback. Conservation Biology of Freshwater Turtles and Tortoises. *Chelonian Research Monographs, No.* 5.

Kahn, J.; Mohead, M., 2010. A protocol for use of Shortnose, Atlantic, Gulf, and Green sturgeons. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, NOAA Technical Memo No. NMFS-OPR-45. March 2010, Silver Spring.

Kennedy, T.P., and C.S. Hasse. 2009. Geomorphic and sediment assessment of the Pearl River in Mississippi and Louisiana. The Nature Conservancy.

Killebrew, F., W. Rogers and J. Babitzke. 2009. Assessment of Instream Flow and Habitat Requirements for Cagle's Map Turtle (*Graptemys caglei*). Edwards Aquifer Authority Contract #00-52-AS) West Texas A&M University Canyon, Texas

King, T. L., B. A. Lubinski, and A. P. Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (*Acipsenser oxyrhynchus oxyrinchus*) and cross-species amplification in the Acipenseridae. Conservation Genetics.

Kaeser, A. 2019. Hurrican Michael-Investigating Impacts to the Apalachicola River Sturgeon Population. The Wandering Sturgeon: A Newsletter on Anything Gulf sturgeon. March 2019 Special Editor's Issue.

Kishinhi, S., P. B. Tchounwou, I. O. Farah, and P. Chigbu. 2006. Recreational Water Quality Control in Mississippi, USA: Bacteriological Assessment in the Pearl River and Ross Barnett Reservoir. Reviews on Environmental Health.

Kohl, B., 2003. Pearl River Fishway: Part 2. Technical Report on the Proposed Fish Bypas at Poole's Bluff Sill, Pearl River, Louisiana-Mississippi. 17 pp., 24 Figs., 5 Tables, 7 photos.

Knoerr, M. 2018. Hatch success and population modeling for the critically endangered bog turtle in North Carolina. Clemson University, Masters Thesis.

Kornarakia, e., D. Matossiana, A. Mazarisa, Y. Matsinosa, D. Margaritoulisb. 2006. Effectiveness of different conservation measures for loggerhead sea turtle (*Caretta caretta*) nests at Zakynthos Island, Greece. Biological Conservation.

Kynard, B., and E. Parker. 2004. Ontogenetic behavior and migration of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi*, with notes on body color development. Environmental Biology of Fishes.

Landry, K. 2012. Ringed Map Turtle (*Graptemys oculifera*) Surveys on the Pearl River Washington Parish. Louisiana Natural Heritage Program Louisiana Department of Wildlife and Fisheries.

Landry, K. and B. Gregory. 2010. Ringed Map Turtle (*Graptemys oculifera*) and Pascagoula Map Turtle (*Graptemys gibbonsi*) Survey of the Bogue Chitto River in Louisiana, Washington and St. Tammany Parish. Louisiana Natural Heritage Program Louisiana Department of Wildlife and Fisheries

Lang, J.W. 1972. Geohydrologic summary of the Pear River Basin, Mississippi and Louisiana. Geological Survey Water-Supply Paper 1899-M.

Lechowicz II, C. 2013. Aspects of the Population Dynamics of Sympatric Map Turtles, *Graptemys barbouri* and *Graptemys ernsti*, in the Lower Choctawhatchee River System of Alabama and Florida. Florida Gulf Coast University. Masters Thesis.

Lee, F.N. 1985. Analysis of the low-flow characteristics of streams in Louisiana. Water Resources Technical Report No. 35. Published by Louisiana Department of Transportation and Development.

Lindeman, P. 1999. Turtle densities and deadwood in southern river drainages. Herpetological Review; Jun; 30, 2; Natural Science Collection.

Lindeman, P. 1999. Surveys of basking map turtles *Graptemys* spp. In three drainages and the importance of deadwood abundance. Biological Conservation 88.

Lindeman, P. 1998. Of deadwood and Map turtles (*Graptemys*): An analysis of species status for five species in three river drainages using replicated spotting-scope counts of basking turtles. Chelonian Conservation and Biology. 3(1). Linnaeus Fund Research Reports.

Lorio, W. (ed.). 2000. Proceedings of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) status of the subspecies**.** Workshop held September 13-14, 2000, Mississippi State University, Science and Technology Research Center, Stennis Space Center, MS. 200 pp. and appendices.

Marchant, S.R. and M.K. Shutters. 1996. Artificial substrates collect Gulf sturgeon eggs. North American Journal of Fisheries Management.

Mason, W.T. and J.P. Clugston. 1993. Foods of the Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society.

McCoy, C. and R. Vogt. 1988. *Graptemys oculifera*. Catalogue of American Amphibians and Reptiles.

McDowall, R.M. 1988. Diadromy in fishes migrations between freshwater and marine environments. Truder Press and Croom Helm. 308 pp.

McElroy, M. 2006. The effect of screening and relocation on hatching and emergence success of loggerhead sea turtle nests at Sapleo Island Georgia. University of Georgia, Masters Thesis.

Miranda, L. E. and D. C. Jackson. 1987: A status survey of Atlantic sturgeon in the Pascagoula and Pearl River systems of Mississippi. Unpublished report for Mississippi Wildlife Heritage Fund research grant, Mississippi Museum of Natural Science, Museum Technical Report. No. 2, Jackson, MS. 27 pp.

Mississippi Department of Environmental Quality (MDEQ). 2019. MDEQ Issues Water Contact Advisory for Pearl River and Other Streams in the Jackson Area. https://www.mdeq.ms.gov/mdeq-issues-water-contact-advisory-for-pearl-river-and-other-streams-in-the-jackson-area/.

Monroe, W.H. 1954. Geology of the Jackson area, Mississippi. Stratigraphic and structural study of the area surrounding the Jackson gas field. Geological Survey, Bulletin 986.

Morrow, J.V, K.J. Killgore, J.P. Kirk, and H.E. Rogillio. 1996. Distribution and population attributes of Gulf sturgeon in the lower Pearl River System, Louisiana. Proceedings of the Annual C nference of the Southeastern Association of Fish and Wildlife Agencies.

Morrow, J.V, K.J. Killgore, J.P. Kirk, and H.E. Rogillio. 1998. Distribution and population attributes of Gulf sturgeon in the lower Pearl River System, Louisiana. Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Murphy, D. and Weiland, P. 2014. The use of surrogates in implementation of the federal Endangered Species Act—proposed fixes to a proposed rule. J Environ Stud Sci. https://link.springer.com/article/10.1007/s13412-014-0167-y

Odenkirk, J.S. 1989. Movements of Gulf of Mexico sturgeon in the Apalachicola River, Florida. Proceedings of the Annual Conference Southeastern Association of Fish and Wildlife Agencies.

Oswalt, S. 2015. Mississippi's forests, 2013. Resource. Bulletin. SRS–204. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Patterson, J., B. Steinberg, and J. Litzgus. 2012. Not just any old pile of dirt: evaluating the use of artificial nesting mounds as conservation tools for freshwater turtles. Fauna & Flora International, Oryx, Parauka, F. M. and M. Giorgianni. 2002: Availability of Gulf sturgeon spawning habitat in northwest Florida and southeast Alabama River systems. Unpublished technical report by United States Fish and Wildlife Service, Panama City, FL, USA, 77 pp.

Parauka, F.M., S.K. Alam, and D.A. Fox. 2001. Movement and habitat use of subadult Gulf sturgeon in Choctawhatchee Bay, Florida. Proceedings Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Parauka, F.M., W.J. Troxel, F.A. Chapman, and L.G. McBay. 1991. Hormone-induced ovulation and artificial spawning of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi*. Prog. Fish-Culturist.

Parkyn, D.C., D.J. Murie, J.E. Harris, D.E. Colle, and J.D. Holloway. 2007. Seasonal movements of Gulf of Mexico sturgeon in the Suwannee River and estuary. American Fisheries Society Symposium.

Pearson, A.J., J.E. Pizzuto, and R. Vargas. 2016. Influence of run of river dams on floodplain sediments and carbon dynamics. Geoderma 272. www.elsevier.com/locate/geoderma.

Phalen, P.S., R.J. Muncy, and T.K. Cross. 1988. Hybrid striped bass movements and habitat in Ross Barnett Reservoir, Mississipp. Procedures of the Annual Conference of the Southeast Association of Fish and Wildlife Agencies 42.

Piller, K.R., H.L. Bart, Jr., and J.A. Tipton. 2004. Decline of frecklebelly madton in the Pearl River based on contemporary and historical surveys. Transactions of the American Fisheries Society 133.

Pine, W.E., M.S. Allen, and V.J. Dreitz. 2001. Population viability of the Gulf of Mexico sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society.

Pine, W.E., H.J. Flowers, K.G. Johnson, and M.L. Jones. 2006. An assessment of Gulf sturgeon movement, spawning site selection, and post-spawn holding areas in the Apalachicola River, Florida. Final Report submitted to the Florida Fish and Wildlife Conservation Commission. University of Florida, Gainesville, FL.

Plummer, M., D. Krementz, L. Powell and N. Mills. 2008. Effects of Habitat Disturbance on Survival Rates of Softshell Turtles (*Apalone spinifera*) in an Urban Stream. Journal of Herpetology, 42(3).

Plummer, M., and N. Mills. 2008. Structure of an urban population of softshell turtles (*Apalone spinifera*) before and after severe stream alteration. Herpetological Conservation 3.

Porto, L. M., R. L. McLaughlin, and D. L. G. Noakes. 1999. Low-Head Barrier Dams Restrict the Movements of Fishes in Two Lake Ontario Streams. North American

Journal of Fisheries Management.

Pringle, C. M. 1997. Exploring how disturbance is transmitted upstream: going against the flow. North American Benthological Society.

Randall, M.T. and K.J. Sulak. 2012. Evidence of autumn spawning in Suwannee River Gulf sturgeon, *Acipenser oxyrinchus desotoi* (Vladykov, 1955). Journal of Applied Ichthyology.

Reynolds, C.R. 1993. Gulf sturgeon sightings, historic and recent - a summary of public responses. U.S. Fish and Wildlife Service. Panama City, Florida.

Richardson, T. and R. Seigel. No date. Habitat use of the norther map turtle (*Graptemys geographica)* in an altered system, Susquehanna River, Maryland, USA. Towson University.

Roghair, C., J. Moran, S. Adams, W. Haag, M. Warren, C. Krause, and C. Dolloff. 2016. Examination of fish, crayfish, mussels and habitat in transitional reaches upstream of Lewis Smith Reservoir, Alabama.

Rogillio, H. E., Rabalais, E. H., Forester, J. S., Doolittle, C. N., Granger, W. J., and J. P. Kirk. 2001: Status, movement and habitat use of Gulf sturgeon in the Lake Pontchartrain Basin, Louisiana. Unpublished report by Louisiana Department of Wildlife and Fisheries, Baton Rouge, LA, USA, 43 pp.

Ross, S.T., R.J. Heise, W.T. Slack, J.A. Ewing, III, and M. Dugo. 2000. Movement and habitat use of the Gulf sturgeon (Acipenser oxyrinchus desotoi) in the Pascagoula drainage of Mississippi: year 4. Mississippi Department of Wildlife, Fisheries, and Parks and Museum of Natural Science. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 15. 58 pp.

Ross, S.T., R.J. Heise, W.T. Slack, and M. Dugo. 2001a. Habitat requirements of Gulf Sturgeon *(Acipenser oxyrinchus desotoi)* in the northern Gulf of Mexico. Department of Biological Sciences, University of Southern Mississippi and Mississippi Museum of Natural Science. Funded by the Shell Marine Habitat Program, National Fish and Wildlife Foundation. 26 pp.

Ross, S.T., R.J. Heise, M.A. Dugo, and W.T. Slack. 2001b. Movement and habitat use of the Gulf sturgeon *(Acipenser oxyrinchus desotoi)* in the Pascagoula drainage of Mississippi: year 5. Department of Biological Sciences, University of Southern Mississippi, and Mississippi Museum of Natural Science. Funded by U.S. Fish and Wildlife Service, Project No. E-1, Segment 16.

Santucci, V.J., Jr., S.R. Gephard, and S.M. Pescitelli. 2005. Effects of multiple low-head dams on fish, macroinvertebrates, habitat and water quality in the Fox River, Illinois. N. American Journal of Fisheries Management. 25.

Sawyer, J. A., P. M. Stewart, M. M. Mullen, T. P. Simon, and H. H. Bennett. 2004. Influence of Habitat, Water Quality, and Land Use On Macro-invertebrate and Fish Assemblages of a Southeastern Coastal Plain Watershed, USA. Aquatic Ecosystem Health & Management.

Scollan, D. and F.M. Parauka. 2008. Documentation of Gulf sturgeon spawning in the Apalachicola River, Florida. U. S. Fish and Wildlife Service, Panama City, Florida.

Secor, D. H., and T. E. Gunderson. 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon (*Acipenser oxyrinchus*). Fishery Bulletin U.S.

Seigel, R., K. Anderson, B. Durkin, J. Cole, S. Cooke, and T. Richards-Dimitrie. 2016. Effectiveness of Nest Site Restoration for the Endangered Northern Map Turtle Report 2: Use of Artificial Nesting Sites and Wildlife Exclusion Fence to Enhance Nesting Success. Towson University. Maryland Department of Transportation.

Selman, W. 2018. Diamonds in the rough: status of two imperiled *Graptemys* species (*Graptemys oculifera* and *G. peralensis*) in the Pearl River of Jackson, MS.

Selman, W. and R. Jones. 2017. Population Structure, Status, and Conservation of Two *Graptemys* Species from the Pearl River, Mississippi. Journal of Herpetology, 51(1).

Selman, W. and P. Lindeman. 2015. Life History and ecology of the Pascagoula map turtle (*Graptemys gibbonsi*). Herpetological Conservation and Biology 10(2).

Selman, W. and P. Lindeman. 2018. Spatial, Seasonal, and Sexual Variation in the Diet of *Graptemys flavimaculata*, a Threatened Turtle of the Pascagoula River System, Mississippi, USA. Copeia, 106(2).

Selman, W., and C. Qualls. 2011. Basking Ecology of the Yellow-Blotched Sawback (*Graptemys flavimaculata*), an Imperiled Turtle Species of the Pascagoula River System, Mississippi, United States. Chelonian Conservation and Biology, 10(2).

Selman, W., C. Qualls, and J. Owen. 2013. Effects of Human Disturbance on the Behavior and Physiology of an Imperiled Freshwater Turtle. The Journal of Wildlife Management 77(5).

Selman, W. and H. Smith. 2017. Diamonds in the Rough: Status of Two Imperiled Graptemys Species (*Graptemys oculifera* and *G. pearlensis*) in the Pearl River of Jackson, MS , Year 1.

Shealy, R. 1976. The natural history of the Alabama map turtle, *Graptemys Pulchra* Baur, in Alabama. Bulletin of the Florida State Museum of Biological Sciences. 21(2).

Shively, S. 1999 Survey for the Ringed Map Turtle (*Graptemys oculifera*) in the Bogue Chitto River Louisiana. Louisiana Natural Heritage Pr ogram, Louisiana Department of Wildlife and Fisheries. E-1-9.

Skalak, K., J. Pizzuto, and D.D. Hart. 2009. Influence of small dams on downstream channel characteristics in Pennsylvania and Maryland: implications for the long-term geomorphic effects of dam removal. Journal of the American Water Resources Association. Vol. 45, No. 1.

Slack, W. T., B. R. Lewis, K. J. Killgore, H. J. Theel, and W. P. Lorentz. 2014. Gulf Sturgeon Carcass Assessment Following the 2011 Pearl River Fish Kill. Report to USFWS, Louisiana Ecological Services Office. USACE Engineer Research and Development Center. 190pp.

Smith, S., C.F., S.J. Meiners, R.P. Hastings, T. Thomas, and R.E. Colombo. 2017. Low-head dams impacts on habitat and the functional composition of fish communities. River Research and Applications 33.

Smith, G., J. Iverson and J. Rettig. 2018. Frequency of propeller damage in a turtle community in a northern Indiana, USA, lake; a long term study. Herpetological Conservation and Biology 13(3).

Spencer, r., J. Dyke and M. Thompson. 2017. Critically evaluating best management practices for preventing freshwater turtle extinctions. Conservation Biology 31(6).

Stabile, J., J.R. Waldman, F. Parauka, and I. Wirgin. 1996. Stock structure and homing fidelity in Gulf of Mexico sturgeon (*Acipenser oxyrinchus desotoi*) based on restriction fragment length polymorphism and sequence analyses of mitochondrial DNA. Genetics.

Stewart, J. 1988. A recovery plan for the ringed sawback turtle *Graptemys oculifera*. U.S. Fish and Wildlife Service, Southeast Region, Atlanta, GA.

Stewart, P. M. and T. O. Swinford. 1995. Identification of sediment and nutrient sources impacting critically endangered mussel species' habitat in a small agricultural stream. In: J. R. Pratt, N. Bowers, J. R. Stauffer (Eds.), *Making Environmental Science: A Festschrift in honor of John Cairns, Jr.* ECOPRINT, Portland, OR.

Stewart, P. M.; Sawyer, J. B.; Parauka, F. M.; Reategul-Zirena, E. G., 2012: Summer holding areas of the Gulf sturgeon within the Conecuh/Escambia River system, Alabama and Florida. In: Watersheds: processes, management and impact. G. I. Kuhn and J. R. Emery (Eds). Nova Science Publ. Inc., Hauppauge, pp.75 .

Sulak, K.J. and J.P. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon, *Acipenser oxyrinchus desotoi*, in the Suwannee River, Florida, USA: a synopsis. Journal of Applied Ichthyology.

Sulak, K.J., M. Randall, J. Clugston, and W.H. Clark. 2004. Critical spawning habitat, early life history requirements, and other life history and population aspects of the Gulf sturgeon in the Suwannee River. Final Report to the Florida Fish and Wildlife Conservation Commission, Nongame Wildlife Program. U.S. Geological Survey, Gainesville, FL.

Sulak, K.J., M. T. Randall, R. E. Edwards, T. M. Summers, K. E. Luke, W. T. Smith, A. D. Norem, W. M. Harden, R. H. Lukens, F. Parauka, S. Bolden, and R. Lehnert. 2009. Defining winter trophic habitat of juvenile Gulf Sturgeon in the Suwannee and Apalachicola rivermouth estuaries, acoustic telemetry investigations. Journal of Applied Ichthyology.

Sulak, K.J., M. T. Randall, and J. J. Berg. 2012. Feeding habitats of the Gulf sturgeon, Acipenser oxyrinchus desotoi, in the Suwannee and Yellow rivers, Florida, as identified by multiple stable isotope analyses. Environmental Biology of Fishes.

Sulak, K. J., F. Parauka, W. T. Slack, R. T. Ruth, M. T. Randall, K. Luke, and M. E. Price. 2016. Status of scientific knowledge, recovery progress, and future research directions for the Gulf Sturgeon, *Acipenser oxyrinchus desotoi* Vladykov, 1955. Journal of Applied Ichthyology 32:87

Tiemann, J.S., S.A. Douglass, A.P. Stodola, and K.S. Cummings. Effects of low-head dams on freshwater mussels in the Vermilion River Basin, Illinois, with comments on a natural dam removal. Transactions of the Illinois State Academy of Science. 109.

Tipton, J., H. Bart, Jr., and K. Piller. 2004. Geomorphic disturbance and its impact on darter (Teleostomi: Percidae) distribution and abundance in the Pearl River drainage, Mississippi. Hydorbiologia 527.

U.S. Army Corps of Engineers (USACE). 1996. Flood Control, Pearl River Basin, Jackson Metropolitan Area Mississippi. Draft feasibility report and environmental impact statement. Corps of Engineers, Mobile, Alabama.

U.S. Fish and Wildlife Service. 1988. Recovery Plan for the Ringed Sawback Turtle, *Graptemys oculifera.* U.S. Fish and Wildlife Service, Atlanta, Georgia.

U .S. Fish and Wildlife Service. 2002. Ringed Map Turtle (*Graptemys oculifera)* 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service, Mississippi Ecological Services Office, Jackson, Mississippi.

USFWS. 2005. Unpublished data. Panama City Field Office, Panama City, FL.

USFWS. 2009. Unpublished data. Panama City Field Office, Panama City, FL.

USFWS. 2012. Unpublished data. Panama City Field Office, Panama City, FL.

U.S. Fish and Wildlife Service Baton Rouge Fish and Wildlife Conservation Office. 2018. Barriers to Migration for Gulf Sturgeon in the Pearl River Watershed.

U.S. Fish and Wildlife Service Baton Rouge Fish and Wildlife Conservation Office. 2019. Personal Communication.

USFWS (U.S. Fish and Wildlife Service) and GSMFC (Gulf States Marine Fisheries Commission). 1995. Gulf sturgeon (*Acipenser oxyrinchus desotoi*) Recovery/Management plan. Atlanta, Georgia. 170 pp.

USFWS (U.S. Fish and Wildlife Service) and National Marine Fisheries Service. 2003. Designation of critical habitat for the Gulf sturgeon. USFWS and National Marine Fisheries Service, Panama City, Florida.

USFWS and NMFS (National Marine Fisheries Service). 2009. Gulf sturgeon (*Acipenser oxyrinchus desotoi*) 5-year review: summary and evaluation. Panama City, FL and St. Petersburg, Florida. 49 pp.

U.S. Office of the Federal Register. 1991. Endangered and threatened wildlife and plants; threatened status for the Gulf Sturgeon. Federal Register 56:189(30 September 1991):49653–49658.

Vladykov, V.D. 1955. A comparison of Atlantic sea sturgeon with a new subspecies from the Gulf of Mexico (*Acipenser oxyrhynchus desotoi*). Journal Fish Research Board Canada.

Wakeford, A. 2001. State of Florida conservation plan for gulf sturgeon (*Acipenser oxyrinchus desotoi*). Florida Marine Research Institute Technical Report TR-8. 100pp.

White R.G. and B. Mefford. 2002. Assessment of behavior and swimming ability of Yellowstone River sturgeon for design of fish passage devices. Montana Cooperative Fishery Research Unit, Montana State University; and Water Resources Research Laboratory, Bureau of Reclamation, Denver. Prepared for the Omaha District Corps of Engineers.

Wooley, C. M., P. A. Moon, and E. J. Crateau. 1982. A larval Gulf of Mexico sturgeon (*Acipenser oxyrhynchus desotoi*) from the Apalachicola River, Florida. Northeast Gulf Science.

Wooley, C.M. 1985. Evaluation ofmorphometric characters used in taxonomic separation of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi,* pp. 97-103 *In* North American Sturgeon, Vol. 6., Developments in Environmental Biology of Fishes, edited by D.W.F. Binkowsi and S. I. Doroshov, Junk Publishing, The Netherlands.

Wooley, C.M. and E.J. Crateau. 1985. Movement, microhabitat, exploitation, and management of Gulf of Mexico sturgeon, Apalachicola River, Florida. North American Journal of Fisheries Management.

Wyneken, J., T. Burke, M. Salmon, and D. Pedersen. 1988. Egg failure in natural and relocated sea turtle nests. Journal of Herpetology 22(1).

Murphy, D. and Weiland, P. 2014. The use of surrogates in implementation of the federal Endangered Species Act—proposed fixes to a proposed rule. J Environ Stud Sci. Springer.

Caro, T. 2010. Conservation by Proxy: Indicator, Umbrella, Keystone, Flagship, and Other Surrogate Species. Island Press, Washington D.C.

Pearl River Basin, Mississippi, Federal Flood Risk Management Project

Annex D2 - IPaC report and USFWS Correspondence

June 2024

IPaC resource list

This report is an automatically generated list of species and other resources such as critical habitat (collectively referred to as *trust resources*) under the U.S. Fish and Wildlife Service's (USFWS) jurisdiction that are known or expected to be on or near the project area referenced below. The list may also include trust resources that occur outside of the project area, but that could potentially be directly or indirectly a ected by activities in the project area. However, determining the likelihood and extent of e ects a project may have on trust resources typically requires gathering additional site-speci c (e.g., vegetation/species surveys) and project-speci c (e.g., magnitude and timing of proposed activities) information.

Below is a summary of the project information you provided and contact information for the USFWS o ce(s) with jurisdiction in the de ned project area. Please read the introduction to each section that follows (Endangered Species, Migratory Birds, USFWS Facilities, and NWI Wetlands) for additional information applicable to the trust resources addressed in that section.

Location

Hinds and Rankin counties, Mississippi

Local office

Mississippi Ecological Services Field O ce

(601) 965-4900
(601) 965-4340

6578 Dogwood View Parkway, Suite A

Jackson, MS 39213-7856

Endangered species

This resource list is for informational purposes only and does not constitute an analysis of project level impacts.

The primary information used to generate this list is the known or expected range of each species. Additional areas of in uence (AOI) for species are also considered. An AOI includes areas outside of the species range if the species could be indirectly a ected by activities in that area (e.g., placing a dam upstream of a sh population even if that sh does not occur at the dam site, may indirectly impact the species by reducing or eliminating water ow downstream). Because species can move, and site conditions can change, the species on this list are not guaranteed to be found on or near the project area. To fully determine any potential e ects to species, additional site-speci c and project-speci c information is often required.

Section 7 of the Endangered Species Act **requires** Federal agencies to "request of the Secretary information whether any species which is listed or proposed to be listed may be present in the area of such proposed action" for any project that is conducted, permitted, funded, or licensed by any Federal agency. A letter from the local o ce and a species list which ful lls this requirement can **only** be obtained by requesting an o cial species list from either the Regulatory Review section in IPaC (see directions below) or from the local eld o ce directly.

For project evaluations that require USFWS concurrence/review, please return to the IPaC website and request an o cial species list by doing the following:

1. Draw the project location and click CONTINUE.
2. Click DEFINE PROJECT.
3. Log in (if directed to do so).
4. Provide a name and description for your project.
5. Click REQUEST SPECIES LIST.

Listed species[1] and their critical habitats are managed by the Ecological Services Program of the U.S. Fish and Wildlife Service (USFWS) and the sheries division of the National Oceanic and Atmospheric Administration (NOAA Fisheries[2]).

Species and critical habitats under the sole responsibility of NOAA Fisheries are **not** shown on this list. Please contact NOAA Fisheries for species under their jurisdiction.

---

1. Species listed under the Endangered Species Act are threatened or endangered; IPaC also shows species that are candidates, or proposed, for listing. See the listing status page for more information. IPaC only shows species that are regulated by USFWS (see FAQ).

2. NOAA Fisheries, also known as the National Marine Fisheries Service (NMFS), is an o ce of the National Oceanic and Atmospheric Administration within the Department of Commerce.

The following species are potentially a ected by activities in this location:

Mammals

NAME	STATUS
Northern Long-eared Bat Myotis septentrionalis Wherever found No critical habitat has been designated for this species. https://ecos.fws.gov/ecp/species/9045	Threatened

Reptiles

NAME	STATUS
Alligator Snapping Turtle Macrochelys temminckii Wherever found No critical habitat has been designated for this species. https://ecos.fws.gov/ecp/species/4658	Proposed Threatened
Ringed Map Turtle Graptemys oculifera Wherever found No critical habitat has been designated for this species. https://ecos.fws.gov/ecp/species/2664	Threatened

Fishes

NAME	STATUS
Gulf Sturgeon Acipenser oxyrinchus (=oxyrhynchus) desotoi Wherever found There is **nal** critical habitat for this species. Your location overlaps the critical habitat. https://ecos.fws.gov/ecp/species/651	Threatened

Insects

NAME	STATUS
Monarch Butter y Danaus plexippus Wherever found No critical habitat has been designated for this species. https://ecos.fws.gov/ecp/species/9743	Candidate

Critical habitats

Potential e ects to critical habitat(s) in this location must be analyzed along with the endangered species themselves.

This location overlaps the critical habitat for the following species:

NAME	TYPE
Gulf Sturgeon Acipenser oxyrinchus (=oxyrhynchus) desotoi https://ecos.fws.gov/ecp/species/651#crithab	Final

Migratory birds

Certain birds are protected under the Migratory Bird Treaty Act[1] and the Bald and Golden Eagle Protection Act[2].

Any person or organization who plans or conducts activities that may result in impacts to migratory birds, eagles, and their habitats should follow appropriate regulations and consider implementing appropriate conservation measures, as described below.

1. The Migratory Birds Treaty Act of 1918.
2. The Bald and Golden Eagle Protection Act of 1940.

Additional information can be found using the following links:

- Birds of Conservation Concern https://www.fws.gov/program/migratory-birds/species
- Measures for avoiding and minimizing impacts to birds https://www.fws.gov/library/collections/avoiding-and-minimizing-incidental-take-migratory-birds
- Nationwide conservation measures for birds https://www.fws.gov/sites/default/ les/documents/nationwide-standard-conservation-measures.pdf

The birds listed below are birds of particular concern either because they occur on the USFWS Birds of Conservation Concern (BCC) list or warrant special attention in your project location. To learn more about the levels of concern for birds on your list and how this list is generated, see the FAQ below. This is not a list of every bird you may nd in this location, nor a guarantee that every bird on this list will be found in your project area. To see exact locations of where birders and the general public have sighted birds in and around your project area, visit the E-bird data mapping tool (Tip: enter your location, desired date range and a species on your list). For projects that occur o the Atlantic Coast, additional maps and models detailing the relative occurrence and abundance of bird species on your list are available. Links to additional information about Atlantic Coast birds, and other important information about your migratory bird list, including how to properly interpret and use your migratory bird report, can be found [below](#).

For guidance on when to schedule activities or implement avoidance and minimization measures to reduce impacts to migratory birds on your list, click on the PROBABILITY OF PRESENCE SUMMARY at the top of your list to see when these birds are most likely to be present and breeding in your project area.

NAME	BREEDING SEASON
American Kestrel Falco sparverius paulus This is a Bird of Conservation Concern (BCC) only in particular Bird Conservation Regions (BCRs) in the continental USA https://ecos.fws.gov/ecp/species/9587	Breeds Apr 1 to Aug 31
Bald Eagle Haliaeetus leucocephalus This is not a Bird of Conservation Concern (BCC) in this area, but warrants attention because of the Eagle Act or for potential susceptibilities in o shore areas from certain types of development or activities.	Breeds Sep 1 to Jul 31
Brown-headed Nuthatch Sitta pusilla This is a Bird of Conservation Concern (BCC) only in particular Bird Conservation Regions (BCRs) in the continental USA	Breeds Mar 1 to Jul 15
Cerulean Warbler Dendroica cerulea This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska. https://ecos.fws.gov/ecp/species/2974	Breeds Apr 26 to Jul 20
Chimney Swift Chaetura pelagica This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.	Breeds Mar 15 to Aug 25
Kentucky Warbler Oporornis formosus This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.	Breeds Apr 20 to Aug 20
King Rail Rallus elegans This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska. https://ecos.fws.gov/ecp/species/8936	Breeds May 1 to Sep 5

Lesser Yellowlegs Tringa flavipes | Breeds elsewhere

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.
https://ecos.fws.gov/ecp/species/9679

Painted Bunting Passerina ciris | Breeds Apr 25 to Aug 15

This is a Bird of Conservation Concern (BCC) only in particular Bird Conservation Regions (BCRs) in the continental USA

Prairie Warbler Dendroica discolor | Breeds May 1 to Jul 31

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.

Prothonotary Warbler Protonotaria citrea | Breeds Apr 1 to Jul 31

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.

Red-headed Woodpecker Melanerpes erythrocephalus | Breeds May 10 to Sep 10

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.

Rusty Blackbird Euphagus carolinus | Breeds elsewhere

This is a Bird of Conservation Concern (BCC) only in particular Bird Conservation Regions (BCRs) in the continental USA

Willet Tringa semipalmata | Breeds Apr 20 to Aug 5

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.

Wood Thrush Hylocichla mustelina | Breeds May 10 to Aug 31

This is a Bird of Conservation Concern (BCC) throughout its range in the continental USA and Alaska.

Probability of Presence Summary

The graphs below provide our best understanding of when birds of concern are most likely to be present in your project area. This information can be used to tailor and schedule your project activities to avoid or minimize impacts to birds. Please make sure you read and understand the FAQ "Proper Interpretation and Use of Your Migratory Bird Report" before using or attempting to interpret this report.

Probability of Presence (■)

Each green bar represents the bird's relative probability of presence in the 10km grid cell(s) your project overlaps during a particular week of the year. (A year is represented as 12 4-week months.) A taller bar indicates a higher probability of species presence. The survey e ort (see below) can be used to establish a level of con dence in the presence score. One can have higher con dence in the presence score if the corresponding survey e ort is also high.

How is the probability of presence score calculated? The calculation is done in three steps:

1. The probability of presence for each week is calculated as the number of survey events in the week where the species was detected divided by the total number of survey events for that week. For example, if in week 12 there were 20 survey events and the Spotted Towhee was found in 5 of them, the probability of presence of the Spotted Towhee in week 12 is 0.25.
2. To properly present the pattern of presence across the year, the relative probability of presence is calculated. This is the probability of presence divided by the maximum probability of presence across all weeks. For example, imagine the probability of presence in week 20 for the Spotted Towhee is 0.05, and that the probability of presence at week 12 (0.25) is the maximum of any week of the year. The relative probability of presence on week 12 is 0.25/0.25 = 1; at week 20 it is 0.05/0.25 = 0.2.
3. The relative probability of presence calculated in the previous step undergoes a statistical conversion so that all possible values fall between 0 and 10, inclusive. This is the probability of presence score.

To see a bar's probability of presence score, simply hover your mouse cursor over the bar.

Breeding Season (■)
Yellow bars denote a very liberal estimate of the time-frame inside which the bird breeds across its entire range. If there are no yellow bars shown for a bird, it does not breed in your project area.

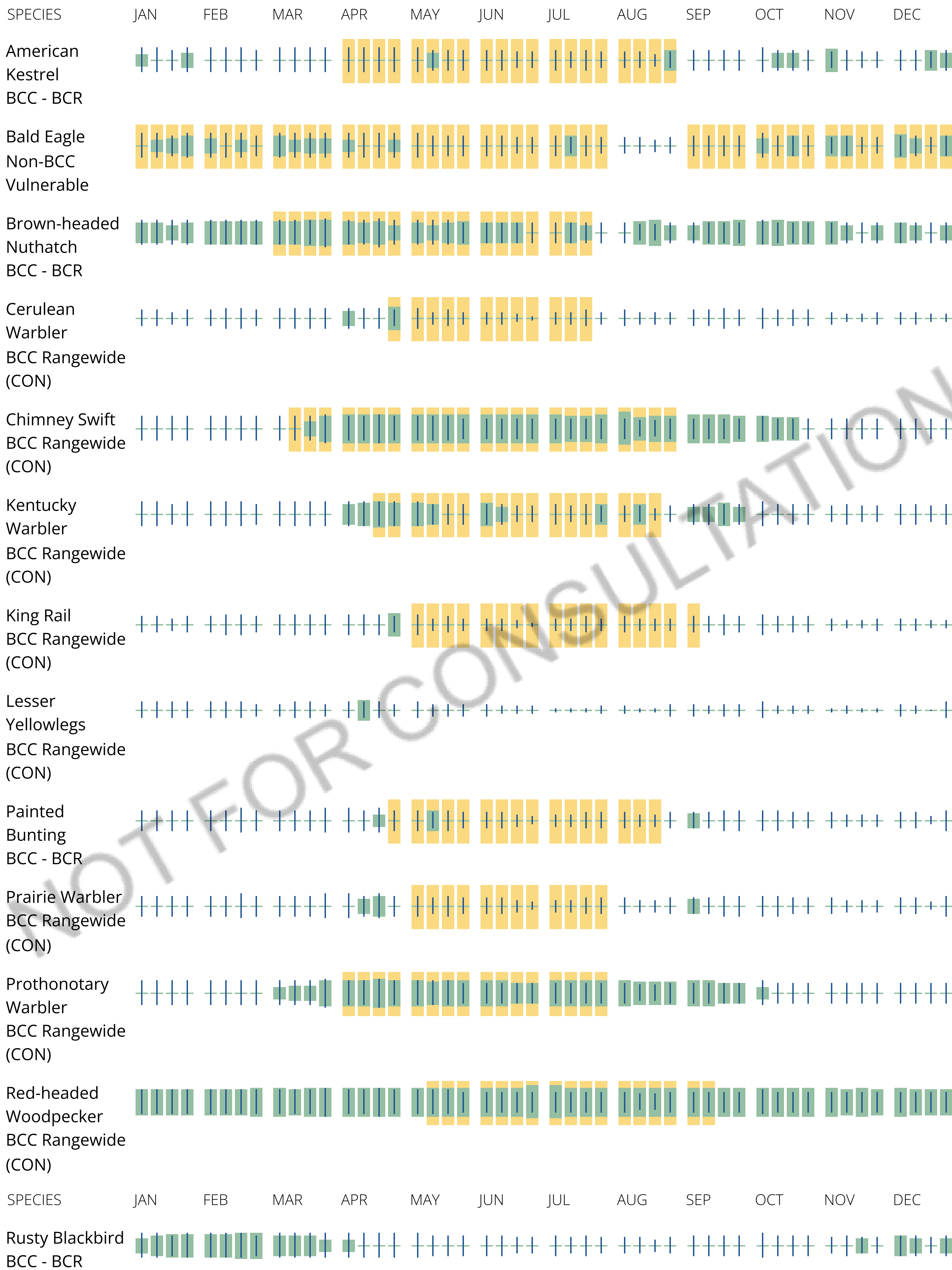
Survey Effort (|)
Vertical black lines superimposed on probability of presence bars indicate the number of surveys performed for that species in the 10km grid cell(s) your project area overlaps. The number of surveys is expressed as a range, for example, 33 to 64 surveys.

To see a bar's survey e ort range, simply hover your mouse cursor over the bar.

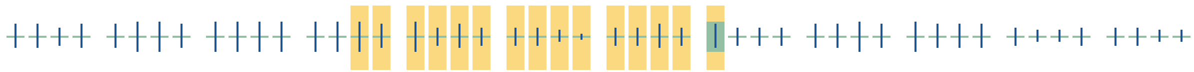
No Data (–)
A week is marked as having no data if there were no survey events for that week.

Survey Timeframe
Surveys from only the last 10 years are used in order to ensure delivery of currently relevant information. The exception to this is areas o the Atlantic coast, where bird returns are based on all years of available data, since data in these areas is currently much more sparse.

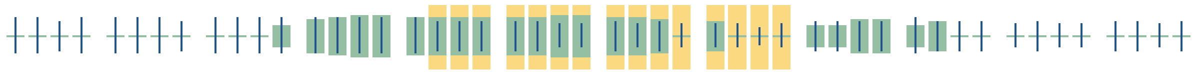
■ probability of presence ■ breeding season | survey e ort – no data


SPECIES
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
American Kestrel
BCC - BCR
Bald Eagle
Non-BCC Vulnerable
Brown-headed Nuthatch
BCC - BCR
Cerulean Warbler
BCC Rangewide (CON)
Chimney Swift
BCC Rangewide (CON)
Kentucky Warbler
BCC Rangewide (CON)
King Rail
BCC Rangewide (CON)
Lesser Yellowlegs
BCC Rangewide (CON)
Painted Bunting
BCC - BCR
Prairie Warbler
BCC Rangewide (CON)
Prothonotary Warbler
BCC Rangewide (CON)
Red-headed Woodpecker
BCC Rangewide (CON)
SPECIES
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
SEP
OCT
NOV
DEC
Rusty Blackbird
BCC - BCR

Willet
BCC Rangewide
(CON)



Wood Thrush
BCC Rangewide
(CON)



Tell me more about conservation measures I can implement to avoid or minimize impacts to migratory birds.

Nationwide Conservation Measures describes measures that can help avoid and minimize impacts to all birds at any location year round. Implementation of these measures is particularly important when birds are most likely to occur in the project area. When birds may be breeding in the area, identifying the locations of any active nests and avoiding their destruction is a very helpful impact minimization measure. To see when birds are most likely to occur and be breeding in your project area, view the Probability of Presence Summary. Additional measures or permits may be advisable depending on the type of activity you are conducting and the type of infrastructure or bird species present on your project site.

What does IPaC use to generate the list of migratory birds that potentially occur in my specified location?

The Migratory Bird Resource List is comprised of USFWS Birds of Conservation Concern (BCC) and other species that may warrant special attention in your project location.

The migratory bird list generated for your project is derived from data provided by the Avian Knowledge Network (AKN). The AKN data is based on a growing collection of survey, banding, and citizen science datasets and is queried and filtered to return a list of those birds reported as occurring in the 10km grid cell(s) which your project intersects, and that have been identified as warranting special attention because they are a BCC species in that area, an eagle (Eagle Act requirements may apply), or a species that has a particular vulnerability to offshore activities or development.

Again, the Migratory Bird Resource list includes only a subset of birds that may occur in your project area. It is not representative of all birds that may occur in your project area. To get a list of all birds potentially present in your project area, please visit the Rapid Avian Information Locator (RAIL) Tool.

What does IPaC use to generate the probability of presence graphs for the migratory birds potentially occurring in my specified location?

The probability of presence graphs associated with your migratory bird list are based on data provided by the Avian Knowledge Network (AKN). This data is derived from a growing collection of survey, banding, and citizen science datasets.

Probability of presence data is continuously being updated as new and better information becomes available. To learn more about how the probability of presence graphs are produced and how to interpret them, go the Probability of Presence Summary and then click on the "Tell me about these graphs" link.

How do I know if a bird is breeding, wintering or migrating in my area?

To see what part of a particular bird's range your project area falls within (i.e. breeding, wintering, migrating or year-round), you may query your location using the RAIL Tool and look at the range maps provided for birds in your area at the bottom of the profiles provided for each bird in your results. If a bird on your migratory bird species list has a breeding season associated with it, if that bird does occur in your project area, there may be nests present at some point within the timeframe specified. If "Breeds elsewhere" is indicated, then the bird likely does not breed in your project area.

What are the levels of concern for migratory birds?

Migratory birds delivered through IPaC fall into the following distinct categories of concern:

1. "BCC Rangewide" birds are Birds of Conservation Concern (BCC) that are of concern throughout their range anywhere within the USA (including Hawaii, the Pacific Islands, Puerto Rico, and the Virgin Islands);
2. "BCC - BCR" birds are BCCs that are of concern only in particular Bird Conservation Regions (BCRs) in the continental USA; and
3. "Non-BCC - Vulnerable" birds are not BCC species in your project area, but appear on your list either because of the Eagle Act requirements (for eagles) or (for non-eagles) potential susceptibilities in offshore areas from certain types of development or activities (e.g. offshore energy development or longline fishing).

Although it is important to try to avoid and minimize impacts to all birds, efforts should be made, in particular, to avoid and minimize impacts to the birds on this list, especially eagles and BCC species of rangewide concern. For more information on conservation measures you can implement to help avoid and minimize migratory bird impacts and requirements for eagles, please see the FAQs for these topics.

Details about birds that are potentially affected by offshore projects

For additional details about the relative occurrence and abundance of both individual bird species and groups of bird species within your project area off the Atlantic Coast, please visit the Northeast Ocean Data Portal. The Portal also offers data and information about other taxa besides birds that may be helpful to you in your project review. Alternately, you may download the bird model results files underlying the portal maps through the NOAA NCCOS Integrative Statistical Modeling and Predictive Mapping of Marine Bird Distributions and Abundance on the Atlantic Outer Continental Shelf project webpage.

Bird tracking data can also provide additional details about occurrence and habitat use throughout the year, including migration. Models relying on survey data may not include this information. For additional information on marine bird tracking data, see the Diving Bird Study and the nanotag studies or contact Caleb Spiegel or Pam Loring.

What if I have eagles on my list?

If your project has the potential to disturb or kill eagles, you may need to obtain a permit to avoid violating the Eagle Act should such impacts occur.

Proper Interpretation and Use of Your Migratory Bird Report

The migratory bird list generated is not a list of all birds in your project area, only a subset of birds of priority concern. To learn more about how your list is generated, and see options for identifying what other birds may be in your project area, please see the FAQ "What does IPaC use to generate the migratory birds potentially occurring in my specified location". Please be aware this report provides the "probability of presence" of birds within the 10 km grid cell(s) that overlap your project; not your exact project footprint. On the graphs provided, please also look carefully at the survey effort (indicated by the black vertical bar) and for the existence of the "no data" indicator (a red horizontal bar). A high survey effort is the key component. If the survey effort is high, then the probability of presence score can be viewed as more dependable. In contrast, a low survey effort bar or no data bar means a lack of data and, therefore, a lack of certainty about presence of the species. This list is not perfect; it is simply a starting point for identifying what birds of concern have the potential to be in your project area, when they might be there, and if they might be breeding (which means nests might be present). The list helps you know what to look for to confirm presence, and helps guide you in knowing when to implement conservation measures to avoid or minimize potential impacts from your project activities, should presence be confirmed. To learn more about conservation measures, visit the FAQ "Tell me about conservation measures I can implement to avoid or minimize impacts to migratory birds" at the bottom of your migratory bird trust resources page.

Facilities

National Wildlife Refuge lands

Any activity proposed on lands managed by the National Wildlife Refuge system must undergo a 'Compatibility Determination' conducted by the Refuge. Please contact the individual Refuges to discuss any questions or concerns.

There are no refuge lands at this location.

Fish hatcheries

There are no fish hatcheries at this location.

Wetlands in the National Wetlands Inventory (NWI)

Impacts to NWI wetlands and other aquatic habitats may be subject to regulation under Section 404 of the Clean Water Act, or other State/Federal statutes.

For more information please contact the Regulatory Program of the local U.S. Army Corps of Engineers District.

Wetland information is not available at this time

This can happen when the National Wetlands Inventory (NWI) map service is unavailable, or for very large projects that intersect many wetland areas. Try again, or visit the NWI map to view wetlands at this location.

Data limitations

The Service's objective of mapping wetlands and deepwater habitats is to produce reconnaissance level information on the location, type and size of these resources. The maps are prepared from the analysis of high altitude imagery. Wetlands are identi ed based on vegetation, visible hydrology and geography. A margin of error is inherent in the use of imagery; thus, detailed on-the-ground inspection of any particular site may result in revision of the wetland boundaries or classi cation established through image analysis.

The accuracy of image interpretation depends on the quality of the imagery, the experience of the image analysts, the amount and quality of the collateral data and the amount of ground truth veri cation work conducted. Metadata should be consulted to determine the date of the source imagery used and any mapping problems.

Wetlands or other mapped features may have changed since the date of the imagery or eld work. There may be occasional di erences in polygon boundaries or classi cations between the information depicted on the map and the actual conditions on site.

Data exclusions

Certain wetland habitats are excluded from the National mapping program because of the limitations of aerial imagery as the primary data source used to detect wetlands. These habitats include seagrasses or submerged aquatic vegetation that are found in the intertidal and subtidal zones of estuaries and nearshore coastal waters. Some deepwater reef communities (coral or tuber cid worm reefs) have also been excluded from the inventory. These habitats, because of their depth, go undetected by aerial imagery.

Data precautions

Federal, state, and local regulatory agencies with jurisdiction over wetlands may de ne and describe wetlands in a di erent manner than that used in this inventory. There is no attempt, in either the design or products of this inventory, to de ne the limits of proprietary jurisdiction of any Federal, state, or local government or to establish the geographical scope of the regulatory programs of government agencies. Persons intending to engage in activities involving modi cations within or adjacent to wetland areas should seek the advice of appropriate Federal, state, or local agencies concerning speci ed agency regulatory programs and proprietary jurisdictions that may a ect such activities.

From: Campbell, Tamara N
To: Gilmore, Tammy F CIV USARMY CEMVN (USA)
Subject: [Non-DoD Source] Re: [EXTERNAL] One Lake T&E IPaC list
Date: Tuesday, March 21, 2023 1:57:32 PM

Tammy,

The wood stork was removed from MS's consultation zone late last year. Currently, it is proposed for delisting with pending review and decision. Therefore, you do not need to consult on the wood stork.

As a candidate species, there are no legal regulations for the monarch butterfly under the Endangered Species Act. Any work to conserve the monarch can be included in a Candidate Conservation Program, which is considered during listing decisions. There may be future official regulations if the monarch is re-evaluated and the USFWS finds that listing is warranted. The monarch may be prioritized in the 2024 workplan at which time, it could be proposed for listing. Currently, it is warranted, but precluded by other priorities.

Both the Pearl River map turtle and alligator snapping turtle were warranted, and proposed for listing.

Is that information helpful?

Sincerely,

TC

Tamara Campbell
Fish & Wildlife Biologist
U.S. Fish and Wildlife Service
6578 Dogwood View Parkway
Jackson, MS 39213
Office: (601) 321-1138
Email: tamara_campbell@fws.gov

From: Gilmore, Tammy F CIV USARMY CEMVN (USA) <Tammy.F.Gilmore@usace.army.mil>
Sent: Tuesday, March 21, 2023 12:03 PM
To: Campbell, Tamara N <tamara_campbell@fws.gov>
Subject: [EXTERNAL] One Lake T&E IPaC list

Good afternoon Tamara,

I am working on getting the BA in the correct format and including the necessary information. The 2018 BA covered the threatened Gulf sturgeon, the threatened Ringed Sawback (Map) Turtle, the threatened Northern longeared bat and the threatened Wood stork. The Service suggested we re-consult on the NLEB (due to up-listing), and conference on the Pearl River map turtle and the alligator snapping turtle as well as conduct additional analysis on the GS.

The attached IPaC search (using the action area from the previous BO) generated a list that doesn't include the wood stork (previously consulted on) and does include the Monarch butterfly (not previously recommended).

Can you please confirm which species you expect to be included in the current BA?

Thank you

Tammy Gilmore
Biologist/Senior NEPA Specialist
USACE New Orleans District- RPEDS
7400 Leake Ave
New Orleans LA 70118
504-862-1002

From: Wagner, Matthew D
To: Gilmore, Tammy F CIV USARMY CEMVN (USA); Campbell, Tamara N; Pearson, Luke S
Cc: Austin, James A; Lombardi, Melissa D
Subject: [URL Verdict: Neutral][Non-DoD Source] New Mussel Species to Consider
Date: Monday, April 10, 2023 8:22:13 AM
Attachments: image001.png
image002.png

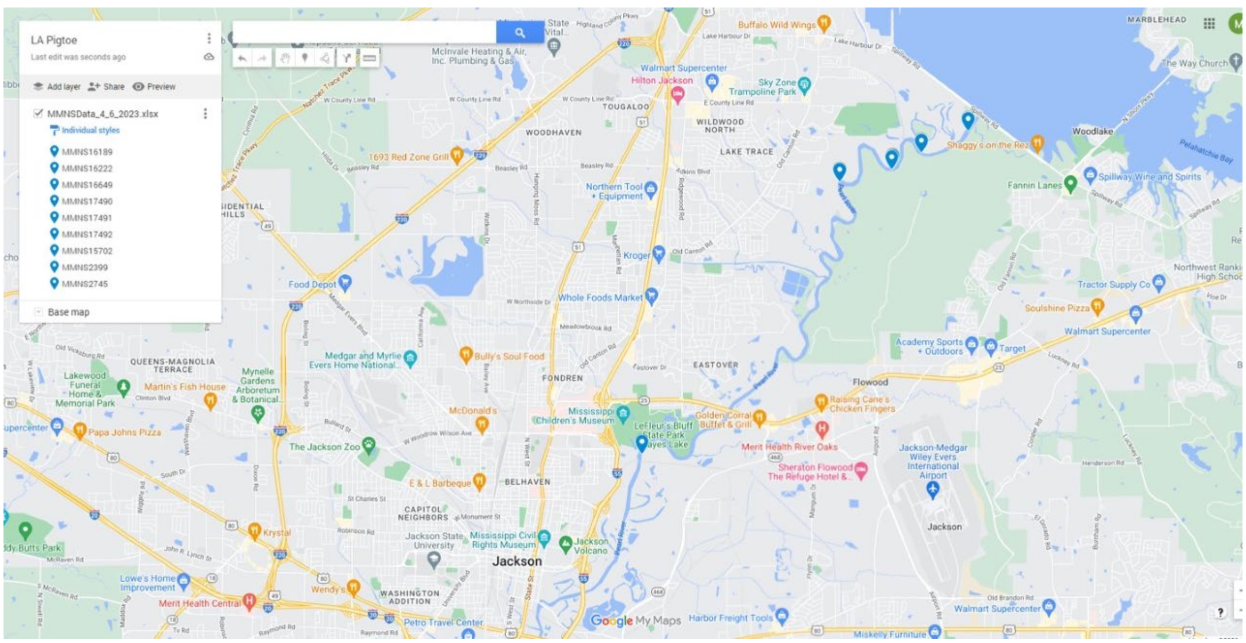
Tammy,

On March 20, 2023, the Service proposed to list the Louisiana Pigtoe as a threatened species under the ESA and designate critical habitat (https://www.federalregister.gov/documents/2023/03/20/2023-05107/endangered-and-threatened-wildlife-and-plants-endangered-species-status-with-critical-habitat-for). At the time of listing the Louisiana Pigtoe was not known to occur in the Pearl River upstream of Angie, Louisiana (toe of the boot). However, biologists from Mississippi Department of Wildlife, Fisheries, and Parks recently completed a mussel survey of the entire Pearl River and collected individuals from the Pearl River below Ross Barnett Reservoir that were morphologically identified as Louisiana Pigtoe. As this species is hard to distinguish from two other mussel species found in the system, the specimens were sent to a geneticist with the USGS Wetland and Aquatic Research Center in Gainesville, Florida for genetic confirmation. The results of this study were submitted to a peer reviewed journal for publication on 3/16/2023. Our unpublished results confirmed the MDWFP identification as Louisiana Pigtoe. Additionally, two additional individuals were identified as Louisiana Pigtoe in the historical shell collections at the Mississippi Museum of Natural Science-using the genetically confirmed specimens as reference. We expect the MDWFP will submit a public comment on the proposed listing rule during the public comment period that ends May 19, 2023, stating the records below Ross Barnett Reservoir should be considered in the assessment of critical habitat.

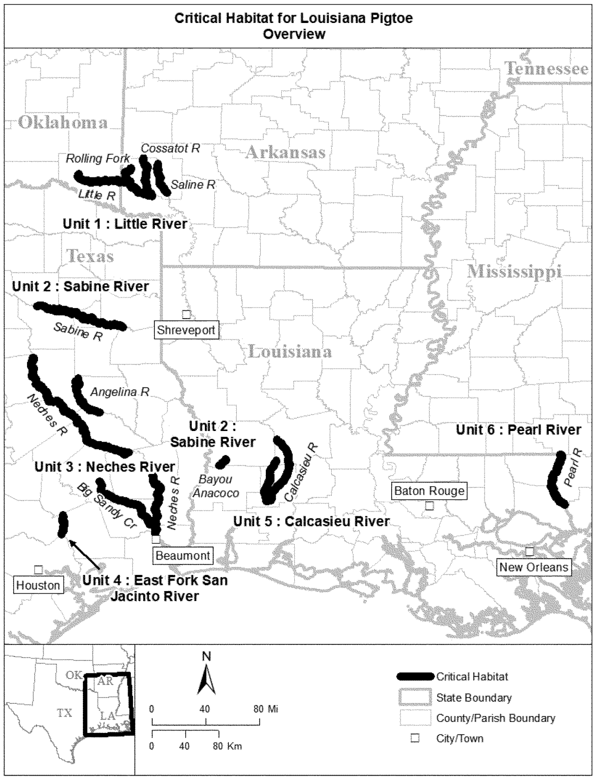
I've included a map of the Louisiana Pigtoe records relevant to the project action area below. Given the confirmed identifications and occurrence records, it would be prudent and appropriate to assume presence of Louisiana Pigtoe in the action area. We do not have estimates for the number of individuals that may occur in the action area as all the work has been qualitative. However, additional quantitative data may be available in June, which may be applicable to the action area.

Impacts to Louisiana Pigtoe as a result of the project are expected to occur. Dredging of the river will cause take of Louisiana Pigtoe in the action area, as mussels cannot move out of the way (see direct mortality section in the linked rule). Additionally, the species does not survive in still or low flow environments, so it would likely not persist or recolonize if the weir is constructed (see altered hydrology section in the linked rule). Additionally, if a weir is constructed the resulting reservoir will fragment the habitat and isolate the population (see habitat fragmentation section in the linked rule).

Potential conservation measures (or RPMs) for Louisiana Pigtoe may include:
Relocation of mussels to suitable habitat outside of the action area prior to dredging or inundation.
Creation of shoal habitat outside of the action area in an area with flow



Although the action area does not occur in proposed critical habitat units, we wanted to provide information regarding proposed critical habitat for Louisiana Pigtoe. The proposed critical habitat includes only the Pearl River south of Bogalusa (see below map from the proposed rule). We expect the Mississippi Department of Wildlife, Fisheries, and Parks will submit their data as new information during the public comment period on the proposed rule to be considered for the final critical habitat. The statutory deadline for the final listing rule and critical habitat designation is March 2024.



If you need additional information or just need to discuss with me feel free to reach out. I look forward to further coordination and partnership as we move ahead with this project.

Thanks,

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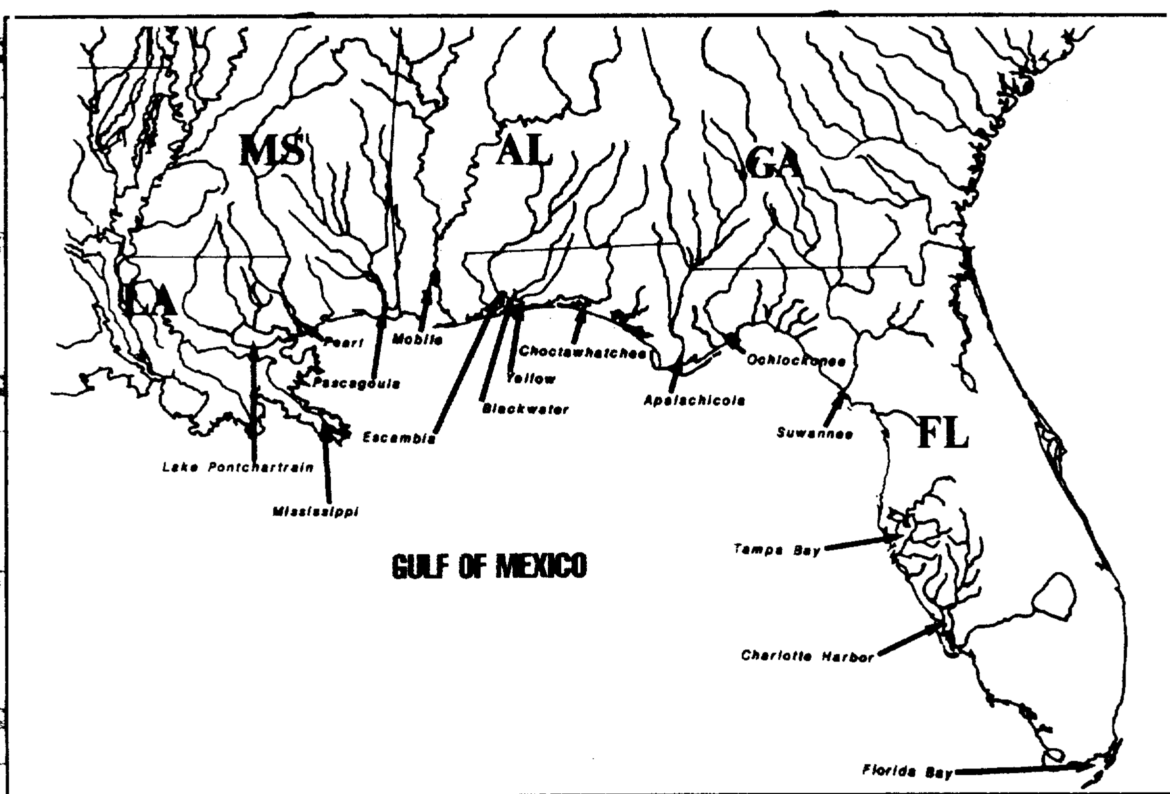
Pearl River Basin, Mississippi, Federal Flood Risk Management Project

Annex D3 - Species Recovery Plans and Status Assessment Reports

June 2024

GULF STURGEON
RECOVERY/MANAGEMENT PLAN



GULF STURGEON
(*Acipenser oxyrinchus desotoi*)

RECOVERY/MANAGEMENT PLAN

Prepared by

The Gulf Sturgeon Recovery/Management Task Team

for

Southeast Region
U.S. Fish and Wildlife Service
Atlanta, Georgia

and

Gulf States Marine Fisheries Commission
Ocean Springs, Mississippi

and

National Marine Fisheries Service
Washington, D.C.

Approved: Noreen K. Clough Date: 9/22/95
Noreen K. Clough
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Approved: Edwin J. Conklin Date: 9-18-95
Edwin J. Conklin
Chairman, Gulf States Marine Fisheries Commission

Approved: Rolland A. Schmitten Date: SEP 15 1995
Rolland A. Schmitten
Assistant Administrator for Fisheries, National Marine Fisheries Service

DISCLAIMER PAGE

Recovery plans delineate reasonable actions which are believed to be required to recover and/or protect listed species. Plans are published by the U.S. Fish and Wildlife Service and the National Marine Fisheries Service, sometimes prepared with the assistance of recovery teams, contractors, state agencies, and others. Objectives will be attained and any necessary funds made available subject to budgetary and other constraints affecting the parties involved, as well as the need to address other priorities. Recovery plans do not necessarily represent the views nor the official positions or approval of any individuals or agencies involved in the plan formulation, other than the U.S. Fish and Wildlife Service and the National Marine Fisheries Service. They represent the official position of the U.S. Fish and Wildlife Service and the National Marine Fisheries Service only after they have been signed by the Regional Director of the Fish and Wildlife Service and the Assistant Director for Fisheries of the National Marine Fisheries Service as approved. Approved recovery plans are subject to modification as dictated by new findings, changes in species status, and the completion of recovery tasks.

LITERATURE CITATIONS

Literature citations should read as follows:

U.S. Fish and Wildlife Service and Gulf States Marine Fisheries Commission. 1995. Gulf Sturgeon Recovery Plan. Atlanta, Georgia. 170 pp.

Additional copies of this plan may be purchased from:

Fish and Wildlife Reference Service:

5430 Grosvenor Lane, Suite 110
Bethesda, Maryland 20814
Telephone: 301/492-6403
or 1-800-582-3421

Fee for recovery plans vary, depending upon the number of pages.

ACKNOWLEDGEMENTS

The Gulf sturgeon would not have received federal protection without the dedication and persistence of a few individuals who raised our consciousness of the plight of this prehistoric species. Alan Huff completed the first life history of Gulf sturgeon and has since been influential in shaping recovery and restoration efforts. Dr. Archie Carr realized the magnificence of this subspecies, initiating sturgeon studies on the Apalachicola and Suwannee rivers. Each of his sons helped him through the years, the last being Stephen, who has continued the studies after Dr. Carr's death. Stephen's work has resulted in a long-term commitment to the subspecies. The Carrs were funded in their efforts by The Florida Phipps Foundation founded by Mr. John H. (Ben) Phipps. The Foundation continues to support Stephen Carr's work. Mr. Jim Barkuloo, while with the U.S. Fish and Wildlife Service (FWS), was instrumental in persuading the FWS to list the species. Dr. Michael Bentzien completed the tedious procedural work to list the subspecies and has continued to support the team's efforts in preparing the Recovery Plan. The Gulf States of Louisiana, Mississippi, Alabama, and Florida provided protection for the Gulf sturgeon before the subspecies was listed, by prohibiting take of "sturgeon." The states continue to provide protection through implementation of surveys and studies on the Gulf sturgeon so that management decisions can be based on scientific data.

Current Species Status: The current population levels of Gulf sturgeon in rivers other than the Suwannee and Apalachicola are unknown, but are thought to be reduced from historic levels. Historically, the subspecies occurred in most major rivers from the Mississippi River to the Suwannee River, and marine waters of the central and eastern Gulf of Mexico to Florida Bay.

Habitat Requirements and Limiting Factors: The Gulf sturgeon is an anadromous fish which migrates from salt water into large coastal rivers to spawn and spend the warm months. The majority of its life is spent in fresh water. Major population limiting factors are thought to include barriers (dams) to historical spawning habitats, loss of habitat, poor water quality, and overfishing.

Recovery Objectives: The short-term recovery objective is to prevent further reduction of existing wild populations of Gulf sturgeon. The long-term recovery objective is to establish population levels that would allow delisting of the Gulf sturgeon in discrete management units. Gulf sturgeon in discrete management units could be delisted by 2023, if the required criteria are met. Following delisting, a long-term fishery management objective is to establish self-sustaining populations that could withstand directed fishing pressure within discrete management units.

Recovery Criteria: The short-term recovery objective will be considered achieved for a management unit when the catch-per-unit-effort (CPUE) during monitoring is not declining from the baseline level over a 3 to 5-year period. This objective will apply to all management units within the range of the subspecies. Management units will be defined using an ecosystem approach based on river drainages, but may also incorporate genetic affinities among populations in different river drainages. Baselines will be determined by fishery independent CPUE levels.

The long-term recovery objective will be considered achieved for a management unit when the population is demonstrated to be self-sustaining and efforts are underway to restore lost or degraded habitat. A self-sustaining population is one in which the average rate of natural recruitment is at least equal to the average mortality rate in a 12-year period. While this objective will be sought for all management units, it is recognized that it may not be achievable for all management units. The long-term fishery management objective will be considered attained for a given management unit when a sustainable yield can be achieved while maintaining a stable population through natural recruitment. Note that the objective is not necessarily the opening of a management unit to fishing, but rather the development of a population that can sustain a fishery. Opening a population to fishing will be at the discretion of state(s) within whose jurisdiction(s) the management unit occurs. As with the long-term recovery objective, this objective may not be achievable for all management units, but will be sought for all units.

Priority 1 Recovery Tasks:

1. Develop and implement standardized population sampling and monitoring techniques (1.3.1).

2. Develop and implement regulatory framework to eliminate introductions of non-indigenous stock or other sturgeon species (2.5.3).

3. Reduce or eliminate incidental mortality (2.1.2).

4. Restore the benefits of natural riverine habitats (2.4.5).

5. Utilize existing authorities to protect habitat and where inadequate, recommend new laws and regulations (2.3.1).

Costs ($000's) of Priority 1 Tasks:

Year	Action 1	Action 2	Action 3	Action 4*	Action 5
FY 1	59	0	125	26	29
FY 2	73	25	125	48	29
FY 3	114	0	125	48	29
FY 4	108	0	75	31	29
FY 5	108	0	25	0	0

Cost of No. 1 Priority Actions: $1,231,000

* Actual restoration costs undetermined

Total Cost of Recovery: $8,413,000

Date of Recovery: Delisting should be initiated by 2023, for management units where recovery criteria have been met.

TABLE OF CONTENTS

TABLE OF CONTENTS (continued)

TABLE OF CONTENTS (continued)

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PREFACE

The U.S. Fish and Wildlife Service (FWS) and National Marine Fisheries Service (NMFS) jointly listed the Gulf sturgeon as threatened under the authority of the Endangered Species Act of 1973, as amended (ESA).

The FWS prepared a Report on the Conservation Status of the Gulf of Mexico Sturgeon *Acipenser oxyrhinchus desotoi* in 1988 as a precursor to the listing process. The Gulf States Marine Fisheries Commission (GSMFC) began an initiative in late 1990 to draft a fishery management plan for the Gulf sturgeon. The drafting team (ad hoc subcommittee of the GSMFC Technical Coordinating Committee, Anadromous Fish Subcommittee), on October 1, 1991, in response to the listing, took action to draft a management/recovery plan. This plan meets the requirements of a fisheries management plan as originally begun by the GSMFC, as well as the requirements associated with an Endangered Species Act recovery plan. The plan incorporates the format that has become standard in federal endangered and threatened species recovery plans in recent years. The FWS published a "Framework for the Management and Conservation of Paddlefish and Sturgeon Species in the United States" in March 1993. This document resulted from a workshop sponsored by the FWS that was attended by representatives of other federal agencies, the states, the private aquaculture community, and academia in January 1992. This recovery plan is consistent with the framework document, and in essence, steps down the recommendations and strategies contained therein.

The plan is intended to serve as a guide that delineates and schedules those actions believed necessary to restore the Gulf sturgeon as a viable self-sustaining element of its ecosystem. Some of the tasks described in the plan are ongoing by the FWS, GSMFC, NBS, and the states of Louisiana, Mississippi, Alabama, and Florida. The inclusion of these ongoing tasks represents an awareness of their importance, and offers support for their continuation. Because of this ongoing research on the subspecies, the plan incorporates personal communications and unpublished data.

LIST OF ABBREVIATIONS

ADCNR	Alabama Department of Conservation and Natural Resources
AGS	Alabama Geological Survey
ANSTF	Aquatic Nuisance Species Task Force
CCC	Caribbean Conservation Corporation
CES	Cooperative Extension Service
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
COE	U.S. Army Corps of Engineers
CWA	Clean Water Act
CZM	Office of Coastal Zone Management
EIRP	Environmental Impact Research Program
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
FDEP	Florida Department of Environmental Protection
FDNR	Florida Department of Natural Resources
FERC	Federal Energy Regulatory Commission
FGFC	Florida Game and Fresh Water Fish Commission
FRTES	Fisheries Resources Trace Elements Survey
FSBC	Florida State Board of Conservation
FWS	United States Fish and Wildlife Service
GCRL	Gulf Coast Research Laboratory
GSMFC	Gulf States Marine Fisheries Commission
GSRMA	Gulf States Resource Management Agencies (TX,LA,MS,AL,FL)
LDWF	Louisiana Department of Wildlife and Fisheries
MDWFP	Mississippi Department of Wildlife, Fisheries, and Parks
MMS	Minerals Management Service
NBS\BSC	National Biological Service, Southeastern Biological Science Center
NCSU	North Carolina Cooperative Research Unit, North Carolina State University
NGO	Nongovernmental organizations
NMFS	National Marine Fisheries Service
NRCS	Natural Resources Conservation Service (formerly SCS)
OCS	Outer Continental Shelf
SCS	Soil Conservation Service
TED	Turtle Excluder Device
USGS	United States Geological Survey
WES	Waterways Experiment Station
WSRFC	Warm Springs Regional Fisheries Center

LIST OF SYMBOLS

m	meter
mm	millimeter
cm	centimeter
kg	kilogram
km	kilometers
in	inches
ft	feet
mi	mile
lb	pound
hr	hour
°F	degrees Fahrenheit
°C	degrees Centigrade
ft/s	feet per second
m/s	meters per second
m^3/s	cubic meters per second
r	correlation coefficient
SD	standard deviation
TL	total length
FL	fork length
P	probability
hectare	not abbreviated
acre	not abbreviated

I. INTRODUCTION

NOMENCLATURE

The scientific name for Atlantic sturgeon is *Acipenser oxyrinchus* Mitchill. This species consists of two geographically disjunct subspecies: the Gulf sturgeon, *Acipenser oxyrinchus desotoi*, which inhabits the Gulf of Mexico watersheds, and the Atlantic coast subspecies, *Acipenser oxyrinchus oxyrinchus*.

Gilbert (1992) discovered that the species name of the Atlantic sturgeon has been "...misspelled for over one hundred years..." as *oxyrhynchus* rather than *oxyrinchus*. Consequently, based on the rules of zoological nomenclature, *oxyrinchus* is used throughout this plan.

Other colloquial names, in addition to Gulf sturgeon, are: Gulf of Mexico sturgeon, Atlantic sturgeon, common sturgeon and sea sturgeon.

TAXONOMY

Class: Osteichthyes
- Order: Acipenseriformes
  - Family: Acipenseridae
    - Genus: Acipenser
      - Species: oxyrinchus
        - Subspecies: desotoi

Type Specimens

The holotype was collected from the mouth of Singing River (West Pascagoula River) in Mississippi Sound off Gautier, Mississippi and is housed in the U.S. National Museum of Natural History, Washington, DC. The paratype was collected with the holotype and is deposited in the Chicago Natural History Museum (Vladykov 1955).

Current Taxonomic Treatment

The Gulf sturgeon is a member of the family Acipenseridae which inhabits the Atlantic, Gulf, Pacific and certain freshwaters of the United States (Ginsburg 1952). The family includes five members of the genus *Acipenser*, and three members of the genus *Scaphirhynchus*.

Other sturgeon likely to be found in the same waters with Gulf sturgeon include the pallid sturgeon, *Scaphirhynchus albus*, the shovelnose sturgeon, *S. platorynchus*, and Alabama sturgeon *S. suttkusi* (Rafinesque 1820; Forbes and Richardson 1908; Williams and Clemmer 1991). *Scaphirhynchus* are freshwater sturgeon that are native to the Mississippi and Mobile River systems. They formerly occurred in the upper Rio Grande River in New Mexico, but have not been recorded since 1874 (Lee et al., 1980). The fish are characterized by a flattened shovelshaped snout and are easily distinguished from Gulf sturgeon. *Acipenser oxyrinchus desotoi* is the only anadromous sturgeon occurring in the Gulf of Mexico.

Based on morphometrics, Wooley (1985) concluded that *A. o. desotoi* is a valid subspecies. Bowen and Avise (1990) analyzed the genetic structure of Atlantic and Gulf sturgeon using mitochondrial DNA (mtDNA) restriction fragment length polymorphism analysis, and postulated that relatively recent genetic contact had occurred between the two regions because of several shared mtDNA clones and clonal arrays. However, Ong et al. (manuscript submitted) used direct sequence analysis of the mtDNA control region and found three fixed nucleotide site differences between *A. oxyrinchus* from the Atlantic and Gulf coasts. They concluded that subspecific divisions are warranted for *A. oxyrinchus*, based on fixed genetic differences between the forms, their allopatric distributions, and their morphometric and life history differences. Ong et al. also postulated that their data, and those of Bowen and Avise (1990), indicate that the reproductive isolation between *A. o. desotoi* and *A. o. oxyrinchus* occurred because of climatic fluctuations in the Pleistocene in conjunction with related changes in the size of the Florida peninsula. Further, they noted that even if the two subspecies occasionally mix in ocean waters, the finding of fixed genetic differences between them suggests that homing fidelity is high in *A. oxyrinchus*.

STATUS

The U.S. Fish and Wildlife Service (FWS) and National Marine Fisheries Service (NMFS) designated the Gulf sturgeon to be a threatened subspecies, pursuant to the Endangered Species Act of 1973, as amended (ESA). The listing became official on September 30, 1991. As part of the listing, a special rule was promulgated to allow taking of the subspecies for educational purposes, scientific purposes, the enhancement of propagation or survival of the subspecies, zoological exhibition, and other conservation purposes consistent with the ESA. The special rule will allow conservation and recovery activities for Gulf sturgeon to be accomplished without a federal permit, provided the activities are in compliance with applicable state laws (FWS 1991a).

DESCRIPTION

Gulf sturgeon are anadromous fish with a sub-cylindrical body imbedded with bony plates or scutes. The snout is greatly extended and bladelike with four fleshy barbels in front of the mouth, which is protractile on the lower surface of the head. The upper lobe of the tail is longer than the lower lobe (Figure 1). The subspecies is light brown to dark brown in color and pale underneath (Vladykov 1955; Vladykov and Greeley 1963).

Characteristics common to both subspecies, *A. o. oxyrinchus* and *A. o. desotoi* are: Scutes strongly developed in longitudinal rows; 7 to 13 (average 9.8) dorsal shields; 24 to 35 (average 28.7) lateral shields behind dorsal fin in pairs; elongated fulcrum at base of lower caudal lobe decidedly longer than base of anal fin; head elongate; snout longer than postorbital distance in individuals up to 95.0 cm (38.0 in), but shorter than postorbital distance in older specimens (Vladykov and Greeley 1963).

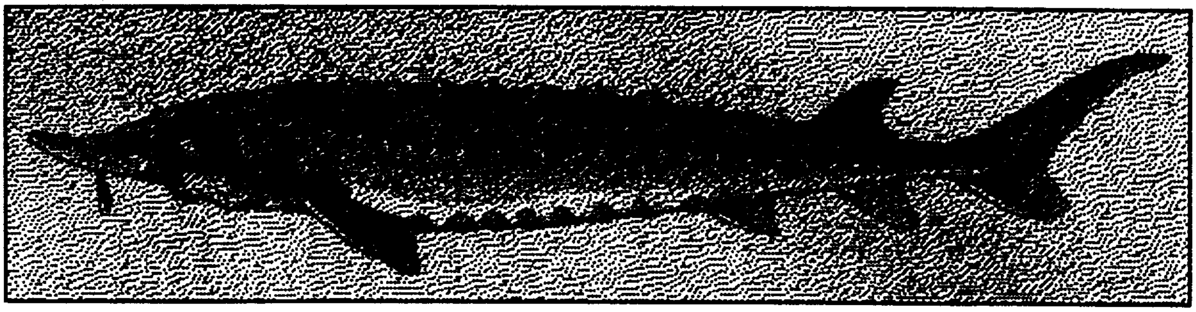


Figure 1: Gulf sturgeon *Acipenser oxyrinchus desotoi* (from Bigelow et al., 1963)

The most significant morphological characteristic to distinguish *A. o. oxyrinchus* from *A. o. desotoi* is the length of the spleen. Wooley (1985) found *A. o. desotoi* specimens had a mean spleen length versus fork length measurement of 12.3% (range 7.9 to 15.8%, SD 2.5, r = 0.212). *Acipenser o. oxyrinchus* specimens had a mean spleen length versus fork length (FL) measurement of 5.7% (range 2.8 to 8.3%, SD 1.8, r = 0.121) for a statistically significant difference ($P \leq 0.05$) and minimal overlap. He concluded that Gulf sturgeon and Atlantic sturgeon populations are allopatric and are sufficiently discrete to be considered distinct stocks for sturgeon population management.

POPULATION SIZE AND DISTRIBUTION

According to Wooley and Crateau (1985) Gulf sturgeon occurred in most major river systems from the Mississippi River to the Suwannee River, Florida and in marine waters of the Central and Eastern Gulf of Mexico south to Florida Bay (Figure 2). Comparison of historic information and current data indicates that Gulf sturgeon populations are reduced from historic levels (Barkuloo 1988). At present, Gulf sturgeon population estimates are unknown throughout its range; however, estimates have been completed for the Apalachicola and Suwannee rivers.

Extant Occurrences of Gulf Sturgeon

Offshore

A Gulf sturgeon was caught on hook and line in 1965 by Dianne Cox, a FWS employee. The 45.7-cm (18-in) Gulf sturgeon was caught in the Gulf of Mexico, 1.6 to 3.2 km (1 to 2 mi) east of Galveston Island in 6.1 m (20 ft) of water (Reynolds 1993).

The incidental catch of Gulf sturgeon in the industrial bottomfish (petfood) fishery in the north-central Gulf of Mexico from 1959 to 1963 was reported by Roithmayr (1965), based on the documentation of one juvenile specimen. The bottomfish fishery worked an area between Point au Fer, Louisiana and Perdido Bay, Florida from shore to 55 m (180 ft).

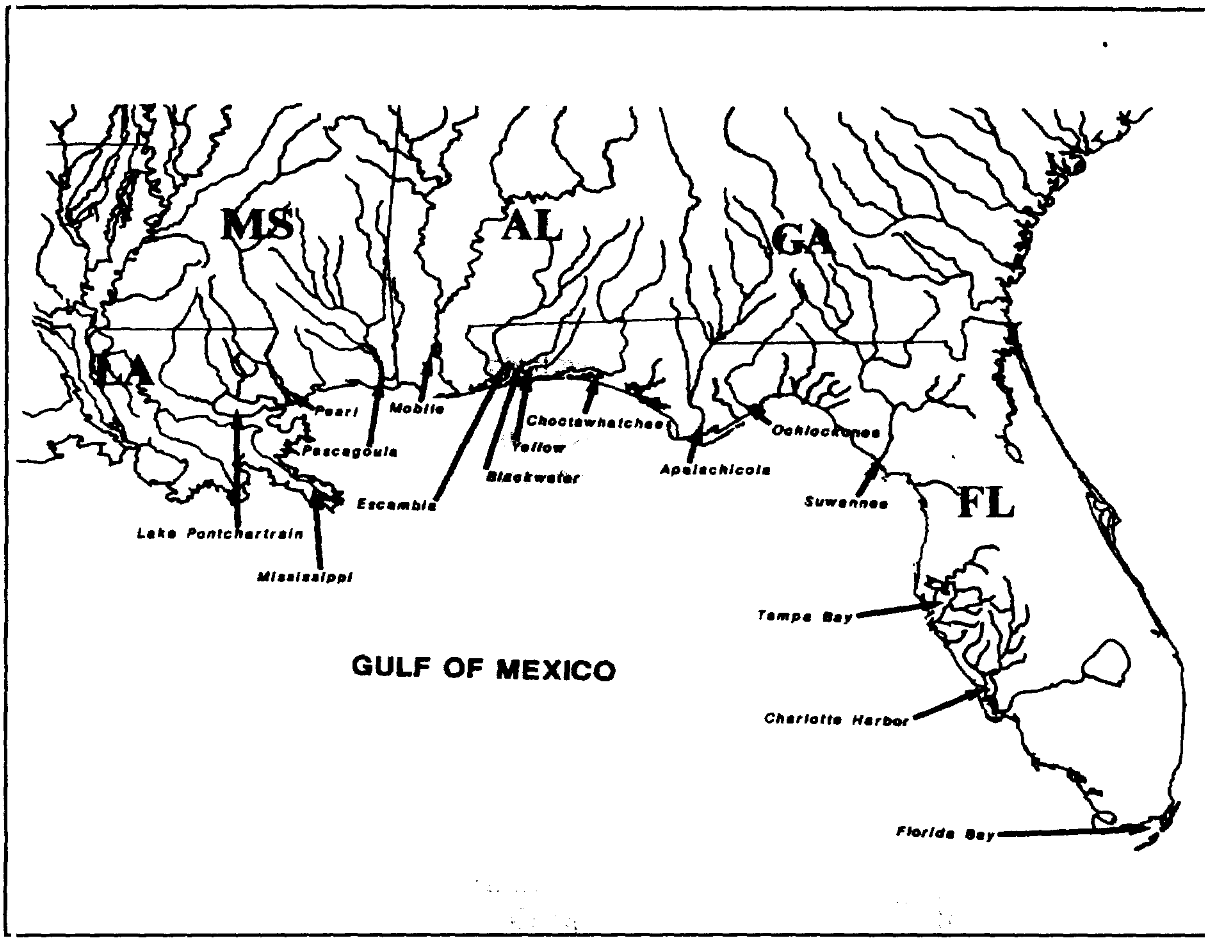


Figure 2: Range of the Gulf Sturgeon

Mermantau River Basin

Mermantau River: The Louisiana Department of Wildlife and Fisheries (1979) reported that an Atlantic sturgeon was caught by a Mr. Hugh Mhire in an otter trawl while shrimping in the Gulf off the mouth of the Mermentau River, Cameron Parish. This specimen was probably a Gulf sturgeon.

Mississippi River Basin

A photograph of a "sea" sturgeon captured at the mouth of the Mississippi River was shown in Fishes and Fishing in Louisiana (1965). Reynolds (1993) reported that a sturgeon measuring 282 cm (111.0 in) and weighing 228.2 kg (503.0 lb) was caught at the mouth of the Mississippi River at Cow Horn Reef in September of 1936.

Mississippi River: A Gulf sturgeon was caught by a commercial fisherman in the auxiliary outflow channel between river km 500.3 (river mi 311.0) of the Mississippi River and river km

16.09 (river mi 10.0) of the Red River on March 28, 1994 (G. Constant, personal communication). The Gulf sturgeon weighed 28.8 kg (63.5 lb) and was 151.2 cm (59.5 in) length and was caught in a 1.2 m (4.0 ft) hoop net.

Lake Pontchartrain Basin

Lake Pontchartrain/Lake Borgne/Rigolets: The Louisiana Department of Wildlife and Fisheries (LDWF) collected twelve Gulf sturgeon weighing 0.22 to 9 kg (0.5 to 19.8 lb) April through June of 1993 (H. Rogillio, personal communication). During a study from January 1990 to March 1993, LDWF collected and tagged 19 Gulf sturgeon weighing 0.25 to 14.5 kg (0.6 to 32.0 lb) from Lake Pontchartrain, Lake Borgne, and the Rigolets (Rogillio 1993). Commercial and sport fishermen incidentally caught 177 Gulf sturgeon measuring up to 220.0 cm (86.6 in) in length and weighing from 1.0 to 68.0 kg (2.2 to 149.9 lb) from Lake Pontchartrain from October 1991 to September 1992 (Rogillio 1993). Reynolds (1993) reported that sturgeon measuring up to 220.0 cm (86.6 in) in length and weighing up to 117.3 kg (258.0 lb) were incidentally caught by shrimp trawlers, netters and recreational anglers from 1989 to 1993 in Lake Pontchartrain. A specimen weighing 53.6 kg (118 lbs) was caught by a hook-and-line fisherman in 1986 (Sentry News 1986). Davis et al. (1970) reported that sturgeon were collected from Lake Ponchartrain during an anadromous fish survey from 1966 to 1969.

Tchefuncte River: Commercial gillnetters incidentally caught 15 Gulf sturgeon weighing from 1.0 to 18.0 kg (2.2 to 39.7 lb) between February and March 1991 in the mouth of the river (H. Rogillio, personal communication). Davis et al. (1970) reported that Gulf sturgeon were collected in trammel nets from the Tchefuncte River during an anadromous fish survey conducted from 1966 to 1969.

Tickfaw River: Davis et al. (1970) reported the collection of sturgeon in trammel nets from the Tickfaw River during an anadromous fish survey from 1966 to 1969.

Tangipahoa River: Davis et al. (1970) reported that sturgeon were collected in trammel nets from the Tangipahoa River during an anadromous fish survey from 1966 to 1969.

Amite River: Davis et al. (1970) reported catch of a sturgeon by a commercial fisherman from the Amite River. Identification of the fish was confirmed by the fisheries biologists with the Louisiana Wild Life (sic) and Fisheries Commission who were conducting an anadromous fish survey.

Pearl River: Esher and Bradshaw (1988) and Bradshaw (personal communication) gill netted a Gulf sturgeon in May 1988 in the lower Pearl River. Sixty-three Gulf sturgeon ranging from juvenile to subadult size were collected from river mile 20 of the Pearl River in 1985 (F. Petzold, personal communication). A 72.7 kg (160.3 lb) female Gulf sturgeon was caught just south of Jackson, Mississippi in 1984 by Miranda and Jackson (1987). The FWS donated a Gulf sturgeon caught by a commercial fisherman in the Pearl River at Monticello to the Mississippi Museum of Natural Science Fish Collection

(MMNS 20206) in 1982 (C. Knight, personal communication; W. McDearman, personal communication). The MDWFP measured and photographed a 119.0 kg (263.0 lb) Gulf sturgeon, 2.2 m (7.25 ft) in length taken by a commercial fisherman below the Ross Barnett Reservoir spillway in 1976 (W. McDearman, personal communication). McDearman and Stewart (personal communication) also note that in the Pearl River between Georgetown and Monticello, Mississippi, there is an area where 2 to 3 Gulf sturgeon are routinely reported by commercial fisherman every 4 to 5 years. In 1971 a Gulf sturgeon from the Pearl River was examined as part of a parasite study (N. Jordan, personal communication). Davis et al. (1970) reported the catch of Gulf sturgeon in hoop nets from the Pearl River at Highway 90 during an anadromous fish survey from 1966 to 1969. The Gulf sturgeon ranged in size from 15.2 cm (6.0 in) to 187.9 cm (74.0 in).

Middle Pearl River: Two Gulf sturgeon were collected in the Middle West Pearl River, St. Tammy Parish, Louisiana, one on March 1, 1995, and the other on March 2, 1995, by the U.S. Army Corps of Engineers, Waterways Experiment Station (WES). The Gulf sturgeon were collected in gill nets and the first sturgeon caught weighed 0.28 kg (0.62 lb) and measured 36.2 cm (14.3 in) in total length. The second Gulf sturgeon weighed 0.28 kg (0.62 lb) and measured 43.5 cm (17.1 in) in total length. Both fish were tagged with Peterson discs and released (M. Chan, personal communication).

Louisiana Department of Wildlife and Fisheries personnel collected 77 Gulf sturgeon from the west Middle Pearl River in 1994 (H. Rogillio, personal communication). The fish ranged in length from 45.7 to 165.1 cm (18 to 65 in). The majority of the fish (84 percent) ranged in length from 74.0 to 114.3 cm (29 to 45 in). The LDWF also collected 14 Gulf sturgeon weighing 1.5 to 14.5 kg (3.3 to 32 lb) in the Middle and west Middle Pearl River from June 1992 through June 1993 (H. Rogillio, personal communication). Two of those specimens were tagged with radio tags. The LDWF also collected 13 Gulf sturgeon weighing 0.27 to 4.3 kg (0.6 to 9.5 lb) in the Middle Pearl River (Drumhole) from April to May 1992 (Rogillio 1993). Commercial fishermen caught one Gulf sturgeon weighing 45.0 kg (99.2 lb) in the Middle Pearl River in February 1991.

Bogue Chitto: Three Gulf sturgeon were also captured by LDWF in the Bogue Chitto River below the Bogue Chitto sill in 1993. The Gulf sturgeon weighed from 2.9 to 4.5 kg (6.5 to 14.5 lb) (H. Rogillio, personal communication).

East Pearl River: Biologists with the FWS gill netted a Gulf sturgeon from the Mikes River, a tributary to the East Pearl River during a fishery survey in the spring of 1992. The fish was 0.7 m (2.3 ft) in length (P. Douglas, personal communication). Davis et al. (1970) reported that one sturgeon was collected in a trammel net from the East Pearl River on November 1, 1968 during an anadromous fish survey conducted from 1966 to 1969.

West Pearl River: Commercial fishermen caught five Gulf sturgeon weighing from 0.1 to 0.3 kg (0.22 to 0.66 lb) in the West Pearl River in October 1990 (H. Rogillio, personal communication).

Mississippi Sound

Bradshaw (personal communication) reported three tag returns from Gulf sturgeon that were incidentally caught by shrimpers working in Mississippi Sound during the fall of 1985. Bradshaw originally collected these Gulf sturgeon from river km 32 (river mi 20) on the Pearl River earlier in 1985. He also noted finding three dead Gulf sturgeon incidentally caught by gillnetters in the western part of the Sound and revived another Gulf sturgeon a gillnetter had caught "on" Horn Island in 1989. Five Gulf sturgeon from Mississippi Sound near Horn Island were examined as part of a parasite study (N. Jordan, personal communication). Of the five sturgeon, one was examined in each of the years 1973, 1976, and 1977, and two in 1982. One Gulf sturgeon [Gulf Coast Research Laboratory (GCRL) #1711] was incidentally caught in a shrimp trawl off the east end of Deer Island in Mississippi Sound in November 1966 in approximately 5.5 m (18 ft) of water. The Gulf sturgeon had a total length (TL) of 75.2 cm (29.6 in). Near this same location J.Y. Christmas (personal communication) reported catching one Gulf sturgeon (GCRL #28) with a TL of 55.2 cm (21.7 in) while sampling with a shrimp trawl in March 1960.

Biloxi Bay

One Gulf sturgeon was incidentally caught in a shrimp trawl in Biloxi Bay off Marsh Point on November 19, 1960 (GCRL #337). The fish was 55.5 cm (22.0 in) TL.

Pascagoula River Basin

Pascagoula Bay: Shepard (personal communication) caught two Gulf sturgeon at the mouth of Bayou LaMotte during the winters of 1991 and 1992 while gillnetting for the J.L. Scott Marine Education Center (GCRL). Reynolds (1993) reported commercial fishermen collecting Gulf sturgeon in and near the mouth of the Pascagoula River in the late 1980's and early 1990's. Shepard (personal communication) reports catching nine Gulf sturgeon from the mouth of the West Pascagoula River while gillnetting from 1983 to 1984. All but one of the sturgeon were caught at the mouth of Bayou LaMotte. The ninth fish was captured near the Sandalwood Canal. One Gulf sturgeon from the mouth of the Pascagoula River was examined in 1970 as part of as part of a parasite study conducted by GCRL (N. Jordan, personal communication).

Pascagoula River: Murphy and Skaines (1994) reported collection of seven Gulf sturgeon in the lower three miles of the Pascagoula River from April to June 1993. Two were radio tagged and released. The fish ranged in length from 46.4 to 111.8 cm (18.3 to 44.0 in) and from 0.8 to 10.4 kg (1.8 to 22.9 lb) in weight. Miranda and Jackson (1987), collected a 78.2 cm (30.8 in) Gulf sturgeon in June 1987 during 30 net-nights from the river. Three Gulf sturgeon were examined from the Pascagoula River as part of a parasite study conducted by GCRL. One was examined in 1978, the second in 1982 and the third in 1984 (N. Jordan, personal communication).

Chickasawhay River: Miranda and Jackson (1987) reported a catch of a 56.7 kg (125.0 lb) Gulf sturgeon in 1985 from the Chickasawhay River, which is a tributary of the Pascagoula River.

Leaf River: Murphy and Skaines (1994) reported that one of two fish radio-tagged from the lower Pascagoula River in May 1993 was located twice in September of that year. The last documented location of the fish was in the Leaf River three miles downstream from McLain, Mississippi approximately 123.8 km (77.0 mi) from its site of capture.

West Pascagoula River: Two Gulf sturgeon from the West Pascagoula River were examined in 1973 and 1979 as part of a parasite study conducted by GCRL (N. Jordan, personal communication). In December 16, 1964, a Gulf sturgeon (GCRL #4501) was collected by T.D. McIlwain in Big Lake off the West Pascagoula River. The sturgeon weighed 0.24 g (0.52 lb) and was 45.6 cm (18.0 in) TL. The water temperature was 13.9°C (57.0°F) with a salinity of 1.1 ppt.

Mobile River Basin

Mobile Bay: A live Gulf sturgeon was picked up on the shoreline of Bayou LaBatre by a fisherman on March 8, 1993 (F. Parauka, personal communication). The fish was 127 cm (50 in) long and weighed 12.5 kg (27.5 lb). The fish was held for observation at the Dauphin Island Sealab until a FWS biologist measured, weighed, radio-tagged, and collected genetic tissue samples and released it into Mobile Bay a day later. Efforts to locate the sturgeon again were unsuccessful. In July 1972 approximately one hundred Gulf sturgeon were observed at the mouth of the Blakeley River in eastern Mobile Bay feeding in shallow water (Vittor 1972). The sturgeon were approximately .91 m (3 ft) in length.

Mobile River: A Gulf sturgeon about 150 cm (59.1 in) long was sighted in the Mobile River near the head of Mobile Bay on October 3, 1992 by an Alabama Department of Conservation and Natural Resources (ADCNR) Marine Resources Division employee. There is a mounted specimen of a juvenile Gulf sturgeon at the Roussos Restaurant in Mobile, Alabama (J. Roussos, personal communication). The specimen is approximately 45.7 to 50.8 cm (18 to 20 in) TL and was collected in 1985 or 1986. The specimen was caught in a shrimp trawl in the Mobile River, presumably at the north end of Mobile Bay.

Tensaw River: The ADCNR reported that a commercial fisherman incidentally caught a 180 cm (70.9 in) Gulf sturgeon in the mouth of the Tensaw River in September 1991 (W. Tucker, personal communication). M. Mettee (personal communication) reported a 180 cm (70.9 in) Gulf sturgeon was incidentally netted and released in the Tensaw River in April 1986 by a commercial fisherman.

Blakeley River: Commercial gillnetters incidentally caught Gulf sturgeon in the Blakely River during the fall from 1989 to 1991.

Tombigbee River: A specimen caught in June 1987 upstream of Coffeeville on the Tombigbee River was verified by an Alabama Geological Survey (AGS) biologist as *Acipenser* (M. Mettee, personal communication). In 1977 a Gulf sturgeon from the Tombigbee River was examined as part of a parasite study (N. Jordan, personal communication). Incidental catches of Gulf sturgeon still occur annually from the Tombigbee River in the remaining riverine habitat below Coffeeville dam (J. Duffy, personal communication).

Alabama River: Incidental catches of Gulf sturgeon still occur annually from the Alabama River in the remaining riverine habitat below Claiborne dam (J. Duffy, personal communication).

Pensacola Bay Basin

Pensacola Bay: A 56.0 cm (22.0 in) TL Gulf sturgeon was collected in Pensacola Bay on January 20, 1978 (Collection No. 10319, Florida Department of Environmental Protection, FDNR).

Escambia River: Two Gulf sturgeon were collected, tagged and released in the Escambia River about 1.6 km (1.0 mi) downstream of highway 184 bridge in September 1994 by the FWS (F. Parauka, personal communication). The fish weighed 15.5 and 20.7 kg (34.0 and 45.5 lb). Incidental catches of Gulf sturgeon have been reported for the Escambia River (G. Bass, personal communication). Recreational anglers reported that prior to 1980 they would see as many as 10 Gulf sturgeon jumping in the river but now it is rare to see even one fish jump during a fishing trip (Reynolds 1993). Prior to a Florida law prohibiting sturgeon fishing in 1984, a limited commercial fishery existed on that river (National Marine Fisheries Service 1987).

Conecuh River: Annual sightings are reported from the Conecuh River in south central Alabama (J. Duffy, personal communication).

Blackwater River: Three Gulf sturgeon were collected in the Blackwater River during a Florida Game and Fresh Water Fish Commission (FGFC) striped bass netting project in March 1991. The fish weighed from 5.0 to 12.0 kg (11.0 to 26.5 lb) (FGFC, unpublished data).

Yellow River: Eighteen Gulf sturgeon were collected, tagged and released in the Yellow River below Boiling Lake in July 1993 by the FWS (F. Parauka, personal communication). The fish weighed from 5.8 to 63.6 kg (12.7 to 140.0 lb). Gulf sturgeon were collected in the Yellow River during a 1961 to 1962 survey by FGFC (1964). Commercial landings were occasionally reported prior to the 1984 fishing prohibition (J. Barkuloo, personal communication).

Choctawhatchee Bay Basin

Santa Rosa Sound: The U.S. Environmental Protection Agency (EPA) reported a 23 kg (50 lb) Gulf sturgeon washed up on the beach in Santa Rosa Sound near Navarre, Florida in 1988 (F. Parauka, personal communication).

Choctawhatchee Bay: Four Gulf sturgeon were collected by FDEP biologists on April 27, 1993 from Jolly Bay at the eastern end of Choctawhatchee Bay. The sturgeon ranged in length from 41.2 to 81.9 cm (16.22 to 32.2 in).

Choctawhatchee River: Fifty adult and subadult Gulf sturgeon were collected, tagged and released at the mouth of the Choctawhatchee River in April 1994 by the North Carolina Cooperative Research Unit, North Carolina State University (NCSU) and the FWS (Potak et al. 1995). Twenty-five of the fish were equipped with radio tags. The fish weighed from 2.5 to 72.7 kg (5.5 to 160.3 lb) and ranged in length from 73.8 to 192.0 cm (29.1 to 75.6 in). Twenty-seven Gulf sturgeon were captured, tagged, and released in the Choctawhatchee River between Howell Bluff and Rocky Landing in 1988, 1990, and 1991 by the FWS (FWS 1988, 1990, 1991b). The fish weighed from 4.5 to 52.3 kg (9.9 to 115.3 lb). In addition, a 0.13 kg (0.29 lb) specimen caught by an angler downstream from Caryville, Florida in 1991 was tagged and released by the FWS (FWS 1991b). Three Gulf sturgeon weighing from 17.0 to 26.0 kg (37.5 to 57.3 lb) were collected in the upper Choctawhatchee River below its confluence with Pea River at Geneva, Alabama in August 1991 by the FWS (FWS, unpublished data). Annual sightings are reported from the Choctawhatchee River in south central Alabama (J. Duffy, personal communication).

> **Pea River:** Three Gulf sturgeon 91.0 to 213.0 cm (35.8 to 83.9 in) in length were collected by the AGS during March 1992 about 1.0 to 3.0 km (0.62 to 1.86 mi) in the Pea River above its confluence with the Choctawhatchee River (M. Mettee, personal communication). Annual sightings are reported from the Pea River in south central Alabama (J. Duffy, personal communication).

Apalachicola, Chattahoochee, Flint River Basin

Apalachicola Bay: A 34.0 kg (74.8 lb) Gulf sturgeon was caught by a commercial fisherman in a shrimp trawl in Apalachicola Bay in November 1989 (F. Parauka, personal communication). The fish was taken to the Apalachicola National Estuarine Reserve for observation and was later tagged and released at the point of capture by the FWS. A 34.5 kg (76.0 lb) Gulf sturgeon was captured, tagged and released in Apalachicola Bay, south of Hwy 98 bridge in March 1988. Also, in March 1987, a 34.0 kg (74.6 lb) Gulf sturgeon was captured, tagged and released in Apalachicola Bay, north of Hwy 98 bridge (F. Parauka, personal communication). Incidental captures by commercial shrimpers and gill net fishermen in Apalachicola Bay were noted by Wooley and Crateau (1985) and reported by Swift et al. (1977).

Apalachicola River: The FWS Panama City, Florida Field Office has monitored the Apalachicola River Gulf sturgeon population since 1979. Three-hundred and fifty Gulf sturgeon were collected below Jim Woodruff Lock and Dam (JWLD), tagged and recaptured from May through September, 1981 through 1993. The number of fish staying below the dam in the summer was estimated using a modified Schnabel method. Fish smaller than 45.0 cm (17.7 in) TL were excluded because of sampling bias caused by net selectivity. Since 1984, the estimated annual number of fish ranged from 96 to 131 with a mean of 115 (FWS 1990, 1991b, 1992). A 145 cm (57.1 in) FL specimen was captured by FDEP (FSBC 640008) on October 28, 1970 in the river. The FGFC (1964) collected Gulf sturgeon during their anadromous fish survey conducted from 1954 to 1964.

A report of the U.S. Commission on Fish and Fisheries (1902) indicated the Apalachicola River provided the largest and most economically important commercial sturgeon fishery in Florida in 1901. Archie Carr (personal communication) noted that 32 families commercially fished for Gulf sturgeon in the mid-1940's. A commercial fishery continued until the late 1970's with only a few families. Sport fishing for Gulf sturgeon in the spring, and to a lesser extent in the fall, in some of the deeper holes in the Apalachicola River below the JWLD produced fish up to 73 kg (160.9 lb) and 2.3 m (7.5 ft) long (Tallahassee Democrat 1958, 1963, 1969).

> **Brothers River:** Archie Carr (1978 and personal communication) began studying Gulf sturgeon in the Apalachicola River in 1975 and caught only eight sturgeon in 23 days of set-netting in Brothers Creek.

Flint River: Swift et al. (1977) noted a report of a 209 kg (460.8 lb) specimen from the Flint River near Albany, Georgia before 1950, prior to the completion of JWLD in 1957.

Ochlockonee River Basin

Ochlockonee River: Four Gulf sturgeon weighing from 2.0 to 4.0 kg (4.4 to 8.8 lb) were collected in the lower Ochlockonee River at the mouth of Womack Creek in June 1991 (FWS/Panama City and National Biological Survey/Southeastern Biological Service Center-Gainesville (NBS/SBSC-G), unpublished data). Gulf sturgeon were commercially fished in the vicinity of Hitchcock Lake in Wakulla County (Swift et al., 1977; Florida Outdoors 1959). The fish were shipped to the town of Apalachicola for processing and sale to the New York City area. Commercial landings comparable to the Apalachicola River fishery were noted in 1901 (U.S. Commission on Fish and Fisheries 1902). However, most commercial fishing for Gulf sturgeon in the river ended in the early 1970's (F. Parauka, personal communication).

Suwannee River Basin

Suwannee River: The Suwannee River appears to support the most viable Gulf sturgeon population among the coastal rivers of the Gulf of Mexico (Huff 1975). The Caribbean Conservation Corporation (CCC) has captured, marked, and released 1,670 spring migrating Gulf sturgeon at the river mouth since 1986. Based on the recapture of marked fish, the annual estimated population size ranged between 2,250 to 3,300 for Gulf sturgeon averaging about 18 kg (39.7 lb) (Carr and Rago, unpublished data). An ongoing complementary study by the NBS/BSC-G (unpublished data) has captured, marked, and released about 1,500 subadults, most of which were less than 15 kg (33.1 lb), throughout the river from March 1988 through March 1992. This river supported a limited commercial Gulf sturgeon fishery from 1899 (U.S. Commission on Fish and Fisheries 1902) until 1984 when the State of Florida prohibited harvest and possession.

Tampa Bay Basin

Tampa Bay: A commercial netter incidentally caught and released a Gulf sturgeon 56.4 cm (1.8 ft) in length, one mile west of Redington Beach near St. Petersburg in December 1992 (Reynolds 1993). Before this time, the most recent Gulf sturgeon catch reported from Tampa Bay was a 144 cm (56.7 in) FL female weighing 25.8 kg (56.9 lb), collected on December 11, 1987 near Pinellas Point (FDEP fish collection records, no collection number). Tampa Bay was the location of the first recorded significant sturgeon fishery on the Gulf of Mexico coast, lasting only three years (U.S. Commission on Fish and Fisheries 1902). The fishery began in 1886-1887 with a catch of 1,500 fish yielding 2,268 kg (5,000 lb) of roe. Two thousand fish and 2,858 kg (6,300 lb) of roe were marketed in 1887-1888. The fishery ended after the 1888-1889 season when only seven sturgeon were caught. Sturgeon catches have been reported sporadically since 1890.

Charlotte Harbor Basin

Charlotte Harbor: A 3.0 kg (6.6 lb) Gulf sturgeon was captured by a commercial mackerel net fisherman near the mouth of Charlotte Harbor on January 29, 1992 (R. Ruiz-Carus, personal communication). The sturgeon was caught on a sand bar near Boca Grande Pass, 2.4 to 3.0 m (7.9 to 9.8 ft) in depth. While specific information was given for this fish, the fishermen related that two or three sturgeon of the same size were released alive from the same net set near Boca Grande Pass. Two other specimens have been reported from Charlotte Harbor (University of Florida/Florida State Museum (UF/FSM) 35332; FSBC 18077), one of which is a 24.3 kg (53.6 lb) specimen now mounted at the Florida Marine Research Institute, FDEP, St. Petersburg, Florida.

BIOLOGICAL CHARACTERISTICS

Habitat

Gulf sturgeon are classified as anadromous, with immature and mature fish participating in freshwater migrations (Huff 1975; Carr 1983; Wooley and Crateau 1985; S. Carr, unpublished data; J. Clugston, unpublished data). Anecdotal information, gillnetting, and biotelemetry have shown that subadults and adults spend eight to nine months each year in rivers and three to four of the coolest months in estuaries or Gulf waters. It appears that Gulf sturgeon less than two years old remain in riverine habitats and estuarine areas throughout the year. Many Gulf sturgeon in the Suwannee River spend summer months near the mouths of springs and cool-water rivers (Foster 1993; S. Carr, unpublished data). The substrate of much of the Suwannee River is sand and limerock, especially in those areas near springs and spring runs.

Wooley and Crateau (1985) reported that Gulf sturgeon in the Apalachicola River utilized the area immediately downstream from JWLD from May through September. The area occupied consisted of the tailrace and spillway basin of JWLD and a large scour hole below the lock. During high flow periods in the late spring when water was passing through open water control gates at JWLD, Gulf sturgeon would congregate in the turbulent flow, often suspended just below the water surface. During the summer, Gulf sturgeon concentrated in the large scour hole below the lock and in the area of the dam spillway basin. This area represented the deepest available water within 25 km (15.5 mi) down-river of the JWLD. Mean total distance moved by Gulf sturgeon during this time was only 0.4 km (0.25 mi). In all cases Gulf sturgeon did not move more than 0.8 km (0.5 mi) from May through September. The area consisted of sand and gravel substrate, water depths ranged from 6.0 to 12.0 m (19.7 to 39.4 ft) with a mean depth of 8.4 m (27.6 ft) and velocities ranged from 60.0 to 90.0 cm/s (2.0 to 3.0 ft/s) with a mean velocity of 64.1 cm/s (2.1 ft/s). Because of the scarcity of historical biological data pertaining to the Gulf sturgeon in the Apalachicola River it is impossible to ascertain whether the area observed as a summer congregation area represents specific historic habitat. It may be the best alternative habitat type available to Gulf sturgeon whose migration upstream was blocked by the construction of JWLD in 1957.

The U.S. Army Corps of Engineers (COE) conducted surveys in this area in November 1991 and October 1992, to characterize flows associated with a strong cross current at the lock approach. In November 1991, velocities were measured at a depth 0.06 and 0.24 m (0.2 and 0.8 ft) of the water column, with velocities ranging from 0.19 to 0.67 m/s (0.61 to 2.19 ft/s) during normal powerhouse generation (two turbines on line with trash gate open). The follow-up survey in October 1992 included an additional measurement within the large scour hole below the lock at a depth within 0.6 m (2 ft) of the bottom. Velocities ranged from 0.08 to 0.92 m/s (0.25 to 3.01 ft/s) for normal powerhouse generation (with or without the trash gate open; with velocities at the bottom of the scour hole ranging from 0.11 to 0.37 m/s (0.36 to 1.2 ft/s) (COE 1993; COE 1994).

The Brothers River, a tributary entering the lower Apalachicola River at river km 19.3 (river mi 12.0) appears to be a staging area for Gulf sturgeon leaving the river (Odenkirk 1989). This was a favorite location for commercial Gulf sturgeon netting in past years (J. Fichera, personal communication). The Brothers River is a sluggish river with deep holes, swampy banks, and a sand and rock bottom. Wooley and Crateau (1985) characterized the habitat as having a mean depth of 11.0 m (36.1 ft), water depths ranged from 8.0 to 18.0 m (26.2 to 59.0 ft) and velocities ranged from 0.58 to 0.75 m/s (1.9 to 2.46 ft/s) with a mean velocity of .60 m/s (1.97 ft/s).

Swift et al. (1977) reported that local fishermen believed that Gulf sturgeon spawning occurred in June in the deeper holes and "lakes" along the rivers. Swift also reported that Gulf sturgeon were caught by sport fisherman from deep holes in the Apalachicola River below Jim Woodruff Dam during the spring and fall in the late 1950's to the late 1960's.

The WES reported the river conditions during collection of two Gulf sturgeon from the west Middle Pearl River on March 1, 1995. The conditions for at the surface and in 7.62 m (25 ft) of water were: temperature of 15.3°C (59.6°F) and 15.3°C (59.5°F); conductivity of 68 μmho's/cm; dissolved oxygen of 9.09 and 8.80 mg/l; pH of 6.64 and 6.57; and turbidity at the surface of 32 NTU (M. Chan, personal communication).

Bradshaw (personal communication) noted that 62 of 63 of the Gulf sturgeon collected from the East Pearl River at river km 32.2 (river mi 20) in 1985 were from one location, a deep, 12.2 m (40 ft) hole. He also reported that another Gulf sturgeon was captured at the same location in 1988.

Swift et al. (1977) noted that young Gulf sturgeon were reportedly captured in shrimp trawls in Apalachicola Bay. Muddy, soft bottom substrates, the dominant habitat of the Bay, comprise about 78% of the open water zone (Livingston 1984). Wooley and Crateau (1985) reported one Gulf sturgeon was captured 3.2 km (2.0 mi) from the mouth of Apalachicola River in the Bay in approximately 2 m (6.6 ft) depth over a mud substrate. Several Gulf sturgeon were collected from Gulf waters adjacent to Apalachicola Bay (Wooley and Crateau 1985). One Gulf sturgeon was caught 1.2 km (.75 mi) south of Cape St. George in 6 m (19.7 ft) of water and another Gulf sturgeon was captured 1.6 km (1.0 mi) south of Cape San Blas in 15 m (49.2 ft) of water. Limited stomach analyses from Suwannee and Apalachicola River Gulf sturgeon indicate that mud and sand bottoms and seagrass communities are probably important marine habitats for Gulf sturgeon (Mason and Clugston 1993).

Migration and Movement

The movements of Gulf sturgeon in the Apalachicola, Suwannee, Pearl, and Choctawhatchee rivers have been and are being monitored by ultrasonic and radio telemetry and by conventional fish sampling gear (Foster 1993; Carr 1983; Wooley and Crateau 1985; Odenkirk 1989; Rogillio 1993; Clugston et al., in press; Potak et al. 1995; S. Carr, unpublished data; Odenkirk et al., unpublished manuscript; F. Parauka, personal communication; H. Rogillio, personal communication). In general, subadult and adult Gulf sturgeon began to migrate into rivers from the Gulf of Mexico as river temperatures increased to about 16 to 23°C (60.8 to 75.0°F). They continued to immigrate through early May, but most arrive when temperatures reach 21°C. Gulf sturgeon have been collected as far upstream as river km 221 (river mi 137.3) in the Suwannee River. In the Suwannee River, most radio-tracked Gulf sturgeon appeared to settle into four 3.0 to 15.0 km (1.9 to 9.3 mi) long reaches of the river during the summer (Foster 1993). Upstream migration in the Apalachicola River is blocked at river km 171 (river mi 106.3) by the JWLD. Nearly all radio-tracked Gulf sturgeon remained in the dam tailrace during the summer (Wooley and Crateau 1985; Odenkirk 1989).

Wooley and Crateau (1985) reported that of 99 Gulf sturgeon tagged below JWLD, Apalachicola River, 6 were incidentally captured by shrimp trawlers during the fall season in Apalachicola Bay and the adjacent Gulf of Mexico. Bradshaw (personal communication) notes three Gulf sturgeon he collected and tagged in 1985 from the East Pearl River at river km 32.2 (river mi 20) that were incidentally caught by shrimpers in Mississippi Sound in the fall of that year. One Gulf sturgeon, a 53.0 cm (20.9 in) FL individual, was caught near the west tip of Cat Island, a distance of 64.6 km (40 mi) from the release point on the river.

Subadult and adult Gulf sturgeon in the Suwannee and Apalachicola Rivers generally began downstream migration in late September and October. Wooley and Crateau (1985) found that the Gulf sturgeon at the JWLD began their downstream migration in late fall when the temperature dropped to 23°C (73.4°F). Most return to the estuary or the Gulf of Mexico by mid-November to early December. In the Suwannee River, young Gulf sturgeon from about 0.3 to 2.5 kg (0.7 to 5.5 lb) remained at the river mouth during the winter and spring and were the only Gulf sturgeon captured during December, January and early February over a three year period from late 1987 to 1991 (Clugston et al. 1995). Based on mark-recapture data, these young fish did not appear to venture far into the Gulf of Mexico. Tagging (J. Clugston, unpublished data) and other life history studies (Huff 1975) found small Gulf sturgeon at river distributaries indicating that they were spawned in the Suwannee River.

Radio telemetry studies on the Choctawhatchee River conducted by NCSU in the summer of 1994, found that 25 tagged Gulf sturgeon did not distribute themselves uniformly throughout the river and did not occupy the deepest or coolest water available (Potak et al. 1995). Most fish were concentrated in relatively shallow straight stretches of the river. Of the 25 fish, 23 remained within two primary summer holding areas in the middle to lower river. They were found outside the main channel, where water velocities were less than the maximum available. Most of the fish were in water depths of 1.5 to 3.0 m (4.9 to 9.9 ft) and substrates were silt or clay.

Tagging and radio telemetry studies conducted by the LDWF during 1993 and 1994 showed subadult and adult Gulf sturgeon frequented or moved between specific areas from May through September. The most southern site is known as the Drum Hole on the west Middle Pearl River to the upper and lower Fridays Ditch on the west Middle Pearl River. Telemetry data showed movement of fish between Fridays Ditch to the West Pearl River at Powerline and Yellow Lake. Movement was also observed from Gulf sturgeon tagged from the Boque Chitto River below the sill at the canal and Lake Pontchartrain at Bayou Lacombe (H. Rogillio, personal communication).

Three sonic-tagged Gulf sturgeon were tracked into saline water and monitored in Apalachicola Bay for one to four hours in late October 1987. In November 1989, a Gulf sturgeon was monitored in Apalachicola Bay for 72 hours and tracked for 30.0 km (18.6 mi) (FWS 1988, 1989). Four Gulf sturgeon were similarly tracked in late October 1991 outside the Suwannee River and remained for about a week in water depths of 3.0 m (9.8 ft) and 5.0 km (3.1 mi) offshore in an area of mud bottom (Carr, unpublished data).

Gulf sturgeon tagging studies in the Apalachicola and Suwannee rivers demonstrate the high probability of recapture in the same river in which the fish were tagged. Between 1986 to 1992, approximately 3,750 Gulf sturgeon were tagged in the Suwannee River, and of nearly 700 recaptures, all but two were recovered in the Suwannee River. Those two recaptures occurred in the Apalachicola River and offshore near Tarpon Springs, Florida. From 1981 to 1993, a total of 350 Gulf sturgeon were tagged in the Apalachicola River. Of those, 160 were recaptured in the Apalachicola River, while six individuals were recaptured in the East Pass of the Suwannee River (S. Carr, unpublished data) and one was recaptured in the Ochlockonee River (F. Parauka, personal communication). Of those six individuals recaptured in the Suwannee River, three were recaptured the following year in the East Pass. Radio-tracking further suggests that individuals return to the same area of the river inhabited the previous summer (Foster 1993; Carr, unpublished data; FWS/Panama City, unpublished data).

Small Gulf sturgeon were noted to move southward along the western Florida coast to Florida Bay during the winters of 1957, 1959, and 1962 (D. Robins in personal communication to Wooley and Crateau 1985). Several sturgeon, estimated at 60 cm (23.6 in) FL, were also collected in fish traps in Government Cut, Miami, Florida during the winters of 1957, 1959, and 1962 (D. Robins, personal communication). Vladykov examined one of the specimens internally and determined it to be *A. o. desotoi*. These occurrences may have been in response to unusually low winter temperatures.

Stocks

Stabile et al. (unpublished manuscript) used RFLP analysis of mitochondrial DNA (mtDNA) of Gulf sturgeon collected from six geographically disjunct drainages along the Gulf of Mexico. The river systems included the Suwannee, Apalachicola, Ochlockonee, Blackwater, and Choctawhatchee rivers in Florida and the Pearl River in Louisiana/Mississippi. Their preliminary data analysis indicates that there are significant differences among Gulf sturgeon stocks. They found the most notable difference existed between the Choctawhatchee River samples and samples from other Gulf of Mexico rivers. In addition, the results indicated a break between the Apalachicola/Suwannee river populations and populations to the west of the Apalachicola River. Further, their data suggest that Gulf sturgeon display region-specific affinities and may exhibit river-specific fidelity.

Stabile et al. (unpublished manuscript) also indicated population-level polymorphisms using direct sequence analysis in sturgeon from the Gulf coast rivers. They found that Gulf sturgeon analyzed from the Pearl River exhibited haplotypes that were different from all other Gulf coast samples. Polymorphisms at other sites indicated possibly useful markers for discriminating sturgeon from the Choctawhatchee and Yellow rivers. No significant differences of mtDNA haplotypes were found among Gulf sturgeon from the eastern Gulf coast. However, these results are considered tentative because of the small sample size.

Food Habits

In the Suwannee River, stomachs of Gulf sturgeon 38 to 188 cm (15.0 to 74.0 in) FL caught in commercial gill nets 10.0 m (32.8 ft), 24.5 cm (9.4 in) stretch fished in the lower river in East Pass contained digested aquatic plant material interspersed with crab hard parts (probably blue crab, *Callinectes sapidus*). The relative abundance of crab parts was greater in stomachs of migrants entering the river in spring and usually absent from those exiting in fall (Huff 1975). Gammaridean amphipods were primarily found in smaller schooled Gulf sturgeon <82.0 cm (32.3 in) caught with trammel nets in shallow water 1.0 to 2.0 m (3.3 to 6.6 ft) in depth over a sand bank at the river's mouth (Alligator Pass). These prey species are associated with sandy substrates. Other food items included isopods (*Cyathura burbanki*), midge larvae, mud shrimp (Callianassidae), one eel (*Moringua* sp.), and unidentifiable animal or vegetable matter. Huff concluded that these small Gulf sturgeon occupied a different habitat than larger Gulf sturgeon harvested in the gill net fishery.

Mason and Clugston (1993) studied the food habits of Gulf sturgeon on the Suwannee River from 1988 to 1990. In the spring, immigrating subadult and adult Gulf sturgeon collected from the river mouth contained gammarid, haustoriid, and other amphipods, polychaete and oligochaete annelids, lancelets, and brachiopods. However, once in fresh water, these Gulf sturgeon did not eat as evidenced by the presence of only a greenish-tinged mucus in their guts during June through October. Stephen Carr (unpublished data) found in the Suwannee River that immigrating, sexually mature Gulf sturgeon were mainly empty of food; however, of food items present, brachiopods and mud shrimp dominated. By contrast, a 13.6 kg (30.0 lb) Gulf sturgeon was captured by bait trawlers on Red Bank Reef three miles from the mouth of the Suwannee River in spring 1986. Its stomach contained six species of lugworm, two species of clam, five species of crustacea, an echinoderm (sand dollar), an unidentifiable marine worm and two dozen lancelets (S. Carr, unpublished data). Mason and Clugston (1993) found that small Gulf sturgeon (0.5 to 4.0 kg) (1.1 to 8.8 lb) collected at the river mouth during the winter and early spring contained amphipod and isopod crustaceans, oligochaetes, polychaetes, and chironomid and ceratopogonid larvae. Although the guts of these young Gulf sturgeon contained small amounts of food as they migrated upstream to about river km 55 (river mi 34), they too contained only a detrital mass and were essentially empty in the freshwater reaches during the summer and fall. It remains unclear why most subadult and adult Gulf sturgeon feed for three to four months in a marine environment and enter fresh water where they do not feed for the following eight or nine months.

Growth

Huff (1975) used cross sections of pectoral fin rays to estimate the age of 631 Gulf sturgeon collected from the Suwannee River. Because back calculation using fin ray sections was not possible, mean fork lengths for fish ages 1 through 17 were calculated (Figure 3). Mean fork length at age 1 was approximately 35.0 cm (13.8 in) and increased to approximately 145.0 cm (57.1 in) at age 17.

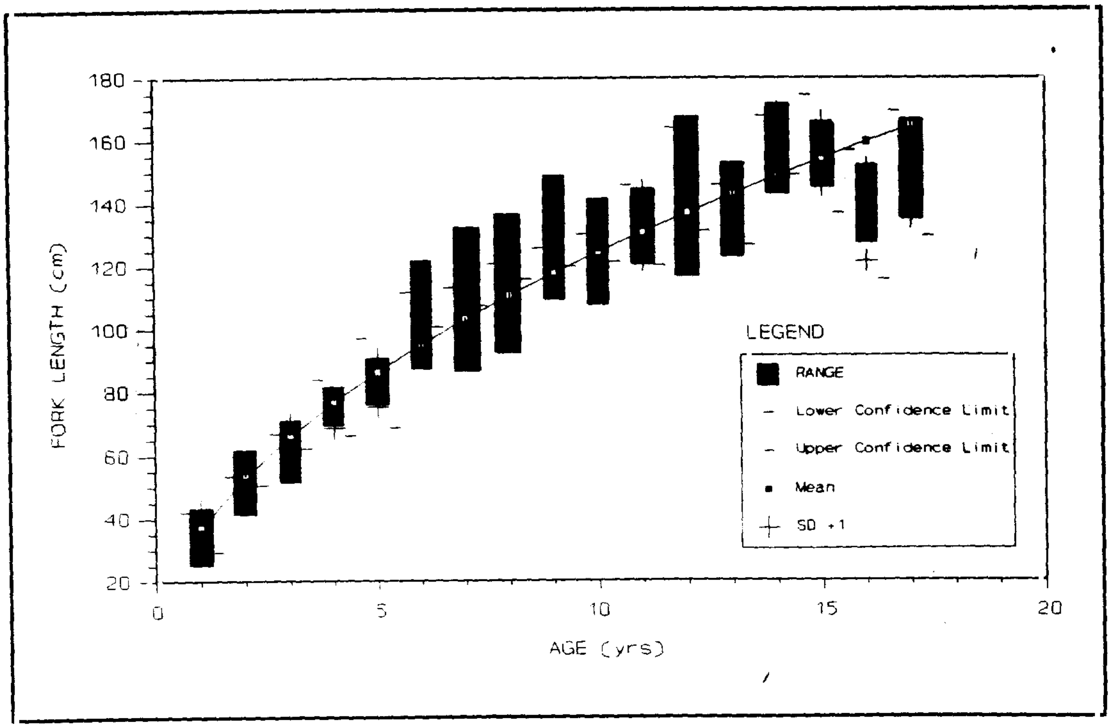


Figure 3: Length-range diagram and regression line,
Gulf sturgeon age groups 1 to 17, from 1972 to 1973 (Huff 1975)

Cross sections of pectoral fin rays were also used to estimate the age of 76 Gulf sturgeon collected from the Apalachicola River, Florida from 1982 to 1990 (Jenkins, unpublished manuscript). Fish ranged from 2 to 28 years old with lengths and weights ranging from 47.0 to 227.0 cm (18.5 to 89.4 in) and 0.2 to 90.7 kg (0.4 to 200.0 lb). Fin rays from four fish exhibited possible spawning belts. Average growth was 24.0 cm (9.4 in) per year for fish two to five years old, and 8.0 cm (3.1 in) per year to the age of eight. Fish marked and later recaptured exhibited similar large growth variations which may be the result of sexual dimorphism. The time of annulus formation was in the late summer and fall, which is a period of weight loss according to mark-recapture studies.

Carr (1983) found that on the average, marked Gulf sturgeon from the Suwannee River gained 30% of body weight in one year. He also noted that little or no growth was seen when recapture occurred during the same season and a little weight was lost by some. Wooley and Crateau (1985) noted that Gulf sturgeon 80.0 to 114.0 cm (31.5 to 44.9 in) FL tagged in early summer in the Apalachicola River below JWLD and subsequently recaptured in the same area in July and September exhibited weight losses of 4% to 15% or 0.5 to 2.3 kg (1.1 to 5.1 lb). Gulf sturgeon from 75.5 to 101.0 cm (29.7 to 39.8 in) FL tagged in September and recaptured the following year between May and September, after spending the winter period feeding in Apalachicola Bay and/or the Gulf of Mexico, showed weight gains of 35% to 137% or 4.3 to 10.2 kg (9.5 to 22.5 lb). These growth rates are considered normal for young Gulf sturgeon.

The recapture of 229 marked fish provided an opportunity to calculate seasonal growth rates of Gulf sturgeon in the Suwannee River (Clugston et al. 1995). It appears that Gulf sturgeon gain weight only during the winter and spring while in marine or estuarine waters and lose weight during the eight to nine month period while in fresh water. In general, Gulf sturgeon weighing between 7.0 kg (15.4 lb) and 27.0 kg (59.5 lb) grew about 11.0 cm (4.3 in) and gained 2.0 to 3.0 kg (4.4 to 6.6 lb) per year. In nearly all cases, however, fish that were marked and recaptured during the same summer lost weight. Those recaptures that spanned the three or four months that most fish were in the Gulf of Mexico increased in weight. Likewise, the young fish collected at the mouth of the river during the winter and spring and recaptured during the same period increased in weight. Lengths and weights were monitored for two Gulf sturgeon hatched and reared for 17 months under laboratory conditions (Mason et al., 1992). In the first year these fish grew to 71.9 cm (28.3 in) and 63.4 cm (25.0 in) in total length and to weights of 1.9 kg (4.2 lb) and 1.4 kg (3.1 lb). After 17 months they grew to 84.6 cm (33.3 in) and 78.7 cm (31.0 in) and to 3.1 kg (6.7 lb) and 2.7 kg (6.0 lb). These two fish received special treatment, and their growth in the laboratory may not represent growth of wild fish. Nevertheless, the data represent the first measured growth of young Gulf sturgeon and provide insight into the species' growth potential.

Reproduction

Timing, location and habitat requirements for Gulf sturgeon spawning are not well documented. Most subadult and adult Gulf sturgeon ascend coastal rivers from the Gulf of Mexico from mid-February through April when some adults are sexually mature and in ripe condition. Studies conducted on the Apalachicola River resulted in the only known collection of wild Gulf sturgeon larvae. Two larvae were collected at river km 168 (river mi 104.2); one on May 11, 1977 (Wooley et al., 1982) and one on May 1, 1987 (Foster et al., 1988). At the time of the 1977 collection, the surface water temperature was 23.9°C (75.0°F), water depth 4.2 m (13.78 ft), flow 365.0 m^3/s (12,888.0 ft^3/s), and velocity of .67 m/s (2.2 ft/s). During the 1987 collection the surface water temperature was 21.6°C (70.9°F), water depth 4.2 m (13.8 ft), flow 437.0 m^3/s (15430.0 ft^3/s), velocity not measured. The larva collected in 1977 was estimated to be 1 to 2 days old while the other larva was estimated to be a few hours old. A third larva was collected on April 3, 1987 at river km 18.7 (river mi 11.6) at a water temperature of 16.1°C (61.0°F), water depth 7.9 m (25.9 ft), flow not measured, and velocity .96 m/s (3.2 ft/s). The larva was estimated to be about 1 to 1.5 days old (FWS 1988).

Huff (1975) spent considerable time using anchored plankton nets to collect Gulf sturgeon eggs and larvae in the Suwannee River but was unsuccessful. However, two Gulf sturgeon eggs were collected in the river on April 22, 1993 (Marchant and Shutters, unpublished manuscript). The eggs were collected in water depths of 5.5 m and 7.3 m (18.0 ft and 24.0 ft) and water temperature 18.3°C (65.0°F) at river km 215 (river mi 134.2), just downstream of the confluence of the Alapaha River. Additional eggs were collected during late March and April 1994 at river km 201 to 221 (river mi 124.9 to 137.3) when water temperatures ranged from 18.8°C to 20.1°C (65.8°F to 68.2°F)(Smith and Clugston, unpublished manuscript). From 1988 through 1992, Gulf sturgeon investigations were conducted throughout the Suwannee River using plankton nets, small-mesh trap nets, trawls and gill nets, and electrofishing equipment. The smallest Gulf sturgeon collected was a 30.6 cm (12.0 in) specimen weighing 85.0 g (0.2 lb) at river km 215.0 (river mi 133.6) on December 3, 1991 (Clugston et al. 1995).

Stephen Carr and F. Tatman (unpublished data) found that 15 ultrasonic-tagged gravid females were associated with springs between river kms 32.0 and 145.0 (river mi 19.9 and 90.1) in the Suwannee River. The bottom habitats surrounding the springs consist mainly of rock. Their consistent association with these springs has led to Carr's speculation that spawning occurs in these areas.

Remnant reproductive populations may still occur in many small and large rivers draining into the Gulf where Gulf sturgeon have historically ranged. Infrequent anecdotal reports and incidental captures of small Gulf sturgeon indicate that reproduction is occurring in tributary rivers. Small Gulf sturgeon are closely associated with the river basin where they were spawned (river-specific affinity). This has been demonstrated in the Suwannee River and Apalachicola River/Bay distributaries, by the occurrence of similar size Gulf sturgeon in similar depths, and on similar substrate. Any analogous occurrence of small Gulf sturgeon suggests that a reproducing population remains nearby.

Spawning Age

Huff (1975) found that sexually mature females ranged in age from 8 to 17 years and sexually mature males from 7 to 21 years in the Suwannee River. The youngest ripe female specimen and the oldest immature female were age 12. The youngest ripe male specimen was 9 years old and the oldest immature male was age 10. Jenkins (unpublished manuscript) estimated a ripe male captured from the Suwannee River in 1990 to be six to seven years old.

Fecundity

Chapman et al. (1993) reported that three mature Gulf sturgeon had 458,080, 274,680, and 475,000 eggs and were estimated to have an average fecundity of 20,652 eggs/kg (9,366 eggs/lb). Smith et al. (1980) estimated that Atlantic sturgeon weighing 50.0 and 100.0 kg (110.2 and 220.5 lb) would yield over 400,000 and 1,000,000 eggs, respectively.

Gulf sturgeon eggs are demersal and adhesive (Vladykov 1963; Huff 1975; Parauka et al., 1991; Chapman et al., 1993). The eggs are globular and vary in color from gray to brown to black. Smith et al. (1980) reported that Atlantic sturgeon eggs ranged in size from 2.5 to 3.0 mm (0.10 to 0.12 in) in diameter. Parauka et al., (1991) found that eggs from Gulf sturgeon averaged 2.10 and 2.20 mm (0.08 to 0.09 in) in diameter.

Reproduction in Hatcheries

Hormone-induced ovulation and spawning of Gulf sturgeon was accomplished in 1989 at a portable hatchery located on the Suwannee River and at the Welaka National Fish Hatchery in

Florida (Parauka et al., 1991). The project was a joint effort involving the FWS, CCC, and University of California, Davis. The initial spawning produced 5,000 fry for fishery research. In 1990, 1991, and 1992, the University of Florida, the FWS, and CCC again successfully induced spawning and produced about 60,000 fry for fish culture programs. Hatching time for the artificially spawned Gulf sturgeon eggs ranged from 85.5 hr at 18.4°C (65.1°F) to 54.4 hr at about 23.0°C (73.4°F) (Figure 4) (Parauka et al., 1991). Also, at temperatures ranging from 15.6 to 17.2°C (60.1 to 63.0°F) and 19.5 to 21.0°C (67.1 to 69.8°F), eggs hatched in 95 and 65 to 70 hr, respectively (FWS 1991b). Chapman et al. (1993) reported that artificially spawned Gulf sturgeon eggs incubated at 20°C (68°F) hatched in 3.5 days. Hatching time for Atlantic sturgeon eggs has been reported to be 94 hr at 20.0°C (68.0°F) (Dean 1893), 121 to 140 hr at 16.0 to 19.0°C (60.8 to 66.2°F) (Smith et al., 1980) and 168 hr at 17.8°C (64.0°F) (Vladykov and Greeley 1963). One-hour-old Gulf sturgeon larvae, hatched under artificial conditions on the Suwannee River in 1989, ranged in length from 0.66 to 0.71 cm (0.26 to 0.28 in) with a mean length of 0.69 cm (0.27 in) (Parauka et al., 1991). Hatching success ranged from 5 to 10%.

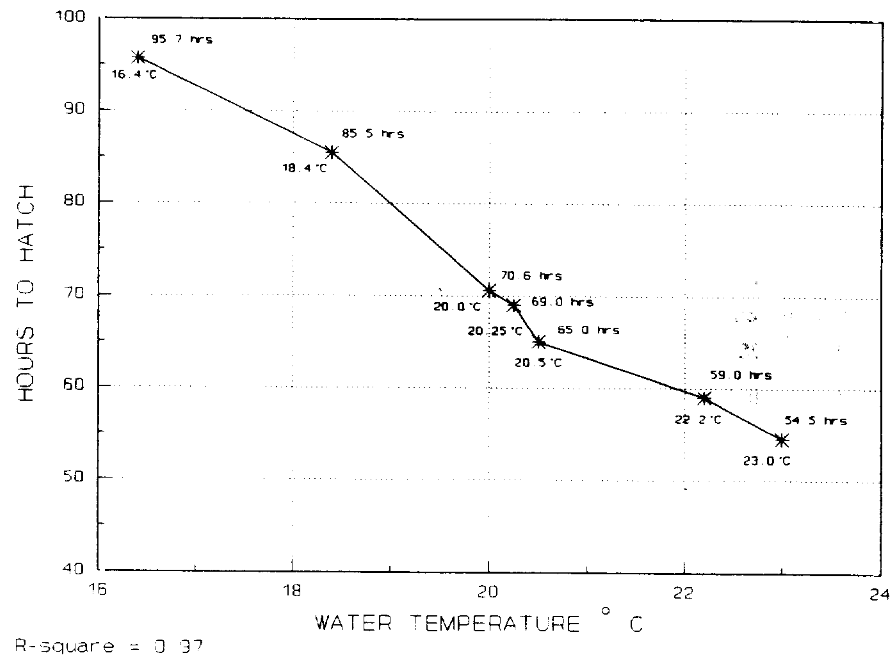


Figure 4: Gulf sturgeon egg incubation periods
at different mean water temperature (F. Parauka et al., 1991; FWS 1991b).

Predator/Prey Relationships

Van Den Avyle (1984) noted there was little written regarding competitors and predators of sturgeon. He pointed out that many fish species live in the same waters as sturgeon and that there is the possibility for competition with other bottom dwelling species. In fresh water, benthic feeders could compete with young sturgeon or feed directly on eggs and larvae. Competition with Gulf sturgeon for food or space in the marine environment is unknown. Scott and Crossman (1973) speculated that the sturgeon's "size and protective plates protect it from most predaceous fishes and its habitat and secretiveness from other predators."

Parasites and Disease

Fish lice *Argulus stizostethi*, an ectoparasitic copepod, have occasionally been observed on the opercula and gill filaments and in the gut of Gulf sturgeon collected in fresh and estuarine water. The numbers noted were not significant (Mason and Clugston 1993; F. Parauka, personal communication). Endoparasites, such as nematodes, trematodes, and leeches were noted in the guts of Gulf sturgeon (Mason and Clugston 1993). Five species of helminth parasites and one parasitic arthropod have been identified in Atlantic sturgeon from the St. Johns River, New Brunswick (Appey and Dadswell 1978). No detrimental effects from these parasites were noted in these studies.

The shovelnose sturgeon serves as host for glochidia of three mussel species. Rates of glochidial infestation on fish gills are typically low, but thought not to be detrimental to the host (R.S. Butler, personal communication). Huff (1975) reported tumor-like growths on several Gulf sturgeon ovaries from the Suwannee River. Macroscopic tumors were found from 7.5% of gill-netted females in Fall 1972, 3.5% of females in Spring 1973, and 4.6% of females in Fall 1973. Examination of this material revealed two types of growth (Harshbarger 1975). One was a perifollicular pseudocyst (surrounding follicles) filled with proteinaceous fluid often containing viable oocytes. The other type was a parafollicular serous cyst (a true separate fluid-filled cyst) containing denser proteinaceous fluid. Both types are considered subclinical, having little or no effect on adjacent organs, general ovarian development, fecundity, or spawning behavior. Microscopic slides (RTLA nos. 979 and 980) containing this material were accessioned by the Registry of Tumors in Lower Animals, Smithsonian Institution (Huff 1975). Moser and Ross (1993) reported the capture of six Atlantic sturgeon from the Brunswick River, North Carolina from June to September 1991 and in April 1992. Three of the specimen were in poor condition with abnormalities characterized by deformed mouths, lesions of the ventral buccal region and/or lesions around the eye. Oral, buccal, and ventral lesions or ulterations are common signs of poor water quality. Veterinarians examined another sturgeon from the Brunswick River that died without external evidence of disease and found the liver and heart tissues to be in poor condition.

FACTORS CONTRIBUTING TO THE DECLINE AND IMPEDIMENTS TO RECOVERY

Many members of the family Acipenseridae, including Gulf sturgeon, virtually disappeared throughout their ranges at the turn of the 20th century. Their decline was likely caused by over-exploitation and exacerbated by damming of rivers and other forms of habitat destruction and water quality deterioration, among other factors (Birstein 1993; Huff 1975; Barkuloo 1988; McDowall 1988; Smith and Clugston, unpublished manuscript).

Exploitation

The Gulf sturgeon was heavily fished because of the high value of its eggs used to produce caviar and its flesh for smoking (Carr 1983; J. Barkuloo, personal communication). Sturgeon also provided isinglass, a semi-transparent gelatin prepared from the swim bladder and used in jellies, wine and beer clarification, special cements, and glues. Directed commercial fishing contributed to the depletion of sturgeon populations. Aperiodic commercial landing statistics are available from 1887 to 1985 for Gulf sturgeon (Huff 1975; Futch 1984; Barkuloo 1988). Commercial landings data for the Suwannee River are available for 1981 to 1984 (Tatman, unpublished data). These records show that the only consistent fisheries for Gulf sturgeon occurred in west Florida. There was a directed fishery in Alabama, while there is no record of a directed commercial fishery in Mississippi, only incidental catches. Davis et al., (1970) notes a minor commercial fishery for Gulf sturgeon in the Lake Pontchartrain and its tributaries during the late 1960's.

Recreational and subsistence fishing may have contributed to population declines. A "snatch-hook" recreational fishery was popular on the Apalachicola River, Florida, during the late 1950's to 1960's (Burgess 1963; Swift et al., 1977) and continued until 1984 when the State of Florida enacted protective measures.

Incidental Catch

Incidental catch of Gulf sturgeon in other fisheries has been documented (Wooley and Crateau 1985; D. Mowbray, personal communication; H. Rogillio, personal communication). Incidental captures by commercial shrimpers and gill net fishermen in Apalachicola Bay were noted by Wooley and Crateau (1985) and reported by Swift et al. (1977). Such catches have also occurred in Mobile Bay, Tampa Bay, and Charlotte Harbor (J. Roussos, personal communication; FDEP, unpublished data). The FWS caught a small Gulf sturgeon in St. Andrew Bay while gill-net collecting for seatrout for contaminant analysis in 1986 (M. Brim, personal communication). Gulf sturgeon are occasionally caught in Gulf coast rivers on set-hooks targeting catfish (J. Duffy, personal communication). Captures of young Gulf sturgeon have been reported in blue crab traps in the Suwannee River estuary (F. Tatman, personal communication). The incidental catch of Gulf sturgeon in the industrial bottomfish (petfood) fishery in the north-central Gulf of Mexico from 1959 to 1963 was reported by Roithmayr (1965). The bottomfish fishery worked an area between Point au Fer, Louisiana and Perdido Bay, Florida from shore to water depths of about 55 m (180 ft). Hastings (1983) and Moser and Ross (1993) report capture and disruption of spawning migrations of shortnose and Atlantic sturgeon in commercial gill nets targeted for shad in the Cape Fear River, North Carolina.

The LDWF records indicate 177 Gulf sturgeon were incidentally captured and reported by commercial fishermen in southeastern Louisiana during 1992 (H. Rogillio, personal communication). Forty-four of these Gulf sturgeon were delivered to the LDWF field office or held until LDWF employees could secure them. Specimens were generally held in captivity for 1 to 7 days by the fishermen. These sturgeon were then measured, weighed, tagged and released by departmental personnel. Seventy-six Gulf sturgeon were captured in trawls, 10 in wing nets, and 91 in gill nets. A mortality of less than 1% was noted. This percentage is based on 177 Gulf sturgeon incidentally captured by commercial fishermen and 51 Gulf sturgeon captured by LDWF personnel during a Gulf sturgeon status survey.

Bradshaw (personal communication) reported three tag returns from Gulf sturgeon he collected in early 1985 which were incidentally caught by shrimpers in Mississippi Sound during the fall of that year. He also noted finding three dead Gulf sturgeon incidentally caught by gillnetters in the western part of the Sound and revived another Gulf sturgeon a gillnetter had caught "on" Horn Island in 1989.

Entrainment of *Acipenser guldenstadti* and *A. stellatus* larvae during dredging operations has been assessed by Veshchev (1982) in the lower Volga River, Russia. He concluded that hydraulic dredging operations caused significant mortality of sturgeon larvae in the Caspian basin.

Hastings (1983) reported anecdotal accounts of adult sturgeon being expelled from dredge spoil pipes while conducting a study on shortnose sturgeon on the Atlantic coast. Whether the "adult sturgeon" was an Atlantic or shortnose sturgeon was not indicated in the report.

Habitat Reduction and Degradation

Gulf sturgeon have evolved within Gulf coast drainages that exhibit seasonal patterns of high and low flows, temperature regimes, sedimentation, and other physical factors. Provision of these essential life requirements are part of and dependent on a fully functioning ecosystem.

Dams have limited sturgeon access to migration routes and historic spawning areas (Boschung 1976; Murawski and Pacheco 1977; Wooley and Crateau 1985; McDowall 1988) (Table 1). While sturgeon are able to pass some water control structures, low-head dams, or sills during high water, these structures can create barriers that preclude normal migration. An example of complete migration restriction occurred in the St. Andrew Bay system, Bay County, Florida. A newspaper account from 1895 reports sturgeon were caught at the head of North Bay in upper St. Andrew Bay (Womack 1991). The account notes that an average of three sturgeon a day were caught and 90.7 kg (200 lb) of fish had been smoked and on sale for $0.10 per lb. The FGFC collected four Gulf sturgeon 173.0 to 201.5 cm (68.1 to 79.3 in) in length from Bear Creek, a tributary to Econfina Creek which drains into North Bay, in May of 1961. A dam was placed across North Bay in 1962 preventing anadromous fish migration, and no reports of Gulf sturgeon from above the dam have been reported since that time. Not only was migration to the creeks cutoff, but approximately 2024 hectares (5,000 acres) of estuarine habitat was converted into a fresh water lake.

Another example of complete restriction to Gulf sturgeon migration is the JWLD on the Apalachicola River. Swift et al. (1977) noted a report of a Gulf sturgeon from the Flint River near Albany, Georgia prior to 1950. Huff (1975) noted Gulf sturgeon migrated 322 km

Table 1: Examples of reduction in available river habitat due to dam, water control structure, or sill construction.

River/Watershed	Total River Length	Location of Impediment	Percent Habitat Remaining
St. Andrew Bay Drainage Bear Creek, Lower Econfina Creek, upper North Bay (now known as Deer Point Lake)	11 km (6.8 mi)	Deer Point Dam County Rd 2321	0%
Apalachicola, Chattahoochee, Flint River Basin (to the fall line)	790 km (491 mi)	JWLD river km 172 (river mi 107)	22%
Mobile Bay Drainage Basin Alabama River	1691 km (1051 mi)	Claiborne Dam river km 130 (river mi 81)	8%
Tombigbee River	988 km (614 mi)	Coffeeville Dam river km 121 (river mi 75)	12%
Pearl River	772 km (480 mi)	Ross Barnett Dam (RBD) river km 486 (river mi 302)	63%
During low water conditions		Pools Bluff Sill river km 78.3 (river mi 48.7)	10%
Bogue Chitto River (during low water conditions)	217 km (135 mi)	Boque Chitto Sill river km 6.4 (river mi 4)	3%
Amite River	274 km (170 mi)	control weir river km 40.7 (river mi 25.3)	15%

(200 mi) upstream in the Apalachicola-Chattahoochee-Flint river system before the dam construction in 1957. There are numerous anecdotal reports of Gulf sturgeon in the Flint and Chattahoochee rivers prior to construction of JWLD (Swift et al. 1977). In spite of many tagging studies conducted on the Apalachicola River, no tags have been returned as a result of Gulf sturgeon moving upstream of JWLD, nor does evidence exist that the Gulf sturgeon passes though the lock system (A. Carr, personal communication; U.S. Fish and Wildlife Service, personal communication). The COE (1978) acknowledged that the dam on the Apalachicola River adversely affect Gulf sturgeon by impeding upstream migration.

An example of barriers that limit movement is found in the Pearl River basin above the Pools Bluff and Bogue Chitto Sills. Gulf sturgeon have been reported to be incidentally collected above the Pools Bluff Sill as far north as the Ross Barnett Reservoir spillway as late as 1984 (J. Stewart, personal communication; R. Jones, personal communication; W. McDearman, personal communication; R. Bowker, personal communication). Based on gauge data (COE, personal communication), the duration of water depths allowing passage of Gulf sturgeon over the sills is limited at the Bogue Chitto Sill and less restrictive at the Pools Bluff Sill (Table 2). It appears Gulf sturgeon movement above the sills is also possible through cutoffs that have developed since the construction of the Pearl River navigation canal (H. Poitevint, personal communication). However, Gulf sturgeon migration is entirely prevented above Jackson, Mississippi by the Ross Barnett Dam at river km 515 (river mi 320). Jones (personal communication) reports that Gulf sturgeon were historically found above this area. He notes the capture of a 154.2 kg (340 lb) female Gulf sturgeon 2.3 m (7.5 ft) from the river 32 km (20 mi) north of Jackson in 1942.

Navigation activities including dam construction, dredging, dredged material, and other maintenance actions could adversely affect Gulf sturgeon habitats depending on the location and timing of the activity. Elimination of deep holes and alterations of rock substrates result in loss of habitat for the Gulf sturgeon in the Apalachicola River (Carr 1983; Wooley and Crateau 1985). At Rock Bluff, river km 148.8 (river mi 92.5), this deep, rocky area frequently used by Gulf sturgeon was filled with dredged spoil material drifting downstream from a within bank disposal site at river km 150 (river mi 93) during routine maintenance dredging. This caused Gulf sturgeon to cease use of this area as a regular habitat (Carr 1983, J. Barkuloo, personal communication). The within bank disposal site is no longer used. Essential habitats of young-of-the-year Gulf sturgeon are unknown, so the impacts of dredging on early life stage habitats of Gulf sturgeon are difficult to assess.

Table 2: Duration Data on Lower Pearl River Sills (COE, personal communication).

Depth Over Sill (m)	Percent Equaled or Exceeded	
	Pools Bluff Sill[1]	Bogue Chitto Sill[2]
.3 m (1.0 ft)	100	90
.61 m (2.0 ft)	70	25
.9 m (3.0 ft)	48	10
1.2 m (4.0 ft)	35	-
1.5 m (5.0 ft)	28	-
1.8 m (6.0 ft)	24	-
2.1 m (7.0 ft)	18	-

[1]Duration based on gauge data for Pearl River at Bogulusa, Louisiana
[2]Duration based on gauge data for Bogue Chitto River at Sun, Louisiana

The entrenchment of the Apalachicola River's streambed due to the trapping of sediments in Lake Seminole, has been attributed to the construction of JWLD (COE 1986). The effects entrenchment occurred in the upper third of the river from the base of the dam to the vicinity of Blountstown, Florida. The streambed elevation lowering was also exacerbated by deepening rock sills, cutting out river bends, and repeated dredging to maintain the channel. This has resulted in elimination of some habitats that had been available to Gulf sturgeon during the summer months prior to the construction of JWLD and navigation channels. For example, as a result of streambed degradation, access to spring-fed tributary creeks has been reduced during low water periods. A cooperative effort by the COE and FGFC removed sedimentation and debris from a midstream spring below the JWLD, navigation km 170.6 (navigation mi 106.0) in January 1994. In addition, the COE obtained environmental clearances and unertook habitat restoration action by the removal of sediments at the mouth of Blue Spring Run, navigation 157.7 (river mi 98.0) in May, 1994.

Cool water habitats are thought to be important to Gulf sturgeon during the summer. Cool-water habitats in streams can be significantly reduced or even eliminated by decreased groundwater levels (Lynn Torak, personal communication). Springs emanating from the streambed originate in the groundwater-flow system and are regulated by relative differences in stream stage, spring-discharge elevation, and groundwater level. Decreased groundwater levels in the vicinity of streams, caused by pumping or climatic variation, can reduce springflow that provides cool-water habitats for the Gulf sturgeon during summer months. Pumping or climate-induced groundwater-level declines can reduce the groundwater component of streamflow (baseflow) in addition to and in the absence of springs. For example, a study in the Albany, Georgia area by Torak et al. (1993) indicates that about 74% of water pumped from the Upper Floridan aquifer in November 1985, approximately 79 million gallons a day, would have discharged to the Flint River under predevelopment conditions. The Flint River is generally unregulated and has a major spring-fed flow component that, in comparison with the Chattahoochee River, contributes the larger share of flow to the Apalachicola River during low-flow periods. The Chattahoochee River is a regulated stream that derives its flow predominantly from surface runoff. Consequently, the Chattahoochee River contributes the major portion of flow to the Apalachicola River during mean- to high-water events. Base-flow of the Flint River has been reduced since the early 1970s, mainly from groundwater and surface water irrigation withdrawals (Leitman et al. 1993). The analysis by Leitman et al. (1993) indicates that the Flint River's percent contribution to the Apalachicola River decreases, instead of increasing as would be expected as the flow in the Apalachicola River decreases. Several springs and spring runs along the upper Apalachicola and Flint Rivers have already exhibited greatly reduced flow or have ceased flowing during periods of drought. If these cool water habitats are important and are reduced in size or eliminated at critical periods of summer, Gulf sturgeon could be subjected to increased environmental stress.

Contaminants may also contribute to population declines. Experiments have shown that DDT and its derivatives and toxaphene are toxic to fish in minute quantities (Johnson and Finley 1980; White et al. 1983). Twelve Gulf sturgeon were collected from the Apalachicola, Suwannee, Choctawhatchee rivers, Ochlockonee Bay and the Gulf of Mexico near Cape San Blas, Florida, at various times between 1985 to 1991. This specimens were analyzed for pesticides and heavy metals (Bateman and Brim 1994). The Gulf sturgeon ranged in size from 1.8 to 49.0 kg (4.0 to 108.0 lb). Concentrations of arsenic, mercury, DDT metabolites, toxaphene, polycyclic aromatic hydrocarbons, and aliphatic hydrocarbons high enough to warrant concern were detected in individual fish. Specific sources of contamination were not identified. Suwannee River Gulf sturgeon had higher concentrations of arsenic in liver samples than Apalachicola River fish. However, Apalachicola River Gulf sturgeon had higher liver mercury concentrations. Organochlorine pesticides were also highest in fish from the Apalachicola River.

Organochlorines enter the environment as pesticides or industrial waste products. Use of most of these compounds has been prohibited because of effects on nontarget species and suspected carcinogenicity in humans and wildlife. Effects include reproductive failure, reduced survival of young, or physiological alterations which can affect the ability of the fish to withstand stress (White et al. 1983). Levels of DDT and derivative compounds in the samples were found at low concentrations in all Gulf sturgeon tissues, however, DDD and/or DDE was detected in 84% of the samples (Bateman and Brim 1994). In addition, amounts detected in reproductive tissue, while relatively low (range non-detect to 4.02 ppm), could affect Gulf sturgeon reproduction because DDT compounds are known to be estrogenic (Fox 1992). Like DDT, toxaphene is persistent in the environment and biomagnifies through the food chain. Toxaphene was the most heavily used insecticide after prohibition of DDT in the 1970s. Toxaphene was detected in four fish, all from the Apalachicola River. The level of toxaphene in the roe of one specimen was 14.00 ppm wet weight and exceeded the Food and Drug Administration (FDA) action level of 5.00 ppm for fish for human consumption. The highest level in muscle tissue (0.48 ppm) fell below the FDA action level for human consumption (Bateman and Brim 1994). Toxaphene is more toxic to fishes than DDT compounds (Johnson and Finley 1980) and has been shown to impair reproduction, reduce growth in adults and juveniles, and alter collagen formation in fry, resulting in "broken back syndrome" (Mayer and Mehrle 1977).

Polycyclic aromatic hydrocarbons (PAH), primarily from petroleum products, are known to be carcinogenic, cocarcinogenic and tumorigenic. Concentrations found in the ovarian tissue sample (total PAH 410 ppb; Apalachicola River) and eggs (total PAH 409 and 815 ppb; Suwannee River) could adversely affect development and survival of some percentage of eggs, larval, and juvenile fish (Bateman and Brim 1994). Aliphatic hydrocarbons are components of oils, fuels, and other petroleum products. Two or more aliphatic compounds were detected in all tissue samples of the Gulf sturgeon. Hall and Coon (1988) stated that it is likely that any animal with demonstrated petroleum hydrocarbon residues in the tissues has suffered effects of the pollutant (Bateman and Brim 1994).

Arsenic is used in herbicides, insecticides, and fungicides and can be toxic to fish in certain metabolic forms. The metal was detected in 92% of the Gulf sturgeon samples, however the metabolic form was not identified. The arsenic concentrations detected in all of the muscle tissue samples were greater than the FDA action limit of 0.50 ppm for swine muscle tissue (Bateman and Brim 1994).

Mercury, predominantly found as methylmercury in fish fillets, is highly toxic and was detected in 87% of the Gulf sturgeon samples. The mercury concentrations in muscle tissue were well below the Florida limited consumption advisory (0.50 ppm) and the FDA consumptive use action level (1.00 ppm) but, almost all tissue samples exceeded the predator protection limit of 0.10 ppm recommended by Eisler (1987) for the protection of fish-eating birds. However, the mercury levels of the Gulf sturgeon in the study were well below those reported by Armstrong (1979) for other fish species, to cause either chronic inability to catch food, rolling from side to side or acute toxicity.

Cadmium, a known teratogen, carcinogen, and probable mutagen was detected in 42% of the Gulf sturgeon samples. The concentrations were in the low to normal range for muscle and liver tissue when compared to fish species in the Fisheries Resources Trace Elements Survey (FRTES) of the NMFS (Bateman and Brim 1994). Low levels of lead were detected in 8%.

<u>Culture and Accidental or Intentional Introductions</u>

Where viable wild populations exist or sturgeon possibly can be reintroduced, the potential harm from incidental or accidental introduction of non-endemic species is a threat to the genetic integrity and biodiversity of entire ecosystems. The likelihood of these introductions increases dramatically where imports and culture of exotic species is allowed or facilitated, and even where laws or regulations exist which prohibit release of non-endemic species. Accidental releases from culture facilities and intentional releases by aquarists tiring of their hobby is a frequent occurrence. Schwartz (1972, 1981) identifies bibliographic citations of hybrid combinations between species of sturgeons (Acipenseridae). Therefore, an introduction, for example, of white sturgeon from the Pacific coast into Gulf river systems could potentially do great harm to Gulf sturgeon stocks.

An introduction has already occurred in Alabama. A white sturgeon, 50.1 cm (1.6 ft) TL, was caught by a commercial fisherman on a trotline in Lake Weiss, about 2.4 km (1.5 mi) south of Cedar Bluff, Alabama in 1989 (M. Pierson, personal communication). Lake Weiss is part of the upper Coosa River system flowing through Georgia and Alabama. In 1992 a white sturgeon, 96.0 cm (3.15 ft) TL, was caught by a fisherman in the Coosa River east of Birmingham (Sun Herald 1992). This sturgeon was caught about 100 km (62.1 mi) downstream from the 1989 capture. The white sturgeon is thought to have been accidentally released from a private fish hatchery located adjacent to the Coosa River in Georgia. The State of Georgia confiscated the white sturgeon from the hatchery in 1990.

A controversial fishery management problem revolves around the issue of hatchery stocks' adversely affect wild stocks. Hatchery technology has been employed for salmon in the Pacific Northwest for well over thirty years, but salmon stocks in many river systems have recently experienced significant declines. Biologists and many opponents of the hatchery programs attribute these declines on loss of genetic diversity caused by hatchery programs. Proponents of hatcheries argue that the basis of the problem is failure to protect habitat, manage water resources, control harvest, and prevent environmental contamination, among other factors.

These problems and failures may continue to contribute to reductions in stocks of Gulf sturgeon. The problems are readily evident and appropriate actions should be taken to correct them before or in conjuction with introduction of hatchery stock.

Other

Finally, life history characteristics of Gulf sturgeon may complicate and protract recovery efforts. Gulf sturgeon cannot establish a breeding population rapidly because of the long period they require to achieve sexual maturity. Further, Gulf sturgeon appear to be river-specific spawners, although immature Gulf sturgeon occasionally exhibit plasticity in movement or occurrence among Gulf basin rivers. Therefore natural repopulation may be non-existent or very low by Gulf sturgeon migrating from other rivers.

Fishery Management Jurisdiction, Laws, and Policies

The take of Gulf sturgeon is prohibited in the state waters of Louisiana, Mississippi, Alabama, and Florida. Section 6(a) of the ESA provides for extended cooperation with states for the purpose of conserving threatened and endangered species. The Departments of the Interior and Commerce may enter into cooperative agreements with a state, provided the state has an established program for the conservation of a listed species. The agreements authorize the states to implement the authorities and actions of the ESA relative to listed species recovery. Specifically, the states are authorized (1) to conduct investigations to determine the status and requirements for survival of resident species of fish and wildlife (this may include candidate species for listing), and (2) to establish programs, including acquisition of land or aquatic habitat or interests for the conservation of fish and wildlife. Federal funding is also provided to states under the agreements to implement the approved programs. All four of the above mentioned states have entered into Section 6 agreements with the FWS. More detailed descriptions of pertinent agencies, laws, and regulations are provided in Appendix A.

CONSERVATION ACCOMPLISHMENTS

Caribbean Conservation Corporation/Phipps Florida Foundation

1. Initiated tagging of Gulf sturgeon in 1975, using monel tags, in the Apalachicola and Suwannee Rivers which resulted in evidence of home-river fidelity, yearly growth rates, in-river weight loss, and an estimate of population size.

2. Initiated telemetry studies of Gulf sturgeon in 1976, providing evidence of the importance of the Floridian Aquifer to Gulf sturgeon ecology and in-river site fixity.

3. Initiated consultations which resulted in prohibition of take of Gulf sturgeon in the State of Florida.

Gulf States Marine Fisheries Commission

1. Initiated a Gulf sturgeon interjurisdictional fishery management plan in 1990 which evolved into the Gulf Sturgeon Recovery Plan.

National Biological Service, Southeastern Biological Science Center, (BSC-G formerly U.S. Fish and Wildlife Service), Gainesville, Florida

1. Since 1987 conducted comprehensive population and life history studies of Gulf sturgeon in the middle and lower Suwannee River, Florida, in cooperation with the CCC.

2. Facilitated survival and abundance estimates for Gulf sturgeon in the Suwannee River by FWS Resource Analysis Branch using CCC long-term data.

4. Developing relational database on physical, chemical, and biological characteristics of the Suwannee River for use with geographic information system (GIS) software.

5. Evaluating habitat characteristics in areas Gulf sturgeon are known to occupy during the summer months.

6. Conducted studies on movement of hatchery reared Gulf sturgeon released into the Suwannee River.

7. Conducted feasibility study for offshore sonic tracking of Gulf sturgeon.

8. Initiated field sampling in Tampa Bay and the Waccasassa, Steinhatchee, and Ochlockonee rivers to determine presence of Gulf sturgeon and evaluate existing habitat.

9. Provided an analysis of food habits of subadult and adult Gulf sturgeon in the Suwannee River.

10. Provided an assessment of the water quality of the Suwannee River and impacts of natural and human-induced disturbances on the food resources of the Gulf sturgeon.

11. Instituted and maintained a voucher specimen reference collection of Gulf sturgeon foods and provided expert assistance in identification of food organisms.

12. Devised and tested methods for culture of key foods used to rear Gulf sturgeon; amphipod crustaceans, brandling worm, West-African nightcrawler, blackworm, and tubificid oligochaetes.

13. Participated in first artificial spawning of the Gulf sturgeon at a temporary streamside facility in 1989-1991 and in 1992-1993 at the NBS\BSC.

14. Provided the first documented growth of Gulf sturgeon fed natural foods in a laboratory from fry stage to 17 months.

15. Conducted food preference study on cultured juvenile Gulf sturgeon comparing survivorship and growth between live and commercially prepared foods.

16. Identified critical thermal maximum and preferred temperature for cultured juvenile Gulf sturgeon.

17. Conducted investigations into plasma osmotic and metabolic responses to a wide range of experimental salinities.

18. Evaluating the retention rate of passive integrated transponders (PIT tags) and coded wire tags in cultured Gulf sturgeon.

State of Alabama

Alabama Department of Conservation and Natural Resources

1. Established a regulation in 1972 prohibiting all take of sturgeon within the jurisdiction of the State of Alabama.

2. Conducted literature search and field survey in 1991 and 1992 to determine historic and current status of Gulf sturgeon and possible reasons for apparent decline.

3. Conducted sampling of juvenile Gulf sturgeon on the Alabama River from 1990-1992.

4. Conducted feasibility work in 1992 regarding the use of ADCNR's Claude Peteet Mariculture Center in Gulf Shores, Alabama, as a Gulf sturgeon hatchery for the Mobile system.

Alabama Geological Survey

1. Conducted Gulf sturgeon sampling in the Alabama, Mobile, Conecuh, and Choctawhatchee river systems.

State of Florida

Florida Department of Environmental Protection (formerly Florida Department of Natural Resources

1. Conducted an anadromous fish survey, including Gulf sturgeon, in 1970-1971.

2. Completed the first life history study of Gulf sturgeon in the Suwannee River, Florida from 1972-1973.

3. Conducted a status review of Gulf sturgeon in Florida waters in 1984, and recommended prohibition of all take of the species within the jurisdiction of the State of Florida.

Florida Game and Fresh Water Fish Commission

1. Completed F10-R Anadromous Fish Study from 1964-1967.

2. In 1987 listed the Atlantic sturgeon as a Species of Special Concern in: Official list of endangered and potentially endangered fauna and flora in Florida. Florida Game and Fresh Water Fish Commission. 19 pp.

3. In conjuction with the COE, Mobile District, removed sedimentation and debris from a midstream spring below the JWLD on the Apalachicola River, navigation km 170.6 (navigation mi 106.0), to restore important thermal refuge habitat for the Gulf sturgeon and other anadromous species in January 1994.

Florida Marine Fisheries Commission

1. Established a regulation in 1984 prohibiting all take of sturgeon within the jurisdiction of the State of Florida.

University of Florida

1. Artificial propagation of Gulf sturgeon 1991-1995.

State of Mississippi

Gulf Coast Research Laboratory

1. Distributed Gulf sturgeon posters at boat ramps and other appropriate locations during 1992 in order to acquire information and reports on Gulf sturgeon sightings.

Mississippi Department of Wildlife, Fisheries, and Parks

1. Established a regulation in 1974 prohibiting all take of sturgeon within the jurisdiction of the State of Mississippi.

2. Listed the sturgeon as an endangered species in 1974.

3. Conducted Gulf sturgeon investigation and documentation in the Pascagoula River during 1993.

Mississippi State University

1. Documented Gulf sturgeon presence in the lower Pearl River in 1985 and 1988.

2. Documented incidental catches of Gulf sturgeon in Mississippi in 1989.

3. Investigated and documented Gulf sturgeon in the Pascagoula River in 1993.

<u>State of Louisiana</u>

Louisiana Department of Wildlife and Fisheries

1. Initiated a survey in 1990 to assess the status of Gulf sturgeon in Louisiana waters.

2. Initiated a radio-tracking project in 1992 on Gulf sturgeon in the Pearl River drainage and continuing into 1994.

3. Established a computerized data base in 1991 on all pallid and Gulf sturgeon sightings and captures in Louisiana and continues to be updated as needed.

4. Conducted Gulf sturgeon tagging using T-bar and monel tags beginning in 1992 and ongoing in 1994.

5. Collected blood and tissue samples for genetic analysis beginning in 1991 and ongoing in 1994.

6. Established a regulation in 1990 prohibiting all take of sturgeon within the jurisdiction of the State of Louisiana.

<u>State of Texas</u>

Texas Parks and Wildlife Department

1. Conducted sampling for sturgeon in the Rio Grande in 1992 - 1993.

2. Documented historic distribution of sturgeon in Texas.

<u>U.S. Army Corps of Engineers, Mobile District, Mobile, Alabama</u>

1. Restored access into Battle Bend Cutoff on the Apalachicola River, approximate river km 46.3 (river mi 28.8) in 1987.

2. Conducted flow/velocity studies below the JWLD to document velocities in Gulf sturgeon habitat areas during low flow conditions during November 1991 and October 1992, as

part of a Biological Assessment associated with the Jim Woodruff Powerhouse Major Rehabilitation Evaluation Report.

3. In conjuction with the FGFC, removed sedimentation and debris from a midstream spring below the JWLD on the Apalachicola River, navigation km 170.6 (navigation mi 106.0), to restore important thermal refuge habitat for the Gulf sturgeon and other anadromous species in January 1994.

4. Obtained environmental clearances and undertook action to restore habitat for the Gulf sturgeon and other anadromous species by removal of sediments at the mouth of Blue Spring Run, Apalachicola River, navigation km 157.7 (river mi 98.0) in March 1994, under the Department of the Army/National Oceanic and Atmospheric Administration Cooperative Agreement to Create and Restore Fish Habitat.

5. Initiated Anadromous Fish Hatchery Reconnaissance Study in 1987.

6. During January 1994, the COE proposed that the Waterways Experiment Station (WES) consider in the FY 1995 Environmental Impact Research Program (EIRP) a proposal to document issues affecting the protection of sturgeon related to O&M activities in North American rivers. This proposal was submitted because of similar concerns expressed by other COE divisions and districts that operation and maintenance (O&M) projects may impact sturgeon populations. It is also proposed to quantify responses of sturgeon to broad ranges of relevant physical conditions so that risk from O&M activities can be predicted. Districts will be surveyed for specific issues on sturgeon and the scope of problems will be defined. The District has been informed from COE headquarters that funds are available for WES to initiate efforts in FY 1995.

U.S. Army Corps of Engineers, Vicksburg District, Vicksburg, Mississippi

1. Funded a study conducted by WES on Gulf sturgeon in the Pearl River during 1994 and 1995.

U.S. Fish and Wildlife Service

Fisheries Resources Office, Panama City Field Office, Florida

1. First documented in-river habitat usage of Gulf sturgeon in 1977.

2. First documented Gulf sturgeon spawning in the Apalachicola River, Florida in 1977.

3. Investigated methods of externally marking Gulf sturgeon beginning in 1981.

4. Documented the movement of Gulf sturgeon in the Apalachicola River using radio and sonic telemetry devices beginning in 1982.

5. Estimated the Gulf sturgeon population size in the Apalachicola River below JWLD beginning in 1983.

6. Reviewed and validated the morphometric characteristics used in the taxanomic separation of Gulf and Atlantic sturgeon in 1985.

7. Developed field techniques and equipment which aided in the handling of Gulf sturgeon in 1985.

8. Investigated the age structure of Gulf sturgeon in the Apalachicola River by utilizing cross-sections from pectoral fin rays beginning in 1986.

9. Initiated artificial propagation of Gulf sturgeon in 1989.

10. Collected samples for and funded genetic studies on Gulf sturgeon throughout their range beginning in 1990.

11. Collected samples for and funded contaminant tissue analyses of Gulf sturgeon from the Apalachicola and Suwannee rivers, Florida beginning in 1990.

12. Initiated a program through news releases and information posters to document Gulf sturgeon sightings (past and present) from Tampa Bay, Florida to the Mississippi River in 1992.

13. Funded development of a dual radio-sonic telemetry tag in 1992.

14. Compiled and maintained a directory/data base of sturgeon and paddlefish researchers beginning in 1992.

17. Produced a report entitled <u>Gulf Sturgeon Sightings, Historic and Recent - a Summary of Public Responses</u> in 1993.

18. Conducted field investigations to develop a population model for the Gulf sturgeon and to delineate riverine habitat requirements in 1993 and 1994, in cooperation with the NBS, North Carolina Cooperative Fish and Wildlife Research Unit.

Ecological Services, Panama City, Florida

1. Funded preparation of an information report on the Gulf sturgeon, entitled: <u>Gulf of Mexico Sturgeon, *Acipenser oxyrhynchus* (Vladykov), Information.</u> 1980. Unpublished. 15 pp. J.L. Hollowell.

2. Completed a document entitled: Report on the Conservation Status of the Gulf of Mexico Sturgeon *Acipenser oxyrhynchus desotoi* in 1988.

3. **Prepared report entitled, Reconnaissance Report on the Feasibility of Constructing an Anadromous Fish Hatchery Apalachicola River, Florida for the COE, Mobile District in 1989.**

4. **Initiated the proposal to list the Gulf sturgeon under the ESA.**

5. Coordinated development of Gulf Sturgeon Management/Recovery Plan from 1992 to 1995.

Ecological Services, Jacksonville, Florida

1. Prepared the listing package to list the Gulf sturgeon as a threatened species under the ESA (listed September 30, 1991 in conjuction with the Department of Commerce-NOAA).

Ecological Services, Jackson, Mississippi

1. Produced a Mobile River Basin Aquatic Ecosystem Recovery Plan in 1995.

Warm Springs Regional Fisheries Center, Georgia

1. Developed Gulf sturgeon artificial feeding program in 1989.

Welaka National Fish Hatchery, Florida

1. Hormone induced spawning of Gulf sturgeon beginning in 1989.

2. Developed Gulf sturgeon artificial feeding program in 1989.

Gulf Coast Fisheries Coordination Office, Ocean Springs, Mississippi

1. Participated as a technical advisor in development of the Gulf sturgeon Management/Recovery Plan from 1992 to 1995

Memorandum of Understanding (MOU) on Implementation of the Endangered Species Act.

Fourteen federal agencies including the COE, NMFS, FWS, NPS, DOD, MMS, CG and EPA signed the MOU in September of 1994. The purpose of the MOU was to establish a general framework for cooperation and participation among the agencies in accordance with responsibilities under the ESA. The agencies are to work together along with appropriate involvement of the public, states, Indian Tribal governments, and local governments, to achieve the common goal of conserving species listed as threatened or endangered under the ESA by protecting and managing their populations and the ecosystems upon which those populations depend. The cooperating federal agencies involved in recovery of the Gulf sturgeon will now be able to work closer together under the umbrella of this MOU.

II. RECOVERY AND FISHERY MANAGEMENT

OBJECTIVES AND CRITERIA

Objectives constitute those results that are desired to be attained through implementation of the Recovery Plan. Criteria are those factors that define how attaining the objective will be pursued, and what will constitute sucess.

1. Short-term Objective: The short-term recovery objective is to prevent further reduction of existing wild populations of Gulf sturgeon within the range of the subspecies. This objective will apply to all management units within the range of the subspecies. Ongoing recovery actions will continue and additional actions will be initiated as needed.

   Criteria:

   A. Management units will be defined using an ecosystem approach based on river drainages. This approach may also incorporate genetic affinities among populations in different river drainages.

   B. A baseline population index for each management unit will be determined by fishery independent catch-per-unit-effort (CPUE) levels.

   C. Change from the baseline level will be determined by fishery independent CPUE over a three to five year period. This time frame will be sufficient to detect a problem and to provide trend information. The data will be assessed annually.

   D. The short-term objective will be considered achieved for a management unit when the CPUE is not declining (within statistically valid limits) from the baseline level.

2. Long-term Objective A: The long-term recovery objective is to establish population levels that would allow delisting of the Gulf sturgeon by management units. Management units could be delisted by 2023 if the required criteria are met. While this objective will be sought for all management units, it is recognized that it may not be achievable for all management units.

   Criteria:

   A. The timeframe for delisting is based on known life history characteristics including longevity, late maturation, and spawning periodicity.

   B. A self-sustaining population is one in which the average rate of natural recruitment is at least equal to the average mortality rate over a 12-year period (which is the approximate age at maturity for a female Gulf sturgeon).

C. This objective will be considered achieved for a management unit when the population is demonstrated to be self-sustaining and efforts are underway to restore lost or degraded habitat.

3. Long-term Objective B: This is a long-term fishery management objective to establish, following delisting, a self-sustaining population that could withstand directed fishing pressure within management units. Note that the objective is not necessarily the opening of a management unit to fishing, but rather, the development of a population that can sustain a fishery. Opening a population to fishing will be at the discretion of state(s) within whose jurisdiction(s) the management unit occurs. As with Long-term Objective A, this objective may not be achievable for all management units, but will be sought for all units.

Criteria:

A. All criteria for delisting must be met.

B. This objective will be considered attained for a given management unit when a sustainable yield can be achieved while maintaining a stable population through natural recruitment.

C. Particular emphasis will be placed on the management unit that encompasses the Suwannee River, Florida, which historically supported the most recent stable fishery for the subspecies.

These objectives and criteria are preliminary. After better identification of population status and evaluation of the adequacy of the habitat to support self-sustaining populations, these objectives and criteria may be revised. The criteria stated above will be more quantitatively defined through identification of management units and through population assessments in those individual management units.

OUTLINE FOR RECOVERY ACTIONS ADDRESSING THREATS

Recovery Outline Narrative

1.0 Determine essential ecosystems, identify essential habitats, assess population status, and refine life history investigations in management unit rivers.

As an initial step to enhance the long-term recovery of populations of Gulf sturgeon, collection of basic biological information is essential. Without a clear understanding of life history requirements, recovery efforts are severely hampered. Presently, lack of information in the marine environment and sparse information in the riverine environment make it difficult to adequately census populations or to implement appropriate recovery actions. Studies to provide this information should be conducted as soon as possible.

1.1 Identify essential habitats important to each life stage in river basin and contiguous estuarine and neritic waters.

Investigations are needed to locate and describe the micro- and macrohabitat characteristics critical for recovery and maintenance of the Gulf sturgeon. Radio and ultrasonic tracking studies of juveniles and adults will help determine movements and habitat utilization over time. Emphasis should be placed on tracking Gulf sturgeon in the estuarine and marine environment where it is believed that most feeding and growth occurs, and where the least information is available. Spawning areas and larval and post-larval movements and distribution within rivers must be determined. When a sufficient number of animals has been monitored and distributions identified, habitat characterization studies can be used to better define essential habitat requirements. Significant ecosystems for the recovery of the Gulf sturgeon will be identified once essential habitats are defined in riverine, estuarine, and marine environments

1.1.1 Conduct and refine field investigations to locate important spawning, feeding, and developmental habitats.

Gulf sturgeon have been successfully tracked with radio and ultrasonic transmitters in riverine systems. These studies have been limited to a very few locations, and usually for a short time spans. Multi-year tracking studies in the estuarine and marine environment have never been accomplished. Knowledge of spawning areas, developmental habitat requirements and feeding requirements are essential to the recovery of Gulf sturgeon in all river basins across the range of the species. Tracking studies appear to be the best way to initially locate important habitat. Technological advances in telemetry should facilitate long-term tracking studies to provide the needed information. The FWS and NBS should expand their efforts to identify and inventory essential habitats of Gulf sturgeon. The Gulf states resource management agencies should continue or initiate studies to identify essential habitats in their respective states. The CCC should continue their multi-year monitoring program on the Suwannee River. New field work by other researchers such as universities and non-government organizations (NGOs) should incorporate this research need into their plans. The NMFS should work with FWS and NBS to identify marine habitats used by adult Gulf sturgeon during winter migration. The MMS should seek funding to obtain this information because of the potential for impacts to the Gulf sturgeon from outer continental shelf oil and gas operations and other non-energy mineral mining activities.

1.1.2 Characterize riverine, estuarine, and neritic areas that provide essential habitat.

When areas of utilization have been delineated (Task 1.1.1), characterization of these habitats should be conducted. Characteristics of the areas regarding particular life history requirements of Gulf sturgeon at various life stages must be determined. Among the parameters that may be important include substrate, depth, instream flow, current, pH, temperature, turbidity, and food availability. The Gulf states resource management agencies, FWS, NMFS, NBS, CCC, NGOs, and universities should refine their studies or surveys to provide these data.

1.2 Conduct life history studies on the biological and ecological requirements of little known or inadequately sampled life stages.

Because of the difficulty in collecting eggs, larvae, and adequate numbers of Gulf sturgeon less than a year old, essentially nothing is known about requirements of these life stages in the wild. Year-class strength is established during these stages, and water temperature, salinity, flow, turbidity, and other factors affect survival rates. As outlined in Task 1.1, intensive field investigations must be initiated to locate and characterize habitats used by early life stages. Likewise laboratory studies on wild and cultured Gulf sturgeon must be conducted to evaluate habitat requirements and tolerances. The University of Florida, NBS, and FWS should expand ongoing investigations into the biology and ecology of Gulf sturgeon. Non-fatal sampling techniques to examine stomach contents need to be determined. Diet studies of fish captured in estuaries should be expanded. Diet of Gulf sturgeon captured offshore (neritic environments) should also be evaluated, not only to assess food preferences, but also to determine habitat use.

It is known that subadult and adult Gulf sturgeon spend winters feeding in estuarine and marine waters. Little is known about specific areas and habitat requirements. Ultrasonic techniques should be improved and studies conducted to document marine habitats frequented by Gulf sturgeon. Identified habitats must be described by depth, water quality, substrate, and food availability. The FWS and NBS should continue ongoing marine habitat investigations of Gulf sturgeon. The NMFS should initiate marine habitat investigations of Gulf sturgeon.

1.3 Survey, monitor, and model populations.

Intensive field investigations have concentrated on Gulf sturgeon life history in the Suwannee and Apalachicola rivers in Florida. Additionally, long-term monitoring of Gulf sturgeon in these systems has resulted in reliable population estimates with which population models are being developed. Outside these systems, few studies have been conducted on the Gulf sturgeon. Information such as distribution, relative abundance, age structure and other biological information should be compiled to identify baseline population status and identify index monitoring sites to evaluate success of recovery and management programs.

1.3.1 Develop and implement standardized population sampling and monitoring techniques.

The assessment of Gulf sturgeon populations Gulfwide are essential to develop and evaluate recovery and management efforts. Standardized programs to address size, age and sex composition, and stock size must be developed so that the condition of each stock can be evaluated over time and compared with those in other river systems. Government agencies, NGOs, and universities investigating Gulf sturgeon should participate in a coordinated effort to develop standardized sampling and monitoring techniques and conduct appropriate programs. Standard operating procedures will facilitate application of statistical data set comparisons between various Gulf coast river systems. In addition, fishery management/recovery decisions could be more accurately formulated with uniform data collection and reporting procedures. The FWS should take the lead in coordinating, preparing and distributing a standardized sampling and monitoring protocol document. The Gulf states resource management agencies should evaluate the status of populations of Gulf sturgeon in their streams and coastal waters. The FWS and NBS in conjunction with other researchers should verify current aging techniques for Gulf sturgeon.

1.3.2 Develop population models.

Modeling is needed to better assess fishery restoration and management options. Capture-recapture models can estimate survival, abundance and recruitment of Gulf sturgeon. Population models should be developed to forecast the future condition of Gulf sturgeon populations and provide estimates on potential rates of recovery. Appropriate models will also help identify future research needs. The FWS and NBS should continue to take the lead in formulating peer accepted population models for the Gulf sturgeon.

1.4 Continue experimental culture of Gulf sturgeon.

Successful artificial propagation of Gulf sturgeon was first accomplished in 1989. Additional work is still needed to refine culture techniques, develop handling and holding procedures for fry and broodstock, maintaining genetic diversity of broodstock, research nutritional requirements and initiate fish health management. In addition, research is needed to document the optimum chemical and physical parameters necessary for maintaining growth and survival of Gulf sturgeon under artificial and natural conditions.

1.4.1 Continue culture of Gulf sturgeon.

State, federal, and NGOs should continue to develop culture techniques for Gulf sturgeon in accordance with the Gulf Sturgeon Hatchery Guidelines, Hatchery Manual for White Sturgeon protocols addressed in the Gulf Sturgeon Recovery Plan, and state and federal laws and regulations. Efforts should be directed towards filling data gaps (i.e. hormone dosages and types, incubation temperatures, egg de-adhesion methods, broodstock reproductive staging, elimination of stress related to capture, handling, and holding, among other factors).

1.4.2 Identify the physical, chemical and biological parameters necessary to maintain growth, health and survival of Gulf sturgeon reared under artificial conditions.

Studies are needed to determine the optimum water quality conditions necessary to maintain growth and survival of fry and fingerlings. In addition, nutritional requirements and artificial feeding methods need to be identified. Research is required to document carrying capacity for various fish rearing facilities, and hauling densities of fry and fingerlings. The FWS, researchers, and universities should continue to implement additional studies to address this need. Also, the FWS should take the lead in providing updated information on artificial propagation of Gulf sturgeon.

1.4.3 Identify and test internal and external markers or techniques useful for differentiation of wild and hatchery-produced Gulf sturgeon.

The identification of non-genetic internal and external markers to differentiate between wild and hatchery-produced Gulf sturgeon is important in the development and regulation of hatchery programs. Unique markers (i.e. PIT tags, coded wire tags, and chemical marking) could allow investigators, law enforcement officers, and others to distinguish hatchery-reared fish from wild stocks. In addition, these markers or techniques may be used in selective enhancement programs and provide a means to evaluate introductions. The FWS and other researchers should continue to investigate and develop useful internal and external markers or techniques.

1.5 Identify genetic characteristics of wild and hatchery-reared Gulf sturgeon.

Research is needed to determine whether or not significant genetic differences exist among Gulf sturgeon from throughout the range of the subspecies. Determining whether genetic differences exist among populations is essential to ensure successful recovery and management of the subspecies. Genetically distinct management units may be identified and could affect reintroduction and/or population augmentation.

1.5.1 Conduct a Gulfwide genetic assessment to determine geographically distinct management units.

Determination of the genetic structure for Gulf sturgeon is essential in formulating future management decisions for the subspecies. It is important that sound restoration efforts of Gulf sturgeon address the genetic structure of the subspecies in order to identify and maintain genetic integrity and diversity. Mitochondrial DNA analysis of Gulf sturgeon should be continued with emphasis placed on obtaining Gulf sturgeon tissues and/or blood from the following river systems:

1. Pascagoula River, Mississippi.
2. Mobile and Alabama rivers, Alabama.
3. Ochlocknee River, Florida.
4. Escambia River, Florida.

A genetic tissue bank should be established and curated where state or federal agencies deposit tissue or blood for genetic analysis. The Gulf states resource management agencies, universities, NGOs, NBS, FWS, and other Gulf sturgeon researchers should establish tissue collection protocol and insure that tissue samples are collected whenever possible.

1.5.2 Assess the potential to develop genetic markers to differentiate wild and hatchery-produced Gulf sturgeon.

The development of genetic markers for differentiating between wild and hatchery produced Gulf sturgeon may be important in the development and regulation of hatchery programs. A unique genetic marker could allow investigators, law enforcement officers, and others to distinguish hatchery reared fish from wild stocks. In addition, hatchery stocks possessing a different genetic mark from wild fish may be used in selective enhancement programs and provide a means to evaluate their introductions. The FWS and NMFS should continue to investigate the potential of viable genetic markers.

2.0 Protect individuals, populations, and their habitats.

In efforts to recover listed species, protection is the most obvious initial step. By virtue of their endangered or threatened status, species may not be able to sustain continuing losses of individuals, and steps should be taken immediately to eliminate any known preventable take. Initial measures to protect individuals, populations, and their habitats can be strengthened or reduced as new information is collected.

2.1 Reduce or eliminate unauthorized take.

Under the ESA, take means "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." "Harm" in the definition of "take" in the ESA means an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. "Harm" in the definition means an act which actually kills or injures wildlife. Such act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. In the case of the Gulf sturgeon, the immediate concern is with lethal or injurious take by non-directed fisheries. Directed fisheries for listed species are prohibited by virtue of the listing. However, a number of fisheries targeting other species use fishing gear that take Gulf sturgeon.

2.1.1 Increase effectiveness and enforcement of state and federal take prohibitions.

Directed take of the Gulf sturgeon is prohibited under the ESA and laws or regulations of Louisiana, Mississippi, Alabama, and Florida. All states within the geographic distribution of the Gulf sturgeon have cooperative agreements with the FWS that require enforcement of federal endangered species laws. Both federal and state officials are empowered to enforce prohibitions on the take of Gulf sturgeon. Appropriate steps should be taken to support and enhance enforcement activities related to restoration and protection of Gulf sturgeon. The Gulf states resource management agencies should evaluate their enforcement programs and if needed, implement appropriate enhancements or actions. The FWS and NMFS should insure that during ESA section 7 consultations, incidental take is stipulated to provide full protection of the species.

On July 1, 1975, the Atlantic sturgeon (*Acipenser oxyrinchus*, including the Gulf sturgeon) was included in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The effect of this listing is that CITES permits are required before international shipment may occur.

2.1.2 Reduce or eliminate incidental mortality.

Incidental catch and mortality of Gulf sturgeon is a difficult or cryptic problem to address because it requires a knowledge of effort and catch composition in a variety of different fisheries. Gear types used in many fisheries are capable of capturing Gulf sturgeon, and it is essential that the magnitude of the problem in each fishery is known before effective steps can be taken to reduce or eliminate mortality. A limited observer program may be needed to evaluate the amount/extent of incidental take or mortality in some fisheries and navigation-related and other activities. When problem fisheries or other activities have been identified, gear or equipment modifications, seasonal restrictions, limited gear or equipment deployment times, and other measures may be employed to reduce mortality of Gulf sturgeon and allow the affected fisheries or other activities to continue to operate.

If incidental take is found to be related to any fishery, the NMFS and the Gulf states should promulgate adequate regulations that protect the Gulf sturgeon from such incidental take. The NMFS should also evaluate Turtle Excluder Devices (TEDs) in commercial shrimp nets to determine if they are effective in allowing Gulf sturgeon to escape from trawls. If they are not effective, funding should be sought to investigate the appropriate gear technology. The NMFS should also fund an observer program, enforcement of regulations, and other necessary actions which reduce or eliminate incidental take of Gulf sturgeon during fishing operations.

In addition, the NMFS and FWS in cooperation with the responsible federal agency should develop methodologies that would cause Gulf sturgeon to avoid areas during navigation-related (includes O&M) activities, Clean Water Act (CWA) Sections 10 and 404, or other construction activities. The NMFS and FWS should assure that the objective of ESA section 7 consultation is to reduce or eliminate incidental take during such activities. As an example, section 7 consultation for a dredging project may result in the COE permitting the activity to occur only during seasons when Gulf sturgeon are not present in the action area.

2.2 Identify and eliminate known or potentially harmful chemical contaminants, and water quantity and water quality problems which could impede recovery of Gulf sturgeon.

Chemical contaminants, water quantity, and water quality factors may have contributed to the decline or are limiting the recovery of Gulf sturgeon. These factors include pesticides (organochlorines), metals (lead, mercury, etc.), industrial byproducts, temperature, pH, suspended solids, dissolved oxygen, water depth, and water velocity. Review of existing data and information is necessary to refine or identify the chemical and water quality and quantity requirements of Gulf sturgeon.

An information search for each management unit or coastal habitat area regarding potential types of chemical contaminant loading, including chemicals from point sources, agriculture, silviculture, industrial activities and urbanization, should be conducted. Existing chemical contaminant field evaluation reports (water, sediment or biota studies) should be examined and the information utilized to make decisions related to field sampling and chemical analysis. Field sampling of water, sediments, and sentinel and/or surrogate species should be conducted, as necessary, to fill critical information gaps. State agencies in Louisiana, Mississippi, Alabama, and Florida, with assistance from the Environmental Protection Agency (EPA) and FWS should collect existing information and provide an assessment report with recommendations. The FWS should provide coordination between the federal and state agencies as needed, compile state reports, and identify a consensus priority listing of chemical contaminant sources that may have impacts on Gulf sturgeon in the river systems. The EPA "Priority Pollutants" for each management unit or habitat area should be assessed by chemical analyses for Gulf sturgeon and other benthic species. The FWS and EPA, using the compiled contaminant data, should prepare the list and conduct necessary analyses.

2.2.1 Identify potentially harmful chemical contaminants and water quality and quantity changes associated with surface water restrictions.

A comprehensive inventory of river basins with existing surface water restrictions is needed to document physical and biological impacts that may negatively affect recovery and management of Gulf sturgeon. The GSMFC, FWS, and COE should coordinate preparation of this inventory with GSMFC taking the lead for final product completion.

2.2.2 Identify and eliminate potentially harmful point and non-point sources of chemical contaminants.

Significant point sources and high-impact non-point source areas of contaminant introductions should be identified. Appropriate actions to reduce or eliminate the contaminants should be taken. With the results of 2.2.1, EPA and state agencies in Louisiana, Mississippi, Alabama, and Florida should take actions to enforce existing regulations or promulgate new ones.

2.2.3 Assess selected contaminant levels in Gulf sturgeon from management units.

Gulf sturgeon tissue analyses should be conducted to evaluate selected chemical contaminants. Appropriate actions should be taken to reduce or eliminate contaminant sources. The EPA should take the lead in efforts to reduce or eliminate identified contaminant sources through their regulatory authorities. The EPA could also assist state agencies in Louisiana, Mississippi, Alabama, and Florida in enforcement of state regulations. During the Triennial Review of state water criteria, EPA should ensure that the states have incorporated adequate water quality standards to protect the Gulf sturgeon and its benthic habitat.

Routine, standardized inspections should be conducted on all incidental catches of Gulf sturgeon (alive or dead) for the presence of gross lesions, tumors or other abnormalities to focus evaluation on chemical contaminants.

Histopathological examinations of liver tissue for cases of incidental Gulf sturgeon mortalities should be conducted to detect the presence of cellular abnormalities or carcinogenic cells.

Chemical analyses of selected tissues should be conducted from incidental mortalities of Gulf sturgeon. The FWS should take the lead in developing protocol to collect samples, conduct training if necessary, process samples for analyses, and prepare summaries of results. Wherever possible, Gulf state resource management agencies should conduct similar analyses.

Appropriate surrogate species should be utilized to better define bio-accumulation of contaminants in particular river basins. An extrapolation formula for estimating potential chemical contaminant impacts to Gulf sturgeon should be developed. The FWS and EPA should lead the efforts to identify appropriate surrogate species, conduct bio-accumulation studies, and develop an extrapolation formula. Appropriate peer review should be conducted during formula development.

2.2.4 Identify and eliminate known and potential impacts to water quantity and quality associated with existing and proposed developments, agricultural uses, and water diversions in management units.

Domestic and industrial effluent, rural and urban run-off, and inter- and intra-water diversions affect the clarity, pH, biological oxygen demand, nutrient and contaminant composition, temperature, sediment loads, and seasonal quantity of river waters. A comprehensive inventory of known or potential problem areas associated with these factors is needed. Once identified, actions to reduce or eliminate problems and promote wise land use should be taken. With the results of 2.2.1, EPA and Gulf states resource management agencies should take actions to enforce existing regulations or promulgate new ones.

Water quality and sediment factors resulting from point and nonpoint sources may negatively affect Gulf sturgeon habitat. Examples include total dissolved solids, suspended solids, turbidity, siltation, pH, temperature, and changes in sediment types. Studies to assess the effect of river water and sediment quality should be conducted to determine the habitat suitability for Gulf sturgeon.

2.2.5 Assess the relationship between groundwater pumping and reduction of groundwater flows into management units, and quantify loss of riverine habitat related to reduced groundwater in-flows.

Groundwater diversions which affect flows into management unit rivers should be identified. The loss of riverine groundwater flows attributed to diversions should be quantified and its effect on Gulf sturgeon evaluated. The U. S. Geological Survey (USGS) should take the lead in implementing appropriate studies including modelling. The Tri-State Study for the Alabama-Tallapoosa-Coosa and Apalachicola-Chattahoochee-Flint river basins funded by the COE and Alabama, Georgia, and Florida should incorporate an effort to provide a preliminary assessment of the effects of groundwater pumping into the groundwater scope of work plan.

2.2.6 Conduct studies to determine the effects of known chemical contaminants in water from management unit rivers on Gulf sturgeon or a surrogate species.

After identification of priority contaminants, physiological and behavioral responses of Gulf sturgeon life stages to long-term exposures to such chemicals should be determined. In particular, newly fertilized eggs, Gulf sturgeon larvae, and juvenile Gulf sturgeon should be tested. The EPA should work with the FWS to conduct bioassays of water from the management unit rivers to determine effects on Gulf sturgeon.

2.3 Develop a regulatory and/or incentive framework to ensure that essential habitats, streamflow, and groundwater in-flows are protected.

Where existing laws and regulations are inadequate to meet recovery objectives, appropriate state and federal agencies should propose new incentives, laws, and/or regulations.

2.3.1 Utilize existing authorities to protect habitat and, where inadequate, recommend new incentives, laws, and regulations.

The ESA provides for the protection and recovery of the Gulf sturgeon and its habitats. Likewise individual Gulf states have regulations and laws for that purpose. Adequate funding levels must be provided to enforce existing protection measures and laws. Federal and state natural resource law enforcement programs are understaffed and underbudgeted to adequately enforce laws protecting the Gulf sturgeon and its habitats. Even with adequate funding, existing authorities may be inadequate to fully protect the Gulf sturgeon and its habitats. Adoption of new incentives, laws or regulations may be necessary to ensure the recovery of the species. Protection measures should be based on the biological requirements of the subspecies and not political boundaries. The FWS should ensure protection of the Gulf sturgeon through the ESA section 7 consultation process with other federal agencies including the COE (federal projects, Section 10/404 permits), MMS (OCS oil and gas lease sales), EPA (National Pollutant Discharge Elimination System permits, Triennial Review).

2.3.2 Identify, protect and/or acquire appropriate land or aquatic habitats on an ecosystem approach.

Habitat components of the Gulf sturgeon which provide essential life requirements should be considered as part of and dependent on a fully functioning ecosystem . These ecosystems should be protected and/or acquired. The Gulf states resource management agencies, FWS, and NMFS should seek appropriate avenues of funding and take action to acquire, manage, and protect identified significant habitats or their ecosystems as appropriate.

For example, spawning habitats should receive maximum protection from disturbance. In order to protect specific habitats, the ecosystem where it occurs also requires protection. Thus, protection of spawning habitats of the Apalachicola River would include the upper 20 km (12.4 mi) of the river and its surrounding basin components. Another example includes the maintenance of habitats such as the springs that occur in the Suwannee River. To protect these springs, it is essential to maintain other ecosystem components including upstream water quality, groundwater flows and quality, and adjacent floodplains.

2.4 Restore, enhance, and provide access to essential habitats.

Gulf sturgeon have evolved within Gulf coast drainages exhibiting seasonal patterns of high and low flows, temperature regimes, sedimentation, and other physical factors which historically may have been much different than those which exist today. The restoration and enhancement of some river and stream habitats, particularly benthic habitat, within the historical range of the Gulf sturgeon may be necessary before its recovery is successful. Within some drainages, man's alterations (mainstem dams, low-head diversions) may be preventing Gulf sturgeon from gaining access to important habitats essential to some aspect of its life history. If such structures are identified as impeding migration or preventing access to critical habitats, action should be taken to restore the natural hydrography or provide a viable bypass route around the structure.

2.4.1 Identify dam and lock sites that offer the greatest feasibility for successful restoration of and to essential habitats (i. e., up-river spawning areas).

Mainstem and low-head diversion dams that are known to be impeding potentially viable Gulf sturgeon populations from reaching historically essential habitats need to be identified. The extent of important habitat types upstream from such structures (e.g., potential spawning sites and summer refugia) should be evaluated.

The GSMFC should take the lead in identifying these sites throughout the Gulf states and preparing summary and recommendations. Federal and non-federal permitted dams should be identified. The COE, FERC, and entities such as the Pearl River Valley Water Supply District should investigate ways of mitigating impacts of federal and private water resource projects or permitted activities on Gulf sturgeon populations.

2.4.2 Evaluate, design, and provide means for Gulf sturgeon to bypass migration restrictions within essential habitats.

The structures preventing upstream migrations to essential habitats should be modified or removed to allow for Gulf sturgeon passage. Specific modifications will depend on the type of obstruction, river hydrology and the importance of the habitat to the recovery of the species in that particular ecosystem. Studies regarding Gulf sturgeon behavior may be required to assist in development and design of fish passages. Modifications which provide for both up- and downstream travel by large and small fish need be considered.

First, an assessment of existing modifications should be conducted. The assessment should consider the effectiveness of the modification for use by other migratory species such as shad and striped bass. Designs should be solicited from engineering and environmental consultants. Passage structures which show promise must be evaluated to document the relative degree of usage by Gulf sturgeon. The NMFS, COE, NBS, FWS, and Federal Energy Regulatory Commission (FERC) should investigate the use of potential passage structures and initiate action or studies to assess the structure's effectiveness for Gulf sturgeon passage.

2.4.3 Operate and/or modify dams to restore the benefits of historical flow patterns and processes of sedimentation.

The operating schedules of the dams need to be evaluated to determine if water releases are benefiting the life history requirements of the Gulf sturgeon. The operations of existing structures found to be detrimental to the life cycle of Gulf sturgeon should be evaluated to determine if modifications to approximate historical flow and sedimentation patterns are possible. The COE and FERC in coordination with the GSMFC, Gulf states resource management agencies, FWS, and NMFS should identify potential modifications to and/or operations of dams and initiate action or studies to assess the feasibility for implementation.

2.4.4 Identify potential modifications to specific navigation projects to minimize impacts which alter riverine habitats or modify thermal or substrate characteristics of those habitats.

Navigation projects that have altered or modified the thermal characteristics or natural substrates of rivers should be evaluated to determine if modifications to approximate historical conditions are possible. The COE should assist the FWS in its efforts to define and protect Gulf sturgeon spawning and other essential habitats in federal project areas. The COE should study, seek funding, implement or take appropriate remedial actions to rectify navigation projects where feasible.

2.4.5 Restore the benefits of natural riverine habitats.

Dams and channel modifications have reduced habitat diversity within the range of the Gulf sturgeon. Diversity of riverine habitat (e.g., main channel, side channel, backwater and braided channel) promotes a corresponding faunal diversity. The Gulf sturgeon evolved in natural riverine settings where such diversity was prevalent. Gulf sturgeon survival could be expected to be compromised if the benefits of riverine habitat diversity are not restored. The FWS should work with the COE to identify ways to restore and protect natural river habitat diversity.

2.4.6 Seek optimum consistency between the purposes of federal and state authorized reservoirs, flood control projects, navigation projects, hydropower projects, and federal and state mandated restorations of fish populations.

Many water projects, such as hydropower and flood control dams and navigation activities, are authorized by state and federal governments for their respective purposes. Also, there are many state and federal programs authorized to restore declining fish populations. Examples include species listed under the ESA, anadromous fisheries addressed under the Anadromous Fish Conservation Act, and coastal fisheries addressed under the Interjurisdictional Fisheries Act and the Magnuson Fisheries Conservation and Management Act.

All government authorized and proposed projects and mandates should be reviewed in order to evaluate the potential to achieve recovery of Gulf sturgeon. The GSMFC should facilitate a multi-agency effort to identify project mandates and prepare a summary and recommendation report in partnership with the appropriate state and federal agencies. Recommendations should be forwarded to each of the States of Louisiana, Mississippi, Alabama, and Florida's State legislature and congressional delegation.

2.5 Maintain genetic integrity and diversity of wild and hatchery-reared stocks.

Major conservation issues that must be addressed by this recovery program relative to health of stocks, genetic conservation of stocks and displacement of stocks. A major concern in any stock restoration and enhancement program is the potential impact of introduced fish on existing wild stocks. This impact can affect wild stocks by a variety of mechanisms:

1. Disease and parasite transfer.
2. Behavioral and ecological interference.
3. Genetic consequences of interbreeding, reduction in gene flow, introduction of strains susceptible to disease.

Problems resulting from failure to protect habitat, to control fishing pressure, to ensure correct management of water resources, to control environmental contamination, and to effectively manage other parameters have contributed to reductions in stocks of Gulf sturgeon. These problems are readily evident and appropriate actions can be taken to correct them. At this point, the potential adverse effects of initiating a stocking program are unknown. The potential effects of initiating any stocking program should be evaluated. An experimental hatchery and strictly limited release program to the wild is prudent until such time as stocking has been thoroughly evaluated.

2.5.1 Evaluate the need to stock hatchery-produced Gulf sturgeon considering habitat suitability and current population status.

An assessment of whether stocking hatchery-produced fish will benefit the overall recovery of the Gulf sturgeon is paramount to the future development of Gulf sturgeon hatchery programs. An evaluation of whether the rivers to be stocked have suitable habitat to support the stocked fish, natural reproduction, and any progeny should be conducted. The recovery of the subspecies cannot be based on a "put and take" Gulf sturgeon fishery. Government agencies, NGOs, and universities investigating Gulf sturgeon should conduct an evaluation of each river system that is under consideration for stocking on the ability of the system, at its current status, to support the stocked fish and assure that natural reproduction can occur. Only ongoing improvements to the river systems should be included in the analyses. Each of the Gulf states resources management agencies should evaluate the river systems in their states. The FWS should take the lead in coordinating the assessment and preparing a summary finding report. No stocking should be conducted without approval by appropriate state agencies.

If it is determined that there is a need for stocking, the stocking should be secondary to other recovery efforts that identify essential habitats and emphasize habitat restoration. The COE should continue to work with the FWS in efforts to construct a permanent hatchery on the Apalachicola River to help in the restoration and maintenance of the Apalachicola River Gulf sturgeon population if it is determined that stocking is necessary for recovery of the subspecies.

2.5.2 Develop policy and guidelines for hatchery and culture operations related to stocking.

Raising hatchery produced fish to a size large enough to overcome lack of suitable habitat increases survival. Also, at larger sizes, these fish can be tagged and recovered, enabling assessment of the efficacy or success of the stocking effort. Peer review and evaluation of a particular stocking effort should be included in any proposal to release hatchery-reared Gulf sturgeon. Gulf states resource management agencies, GSMFC, FWS, NMFS, NGOs, universities, and other involved researchers should prepare a hatchery and culture operations plan relating to stocking policy/guidelines. The FWS should take the lead in coordinating, seeking peer review, and completing the document.

2.5.3 Develop and implement a regulatory framework to eliminate accidental and intentional introductions of non-indigenous stock or other sturgeon species.

Release of hatchery-reared fish without a program of monitoring does not fulfill government's role as a steward of renewable natural resources. Monitoring and systematic assessment of stocks will assist in determining the impact of accidental and intentional releases of non-indigenous stock or other sturgeon species. This recovery plan recognizes that it is irresponsible to intentionally release fish without review or concurrence from the recovery team or coordinator, and therefore undocumented intentional releases should not occur. In the case of federal agencies who undertake actions that may affect a listed species (stock introductions), consultation with FWS and/or NMFS is required under section 7 of the ESA.

At a minimum, the recommendations of the Aquatic Nuisance Species Task Force (ANSTF) which was established under the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 should be conducted. The task force developed recommendations regarding direct introductions and indirect, accidental release from public and private sector facilities. All State agencies within the subspecies' range and GSMFC, FWS, NBS, NMFS, NGOs, universities, and other involved researchers should prepare a consensus policy regarding introduction of non-indigenous sturgeon stocks into the range of Gulf sturgeon in accordance with the options or actions identified by the ANSTF to reduce risks and adverse consequences associated with introductions. States should implement necessary actions for promulgating regulations consistent with the policy.

3.0 Coordinate and facilitate exchange of information on Gulf sturgeon conservation and recovery activities.

Any research and/or management activities on fish species which transcend jurisdictional boundaries must be coordinated. Management and recovery actions must be consistent across the range of the subspecies in order to be effective. Gulf sturgeon recovery efforts will be enhanced by the coordination of activities and exchange of information regarding the biology and management of all sturgeon species.

3.1 Coordinate research and recovery actions.

Coordination activities involving state and federal resource management agencies, NGOs, and universities with an interest in the Gulf sturgeon should be conducted at least every two years. Such coordination will provide for studies and management plans which will reduce duplication of effort, enhance cooperation, and optimize agency manpower and funding. The FWS and GSMFC should take the lead in conducting the coordination activities.

3.2 Develop an effective communication program or network for obtaining and disseminating information on recovery actions and research results.

All recovery participants including state and federal agencies, NGOs, and universities working on Gulf sturgeon are strongly urged to publish research findings in technical publications. Unpublished reports (gray literature), bibliographies, and available data on Gulf sturgeon should be compiled and published or otherwise made available to all participants. Acquiring, disseminating, and maintaining information regarding Gulf sturgeon recovery activities should be centralized. The FWS should take the lead in collecting and centralizing information regarding Gulf sturgeon recovery activities.

In order to ensure effective communication among the various entities involved in Gulf sturgeon research, recovery and management, a newsletter should be developed and disseminated on a regular basis. This newsletter would provide all interested parties with the most up-to-date information regarding progress toward achieving the goals of the Recovery Plan. The FWS should take the lead in preparing, printing, and disseminating the newsletter and coordinating with other existing sturgeon newsletters.

3.3 Develop a non-scientific constituency and public information program directed toward enhancing recovery actions.

In order for Gulf sturgeon recovery actions to be successful, the general public must be aware of such actions and understand the need for them. An information and education program must be developed to inform the public of the causes of the decline of Gulf sturgeon, to increase the public's awareness, understanding, and involvement in Gulf sturgeon recovery efforts and to promote wise use of land in watersheds. Educational materials such as brochures, newspaper and magazine articles, publications, posters, and slide and television presentations, among others, must be produced and disseminated to target audiences, such as commercial and recreational fishermen, boaters, and civic organizations. The Gulf states resource management agencies, FWS, NBS, and NMFS should seek funding for the development of educational material for dissemination to the public. The FWS or GSMFC should take the lead in coordinating this effort providing a centralized location for storage of information if necessary.

4.0 Implement recovery program.

Existing budgets of involved agencies and other parties are not capable of fully funding the Gulf sturgeon recovery plan. Competition for funding under the ESA is intense, partly due to the low level of appropriations to the program and the increasing number of listed species. In order to assure that actions which would result in recovery of the Gulf sturgeon are implemented, funding for activities must be secured and a designated lead recovery office must be identified. Involvement of NGOs, and universities should be solicited.

4.1 Designate and fund a Gulf sturgeon recovery lead office.

Funding to support a FWS recovery lead office must be identified to coordinate a multi-agency, multi-disciplinary recovery implementation committee. The lead office should document all research, recovery, and management information and plans. Work would be combined with other FWS duties. The lead office should be in a location which facilitates coordination with all Gulf sturgeon activities. The lead office should be funded until the Gulf sturgeon is considered recovered according to the Recovery Plan.

4.2 Seek funding for Gulf sturgeon recovery activities.

The recovery lead office, with support from involved agencies, NGOs, universities, and the public should seek to bring high visibility to the need for funding of Gulf sturgeon recovery activities. Funding strategies to acquire Congressional appropriations and other funding sources should be developed. The recovery lead office should facilitate this effort and coordinate a unified funding package for Gulf sturgeon recovery activities in the southeast.

4.3 Implement projects or actions which will achieve recovery plan objectives.

Based on the recovery plan, a series of specific projects will be identified which could bring about improvements in the habitat or stock condition of Gulf sturgeon in specific river systems throughout the range of the species. Projects should be submitted to the appropriate agencies or funding sources for consideration. The Gulf states resource management agencies should be given first opportunity to implement the identified projects, through joint efforts with FWS, NBS, NMFS, universities, NGOs, or other interested researchers.

4.4 Develop and implement a program to monitor population levels and habitat conditions of known populations in the management units as well as newly discovered, introduced, or expanding populations.

The status of the subspecies and its ecosystems should be monitored to assess any progress toward recovery while recovery actions are ongoing and following completion of actions. A standardized assessment program should be designed by a multi-agency group coordinated by the recovery lead office and the GSMFC. The Gulf states resource management agencies, federal agencies, universities, NGOs, and other researchers should conduct an annual assessment of the management unit population levels in their area of responsibility or as appropriate. The recovery lead office should maintain, collate, and review the assessments preferably on an annual basis but at least every two years. This information should be summarized for distribution and used in the Congressionally required biennial species status reports.

5.0 Monitor recovery program.

A recovery plan benefits a species only if it is implemented. The plan and its implementation must be strong enough to provide adequate guidance to species managers but be flexible enough so that it may be changed or revised to recover the species. In addition, the FWS and NMFS are required by Congress to track the status of all listed species and the implementation of recovery plans, financial expenditures for each species or clusters of species, and status of recovered species.

5.1 Assess overall success of the recovery program and recommend action.

The recovery program must be evaluated periodically to determine if it is making progress in achieving recovery objectives and to recommend future actions. These actions could include changes in recovery objectives, continuing or increasing protection, implementing new measures, revising recovery plans and recommending delisting. The recovery program should be preferably evaluated annually but at least biennially. The recovery lead office should be responsible for collection of the required information and preparation of the Congressional reports. As part of this effort, the lead office should prepare standardized reporting forms so that the affected parties can easily provide the necessary information. Reporting requirements should continue for five years after the delisting of the Gulf sturgeon.

LITERATURE CITED

Appey, R.G. and M.J. Dadswell. 1978. Parasites of *Acipenser brevirostrum* LeSueur and *Acipenser oxyrhynchus* Mitchell (Osteichthyes: Acipenseridae) in the Saint John River Estuary, N.B., with a description of *Caballeronema pseudoargumentosus* Sp. N. (Nematoda: Spirurida). Can J. Zool. 56(6):1382-1391.

Armstrong, F.A. 1979. Effects of mercury compounds on fish. In Nriagu (ed) The Biogeochemistry of Mercury in the Environment, pp. 657-670. New York. Elsevier/North Holland Biomedical Press.

Barkuloo, J.M. 1988. Report on the conservation status of the Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi*. U.S. Fish and Wildlife Service. Panama City, Florida. 33 pp.

Bateman, D.H. and M.S. Brim. 1994. Environmental contaminants in Gulf sturgeon of Northwest Florida 1985-1991. U.S. Fish and Wildlife Service. Pub. No. PCFO-EC 94-09. Panama City, Florida. 23 pp. with appendices.

Bigelow, H.B., M.G. Bradbury, J.R. Dymond, J.R. Greenley, S.F. Hildebrand, G. W. Mead, R.R. Miller, L.R. Rivas, W.C. Schroeder, R.D. Suttkus, and V.D. Vladykov. 1963. Fishes of the Western North Atlantic. Number I. Part 3. Soft-rayed bony fishes. Sears Foundation for Marine Res., Yale University, New Haven, CT. 630 pp.

Birstein, V.J. 1993. Sturgeons and paddlefishes: threatened fishes in need of conservation. Conservation Biology. Vol. 7. No. 4. December. pp. 773-787.

Boschung, H. (ed). 1976. Endangered and threatened plants and animals of Alabama. Bulletin Alabama Museum of Natural History. Number 2. University of Alabama. p. 57.

Bowen, B.W. and J.C. Avise. 1990. Genetic structure of Atlantic and Gulf of Mexico populations of sea bass, menhaden, and sturgeon: influence of zoogeographic factors and life history patterns. Marine Biology 107:371-381.

Burgess, R.F. 1963. Florida sturgeon spree. Outdoor Life. March:44.

Carr, A. 1978. An ecological study of the sturgeon in the Apalachicola and Suwannee Rivers. Unpublished manuscript. 6 pp.

Carr, A. 1983. All the way down upon the Suwannee River. Audubon Magazine. p. 80-101.

Chapman, F.A., S.F. O'Keefe, and D.E. Campton. 1993. Establishment of parameters critical for the culture and commercialization of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi*. Fisheries and Aquatic Sciences Dept., Food Science and Human Nutrition Dept., University of Florida, Gainesville, FL. Project Final Report. NOAA No. NA27FD0066-01. National Marine Fisheries Service. St. Petersburg, FL.

Clugston, J.P., A.M. Foster, and S.H. Carr. 1995. Gulf sturgeon, *Acipenser oxyrinchus desotoi*, in the Suwannee River, Florida, USA. Proc. of International Symposium on Sturgeons. Moscow, Russia. Editors: A.D. Gershanovich and T.I.J. Smith. Sept. 6-11, 1993. 370 pp.

Davis, J.T., B.J. Fontenot, C.E. Hoenke, A.M. Williams, and J.S. Hughes. 1970. Ecological factors affecting anadromous fishes of Lake Pontchartrain and its tributaries. Louisiana Wildlife and Fisheries Commission, Baton Rouge, Louisiana. Fisheries Bulletin No. 9. 63 pp.

Dean, B. 1893. Recent experiments in sturgeon hatching on the Delaware. Trans. N.Y. Acad. Sci. 13:69-74.

Doroshov, S.I., W.H. Clark, Jr., P.B. Lutes, R.L. Swallow, K.E. Beer, A.B. McGuire, and M.D. Cochran. 1983. Artificial propagation of the white sturgeon, *Acipenser transmontanus* (Richardson). Aquaculture 32:93-104.

Eisler, R. and J. Jacknow. 1985. Toxaphene hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.4). 26 pp.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service. Biological Report 85/1.11.

Esher, R.J. and D.K. Bradshaw. 1988. An ecological survey of the Stennis Space Center area potentially impacted by an advanced solid rocket motor manufacturing and test facility, Mississippi State University Research Center, NASA, John C. Stennis Space Center. 49 pp.

Florida Game and Fresh Water Fish Commission. 1964. State of Florida completion report of lake and stream survey. Federal Aid Project F-6-R. July 1, 1954 - June 30, 1964.

Florida Outdoors. 1959. The sturgeon: lively antique. p. 12-15.

Forbes, S.A and R.E. Richardson. 1908. The fishes of Illinois. Illinois State Laboratory of Natural History. Urbana, Illinois. 357 pp.

Foster, A.M. 1993. Movement of Gulf sturgeon, *Acipenser oxyrinchus desotoi* in the Suwannee River, Florida. Master Thesis, University of Florida, Gainesville, FL. 131 pp.

Foster, A.M., L.A. Patrick, and J.M. Barkuloo. 1988. Striped bass and sturgeon egg and larva studies on the Apalachicola River. 1987 Progress Report. 12 pp. and Appendices. U.S. Fish and Wildlife Service. Panama City, Florida.

Fox, G.A. 1992. Epidemiological and pathobiological evidence of contaminant-induced alterations in sexual development in free-living wildlife. In: chemically-induced alterations in sexual and functional development: the wildlife/human connection. Princeton Scientific Publishing Co., Inc.

Futch, C.R. 1984. Florida sturgeon management - A prospectus. Florida Department of Natural Resources, Division of Marine Resources. 11 pp.

Gilbert, C.R. (ed.) 1992. Atlantic sturgeon. In Rare and endangered biota of Florida, Volume II: Fishes. University Presses of Florida, Gainesville. pp. 5-8.

Ginsburg, I. 1952. Eight new fishes from the Gulf Coast of the United States, with two new genera and notes on geographic distribution. Journal of the Washington Academy of Sciences. 42(3): 84-101.

Hall, R.J. and N.C. Coon. 1988. Interpreting residues of petroleum hydrocarbons in wildlife tissues. U.S. Fish and Wildlife Service. Biological Report 88(15).

Harshbarger, J.C. 1975. Activities report registry of tumors in lower animals: 1974 supplement. National Museum of Natural History, Smithsonian Institution, Washington DC.

Hastings, R.W. 1983. A study of the Shortnose sturgeon *Acipenser brevirostrum*, population in the upper tidal Delaware River: assessment of impacts of maintenance dredging. Final Report. 81 5670 USACE Shortnose Sturgeon - Delaware River (Hastings) DACW 61-81-C-0138. Center for Coastal and Environmental Studies for U.S. Army Corps of Engineers, Philadelphia District. 117 pp.

Hayes, L.R., M.L. Maslia, and W.C. Meeks. 1983. Hydrology and model evaluation of the principal artesian aquifer, Dougherty Plain, southwest Georgia. Georgia Department of Natural Resources. Environmental Protection Division. Bulletin 97. 93 pp.

Huff, J.A. 1975. Life History of the Gulf of Mexico Sturgeon, *Acipenser oxyrhynchus desotoi* in Suwannee River, Florida. Mar. Res. Publ. No. 16. 32 pp.

Jenkins, L.G. unpublished manuscript. Age and growth of Gulf sturgeon in the Apalachicola River, Florida. U.S. Fish and Wildlife Service. Panama City, Florida.

Johnson, W.L. and M.T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. U.S. Fish and Wildlife Service. Resource Publication No. 137, Washington, D.C.

Lee, S.D., C.R. Gilbert, C.H. Hocutt, R.E. Jenkins, D.E. McAllister, and J.R. Stauffer. 1980. Atlas of North American freshwater fishes. Publication #1980-12. North Carolina Biological Survey. 867 pp.

Leitman, S., A. Dzurik, S. Ovenden, and D. Wilber. 1993. Effects of irrigation withdrawals in the Dougherty Plain on base-flow in the Apalachicola River. p. 73-77. Proceedings of the 1993 Georgia Water Resources Conference, April 20 and 21, 1993. The University of Georgia. Athens, Georgia.

Livingston, R.J. 1984. The ecology of the Apalachicola Bay system: an estuarine profile. U.S. Fish and Wildlife Service. FWS/OBS/82/5. 149 pp.

Louisiana Department of Conservation. 1965. Fishes and fishing in Louisiana. Bulletin No. 23. Reprinted by Claitor's Book Store, Baton Rouge, Louisiana.

Louisiana Department of Wildlife and Fisheries. 1979. Letter of additional comments -- Mermentau Basin Louisiana Project to Col. Thomas Sands, New Orleans District, U.S. Army Corps of Engineers. October 4. 3 pp.

Marchant, S.R. and M.K. Shutters. 1994. unpublished manuscript. Artificial substrates collect the first Gulf sturgeon egg. National Biological Survey, Southeastern Biological Science Center, Gainesville, FL. 8 pp.

Mason, W.T. Jr., J.P. Clugston, and A.M. Foster. 1992. Growth of laboratory-held Gulf of Mexico sturgeon *Acipenser oxyrhynchus desotoi*. Prog. Fish Culturist 54:59-61.

Mason, W.T. Jr. and J.P. Clugston. 1993. Foods of the Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society 122:378-385.

Mayer, F.L., and P.M. Mehrle. 1977. Toxicological aspects of toxaphene in fish: A summary. Trans. N. Am. Wildlife Naturals Resource Conference, 42:365-373.

McDowall, R.M. 1988. Diadromy in fishes migrations between freshwater and marine environments. Truder Press and Croom Helm. 308 pp.

Miranda, L.E. and D.C. Jackson. 1987. A Status Survey of Atlantic Sturgeon in the Pascagoula and Pearl River Systems of Mississippi, Mississippi Cooperative Fish and Wildlife Unit and Mississippi Department of Wildlife and Fisheries. unpublished report. 27 pp. Mississippi State Game and Fish Commission Public Notice No. 1790.

Moser, M.L. and S.W. Ross. 1993. Distribution and movements of shortnose sturgeon *Acipenser brevirostrum* and other anadromous fishes of the lower Cape Fear River, North Carolina. Final Report to the U.S. Army Corps of Engineers, Wilmington District. 153 pp. May.

Murawski, S.A. and A.L. Pacheco. 1977. Biological and fisheries data on Atlantic sturgeon, *Acipenser oxyrhynchus* (Mitchill). National Marine Fisheries Service - Technical Series Report No. 10. 69 pp.

Murphy, M.J. and J. Skaines. 1994. Habitat and movement of the Gulf sturgeon (*Acipenser oxyrhynchus desotoi*) in the Pascagoula River, Mississippi. Museum Tech. Rep. No. 29. 20 pp. Mississippi Museum of Natural Science. Mississippi Department of Wildlife, Fisheries and Parks. Jackson, MS

National Marine Fisheries Service. 1987. Fish Landing reports. Miami, Florida. Florida Landings for 1981, 1982, 1983, and 1984. Computer printout. Southeast Fisheries Center.

Odenkirk, J., F.M. Parauka and P.A. Moon. 1988. Unpublished manuscript. Radio tracking and spawning observations of the Gulf of Mexico sturgeon. U.S. Fish and Wildlife Service. Panama City, Florida.

Odenkirk, J.S. 1989. Movements of Gulf of Mexico sturgeon in the Apalachicola River, Florida. Proc. Annu. Conf. Southeastern Assoc. Fish and Wildlife Agencies 43:230-238.

Ong, T., I. Wirgin and J.R. Waldman. unpublished manuscript. Subspecies status of *Acipenser oxyrinchus* as assessed by mitochondrial DNA sequencing analysis. New York University Medical Center and Hudson River Foundation. 31 pp.

Parauka, F.M., W.J. Troxel, F.A. Chapman, and L.G. McBay. 1991. Hormone-induced ovulation and artificial spawning of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi*. Prog. Fish-Culturist 53(2):113-117.

Potak, K. J. Hightower, and K. Pollock. 1995. Abundance and mortality of the threatened Gulf sturgeon. 1995 Annual Report. North Carolina Cooperative Fish and Wildlife Research Unit, Dept. of Zoology, North Carolina State University, Raleigh, North Carolina. 11 pp.

Rafinesque, C.S. 1820. *Ichthyologia Ohiensis*, or natural history of the fishes inhabiting the river Ohio and its tributary streams, preceded by a physical description of the Ohio and its branches. Lexington, Kentucky. 90 pp.

Reynolds, C.R. 1993. Gulf sturgeon sightings, historic and recent - a summary of public responses. U.S. Fish and Wildlife Service. Panama City, Florida. 40 pp.

Rogillio, H.E. 1993. Status survey of Gulf sturgeon in Louisiana. Louisiana Department of Wildlife and Fisheries. Project E1-1. 6 pp.

Roithmayr, C.M. 1965. Industrial bottomfish fishery of the northern Gulf of Mexico, 1959-63. U.S. Dept. of the Interior. Fish and Wildlife Service. Bur. of Commercial Fisheries. Special Sci. Rep.- Fisheries No. 518. Washington, D.C. 23 pp.

Schwartz, F.J. 1972. World literature to fish hybrids with an analysis by family, species, and hybrid. Publ. Gulf Coast Research Laboratory Mus. 3:1-328.

Schwartz, F.J. 1981. World literature to fish hybrids with an analysis by family, species, and hybrid: Supplement 1. NOAA Tech. Rep. NMFS SSF 750: 1-507.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Fish. Res. Board Can. Bull 184. 966 pp.

Sentry News. 1986. Quite a catch. October. Slidell, Louisiana.

Smith, T.I.J., E.K. Dingley, and D.E. Marchette. 1980. Induced spawning culture of the Atlantic sturgeon, *Acipenser oxyrhynchus* (Mitchell). Prog. Fish Culture 42: 147-151.

Smith, T.I.J. and J.P. Clugston. unpublished manuscript. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. Presented: International Conference on Sturgeon Biodiversity and Conservation July 28-30, 1994, New York. 32 pp.

Stabile, J., J.R. Waldman, J. Hart, and I. Wirgin. unpublished manuscript. Differentiation of Gulf sturgeon *Acipenser oxyrinchus desotoi* populations based on sequence and RFLP analyses of mitochondrial DNA. New York University Medical Center and Hudson River Foundation.

Swift, C., R.W. Yerger, and P.R. Parrish. 1977. Distribution and natural history of the fresh and brackish water fishes of the Ochlockonee River, Florida and Georgia. Bull. Tall Timbers Res. Sta. No. 20. October. pp. 18-19. Tallahassee, FL.

Sun Herald. 1992. Man lands in hot water after catching odd fish. November 30. Biloxi, Mississippi.

Tallahassee Democrat. 1958. Photograph of fisherman (Bill Humphrey) and 33.5 kg (74 lb) sturgeon, caught Jim Woodruff Dam in the Apalachicola River. May 15. Tallahassee, FL.

Tallahassee Democrat. 1963. Photograph of fisherman (Jerry Schuler) and 73 kg (161 lb) sturgeon, measuring 2.3 m (7.5 ft) caught near Rock Bluff Dam in the Apalachicola River. Numerous sturgeon had been caught in the deeper river holes lately. September 3. Tallahassee, FL.

Tallahassee Democrat. 1969. Photograph of fisherman (Donald Tucker) and 34.0 kg (75 lb) sturgeon, caught Jim Woodruff Dam in the Apalachicola River. May 6. Tallahassee, FL.

Torak, L.J., G.S. Davis, G.A. Strain, and J.G. Herdon. 1993. Geohydrology and evaluation of water-resource potential of the upper Florida aquifer in the Albany area, Southwestern Georgia. U.S. Geological Survey. Water-Supply Paper 2391. 59 pp.

U.S. Army Corps of Engineers. 1978. Appendix III. A study of the diadromous fishery resources of the Apalachicola-Chattahoochee-Flint River System, Alabama, Georgia and Florida. In Coordination report on navigational improvements for Apalachicola River below Jim Woodruff Dam, Florida.

U.S. Army Corps of Engineers. Mobile District. 1986. Navigation maintenance plan for the Apalachicola-Chattahoochee-Flint waterway. Vol 1. September. 192 pp. with appendices.

U.S. Army Corps of Engineers. Mobile District. 1993. Draft Jim Woodruff lock and dam, Apalachicola River, Florida and Georgia. Powerhouse major rehabilitation evaluation report. 61 pp. and appendices.

U.S. Army Corps of Engineers. Mobile District. 1994. Letter to U.S. Fish and Wildlife Service on Gulf sturgeon draft public/agency recovery plan regarding velocity measurements taken in the Apalachicola River below Jim Woodruff Lock and Dam in November 1991 and October 1992. April 4, 1994. 2 pp. with attachments.

U.S. Commission on Fish and Fisheries. 1902. Report of the Commissioner (Part XXVII) for the year ending June 30, 1901. Government Printing Office. pp. 39-40 and 155.

U.S. Fish and Wildlife Service. 1988. Fisheries activities, Fish and Wildlife Service Field Office, Panama City, Florida. Unpublished Annual Report for 1987. 20 pp.

U.S. Fish and Wildlife Service. 1989. Fisheries activities, Fish and Wildlife Service Field Office, Panama City, Florida. Unpublished Annual Report for 1988. 34 pp.

U.S. Fish and Wildlife Service. 1990. Fisheries activities, Fish and Wildlife Service Field Office, Panama City, Florida. Unpublished Annual Report for 1989. 19 pp. with appendices.

U.S. Fish and Wildlife Service. 1991a. Endangered and threatened wildlife and plants; determination of threatened status for the Gulf sturgeon. Federal Register 56(189): 49653-49658.

U.S. Fish and Wildlife Service. 1991b. Fisheries activities, Fish and Wildlife Service Field Office, Panama City, Florida. Unpublished Annual Report for 1990. 35 pp. with appendices.

U.S. Fish and Wildlife Service. 1992. Fisheries activities, Fish and Wildlife Service Field Office, Panama City, Florida. Unpublished Annual Report for 1991. 34 pp.

U.S. Fish and Wildlife Service. National paddlefish and sturgeon steering committee. 1993. Framework for the management and conservation of paddlefish and sturgeon species in the United States. Washington, D.C. 41 pp.

Van Den Avyle, M.J. 1984. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (south Atlantic)--Atlantic sturgeon. U.S. Fish Wildl. Serv. Biol. Rep. 81(11.25). U.S. Army Corps of Engineers, TR EL-82-4. 17pp.

Veshchev, P.V. 1982. Effect of dredging operations in the Volga River on migration of sturgeon larvae. Journal of Ichthyology 22:108-122.

Vittor, B.A. 1972. The ecological consequences of channel dredging in D'Olive Bay, Alabama. University of Alabama Marine Science Program. Dauphin Island, AL. Final Report. DACW01-72-C-0085. U.S. Army Corps of Engineers, Mobile District, AL.

Vladykov, V.D. 1955. A comparison of Atlantic sea sturgeon with a new subspecies from the Gulf of Mexico (*Acipenser oxyrhynchus desotoi*). Journal Fish Research Board Canada 12(5):754-761.

Vladykov, V.D. and J.R. Greeley. 1963. Order Acipenseroidei. In fishes of the western North Atlantic. Memoir Sears Foundation for Marine Research 1(3): 24-58.

White, D.H., C.A. Mitchell, H.D. Kennedy, A.J. Krynitsky, and M.A. Ribick. 1983. Elevated DDE and toxaphene residues in fishes and birds reflect local contamination in the lower Rio Grande Valley, Texas. The Southwestern Naturalist 28(3):325-333.

Williams, J.D. and G.H. Clemmer. 1991. *Scaphirhynchus suttkusi* a new sturgeon (Pisces: Acipenseridae) from the Mobile Basin of Alabama and Mississippi. Bulletin Alabama Museum of Natural History 10:17-31.

Womack, M. 1991. Newspaper article: Newspaper excerpts recount "mother of battle." Out of the Past series. The News Herald, Panama City, Florida. April 7.

Wooley, C.M. 1985. Evaluation of morphometric characters used in taxonomic separation of Gulf of Mexico sturgeon, *Acipenser oxyrhynchus desotoi*. pp. 97-103, in F. Binkowski and S.I. Doroshov, editors. North American sturgeon. Developments in environmental biology of fishes. Volume 6. Dr. W. Junk Publishers. The Netherlands.

Wooley, C.M. and E.J. Crateau. 1985. Movement, microhabitat, exploitation and management of Gulf of Mexico sturgeon, Apalachicola River, Florida. N. Amer. J. Fish. Manage. pp. 590-605.

Wooley, C.M., P.A. Moon, and E.J. Crateau. 1982. A larval Gulf of Mexico sturgeon (*Acipenser oxyrhynchus desotoi*) from the Apalachicola River, Florida. Northeast Gulf Science 5(2):57-58.

Barkuloo, J.M. (retired). 1993. U.S. Fish and Wildlife Service, Panama City, Florida.

Bass, G. 1993. Florida Game and Fresh Water Fish Commission, Holt, Florida.

Bowker, R. 1994. U.S. Fish and Wildlife Service, Ecological Services, Jackson, Mississippi.

Bradshaw, D. 1989. Letter to J.M. Barkuloo, FWS, commenting on draft *Acipenser oxyrinchus desotoi* status report. Mississippi State University Research Center. John Stennis Space Center, MS.

Brim, M. 1993. U.S. Fish and Wildlife Service, Ecological Services, Panama City, Florida.

Butler, R. 1993. U.S. Fish and Wildlife Service, Ecological Services, Jacksonville, Florida.

Carr, A. 1980. Telephone conversation with Jim Barkuloo, FWS, concerning his work on Gulf sturgeon as summarized by Jim Barkuloo in a June 26 memo.

Carr, S. and P. Rago. 1993. Caribbean Conservation Corporation, Gainesville, Florida and U.S. Fish and Wildlife Service, Leetown, Virginia (Rago is now with NMFS).

Carr, S. 1988. Letter to J.M. Barkuloo, FWS, commenting on draft *Acipenser oxyrinchus desotoi* status report. Phipps Florida Foundation sturgeon project.

Carr, S. 1993. Caribbean Conservation Corporation. Gainesville, FL.

Carr, S. and F. Tatman. 1993. Carribean Conservation Corporation, Gainesville, Florida.

Clugston, J.P. 1993. National Biological Service, Southeastern Biological Science Center, Gainesville, Florida.

Christmas, J.Y. 1994. Report of a Gulf sturgeon catch. Gulf Coast Research Laboratory.

Douglas, P. 1993. U.S. Fish and Wildlife Service, Ecological Services, Daphne, Alabama.

Duffy, J. 1993. Alabama Department of Natural Resources, Marine Resources Division, Gulf Shores, Alabama.

Fichera J. 1989. Commercial fisherman, Apalachicola, Florida.

Florida Game and Fresh Water Fish Commission. 1993. Holt, Florida.

Jones, R. 1990. Letter to David Wesley, U.S. Fish and Wildlife Service concerning the proposal to list the Gulf sturgeon as a threatened species. Mississippi Department of Wildlife, Fisheries and Parks. Research Section. June 5.

Jordan, N. 1993. Gulf Coast Research Laboratory (GCRL). Ocean Springs, MS.

Knight, C. 1994. Memo to Bob Bowker, U.S. Fish and Wildlife Service, Jackson, Mississippi, concerning catch of a Gulf sturgeon in the Pearl River above the sill at Bogalusa, Louisiana, in 1982. April 11.

McDearman, W. 1994. Letter to Field Supervisor, U.S. Fish and Wildlife Service, Panama City, Florida, concerning occurrence of Gulf sturgeon on the upper Pearl River and the passage of Gulf sturgeon above the sill at Bogalusa, Louisiana. Mississippi Department of Wildlife, Fisheries and Parks. Museum of Natural Science. April 21.

Mettee, M. 1993. Alabama Geological Survey, Tuscaloosa, Alabama.

Mowbray, D. 1993. Alabama Department of Conservation and Natural Resources, Hartford, Alabama.

National Biological Service. (formerly U.S. Fish and Wildlife Service). 1993. Southeastern Biological Science Center, Gainesville, Florida.

Parauka, F.M. 1992. U.S. Fish and Wildlife Service, Office of Fisheries Assistance, Panama City, Florida.

Petzold, F. 1993. Mississippi State University.

Pierson, M. 1993. Alabama Power Company. Calera, Alabama.

Poitevint, H. 1994. Memo to Lorna Patrick, U.S. Fish and Wildlife Service, Panama City, Florida, concerning Gulf sturgeon migration in the Pearl River above Pools Bluff and Bogue Chitto sills. April 7. U.S. Fish and Wildlife Service. Southeast Louisiana Refuges, Slidell, Louisiana.

Robins, D. 1993. University of Miami, Florida.

Rogillio, H. 1993. Louisiana Department of Wildlife and Fisheries, Fisheries Section, Slidell, Louisiana.

Roussos, J. 1993. Roussos Restaurant. Mobile, Alabama.

Ruiz-Carus, R. Florida Department of Environmental Protection (formerly Florida Department of Natural Resources). Marine Research Institute. St. Petersburg, Florida.

Shepard, Steve. 1992. Response to U.S. Fish and Wildlife Service survey on Gulf sturgeon sightings. Gauiter, Mississippi.

Stewart, J. 1987. Memo to James Barkuloo, U.S. Fish and Wildlife Service, Panama City, Florida concerning catch of sturgeon in Alabama, Mississippi, and Louisiana waters. U.S. Fish and Wildlife Service. Jackson, Mississippi. January 8.

Tatman, F. 1993. Caribbean Conservation Corporation and commercial fisherman. Gainesville, Florida.

Torak, L. 1994. U.S. Geological Survey, Water Resources Division. Doraville, Georgia.

Tucker, W. 1993. Alabama Department of Conservation and Natural Resources. Spanish Fort, Alabama.

U.S. Army Corps of Engineers. 1994. Analysis of duration of water depth over the Pool Bluff and Bogue Chitto sills, Pearl River basin. Requested by the U.S. Fish and Wildlife Service, Panama City, Florida. July 21.

U.S. Fish and Wildlife Service. 1993. Office of Fisheries Assistance. Panama City Field Office. Panama City, Florida.

Winger, P. National Biological Service. (formerly U.S. Fish and Wildlife Service) Field Research Station, Athens, Georgia.

III. IMPLEMENTATION SCHEDULE

The Implementation Schedule indicates task priorities, task numbers, task descriptions, duration of tasks, potential or participating parties, and lastly, estimated costs (Table 3). These tasks, when accomplished, will bring about the recovery objectives for the Gulf sturgeon as discussed in Part II of this plan.

Parties with authority, responsibility, or expressed interest to implement a specific recovery task are identified in the Implementation Schedule. When more than one party has been identified, the proposed lead party is indicated by an asterisk (*). The listing of a party in the Implementation Schedule does not imply a requirement or that prior approval has been given by that party to participate or expend funds. However, parties willing to participate will benefit by being able to show in their own budget submittals that their funding request is for a recovery task which has been identified in an approved recovery plan and is therefore part of the overall coordinated effort to recover the Gulf sturgeon. Also, Section 7(a)(1) of the ESA directs all federal agencies to utilize their authorities in furtherance of the purposes of the ESA by carrying out programs for the conservation of threatened and endangered species.

Following are definitions to column headings and keys to abbreviations and acronyms used in the Implementation Schedule:

<u>Task Number & Task</u>: Recovery tasks as numbered in the recovery outline. Refer to the Narrative for task descriptions.

<u>Priority Number</u>: All priority 1 tasks are listed first, followed by priority 2 and priority 3 tasks.

> Priority 1 - All actions that must be taken to prevent extinction or to prevent the subspecies from declining irreversibly in the foreseeable future.
>
> Priority 2 - All actions that must be taken to prevent a significant decline in subspecies population/habitat quality, or some other significant negative impact short of extinction.
>
> Priority 3 - All other actions necessary to provide for full recovery (or reclassification) of the species.

<u>Task Duration</u>: Years to complete the corresponding task. Study designs can incorporate more than one task, which can reduce the time needed for task completion.

> Underway - Task already being implemented.
>
> Continuing - Task necessary until recovery.

Responsible or Participating Party: Federal or state government agencies or universities (party) with the responsibility and/or capability to fund or carry out the corresponding recovery task.

FWS Region - FWS Regions (only states in the Gulf sturgeons's range are listed)
- 2 - Albuquerque (Texas)
- 4 - Atlanta (LA, MS, AL, FL)

FWS Program - Division or program of the FWS
- FF- Fisheries
- FRO- Fisheries Resources Office
- ES- Ecological Services
- LE- Law Enforcement
- WNFH- Welaka National Fish Hatchery
- WSRFC- Warm Springs Regional Fisheries Center
- GCFCO- Gulf Coast Fisheries Coordination Office

Other Federal Agencies
- COE - U.S. Army Corps of Engineers
- EPA - U.S. Environmental Protection Agency
- MMS - Minerals Management Service
- NMFS - National Marine Fisheries Service
- FERC - Federal Energy Regulatory Coummission
- NBS - National Biological Service/Southestern Biological Science Center Gainesville, FL
- NRCS - Natural Resources Conservation Service

State Agencies
- GSRMA - Gulf States Resource Management Agencies
  - Louisiana Department of Wildlife and Fisheries
  - Mississippi Department of Wildlife, Fisheries, and Parks
  - Alabama Department of Conservation and Natural Resources
  - Florida Department of Environmental Protection
  - Texas Parks and Wildlife Department
- CES - Cooperative Extension Service (all GSRMA)

Other Parties
- GSMFC - Gulf States Marine Fisheries Commission
- CCC - Caribbean Conservation Corporation
- UF - University of Florida

Cost Estimates: Estimated fiscal year cost, in thousands of dollars, to complete the corresponding task. The costs associated with a task or party represent the estimated dollar amount to complete the task and are not necessarily the fiscal responsibility of the associated party.

Study designs can incorporate more than one task, which when combined can reduce the cost from when tasks are conducted separately. Cost for implementing "continuing" recovery tasks are in excess of what is displayed for the five years in the schedule.

Comments: Additional information if appropriate.

TABLE 3. IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

Priority	TASK #	TASK DESCRIPTION	TASK DURATION (YEARS)	RESPONSIBLE PARTY: FWS			ESTIMATED FISCAL YEAR COSTS ($000): FY 1		FY 2		FY 3		FY 4		FY 5		COMMENTS
				Region	Program	OTHER	FWS	Other	FWS	Other	FWS	Other	FWS	Other	FWS	Other	
1	1.3.1	Develop and implement standardized population sampling and monitoring techniques	underway	4	FF* FRO-PC	NBS* GSRMA COE	1 6	30 20 2	1 20	30 20 2	7 40	30 32 5	1 40	30 32 5	1 40	30 32 5	Tasks 1.1.1, 1.3.1, 2.5.1, and 1.5.1 can be conducted concurrently
1	2.5.3	Develop and implement a regulatory framework to eliminate acidental and intentional introductions of non-indigenous stock or other sturgeon species	1	4	FF FRO-PC* ES-PC GCFCO	NBS* GSRMA GSMFC UF			5 8 2 2	2 4 1 1							Some of this effort will be dependent on the outcome of 2.5.1
1	2.1.2	Reduce or eliminate incidental mortality	underway continuing	4	FRO-PC* ES	GSMFC* GSRMA NMFS	15	15 20 75	15	15 20 75	15	15 20 75		75		25	Majority of funding for fish excluder devices & sampling protocols
1	2.4.5	Restore the benefits of natural riverine habitats	underway continuing	4	ES FRO-PC GCFCO	NBS COE GSRMA	2 2 2	2 10 8	10 2 2	2 20 12	10 2 2	2 20 12	20 5 3	3			Wk funded under existing programs. Actual restoration costs undetermined.
1	2.3.1	Utilize existing authorities to protect habitat and where inadequate, recommend new incentives, laws, and regulations	underway continuing	4	ES* GCFCO	EPA* COE GSRMA GSMFC	5 3	5 5 8 3	5 3	5 5 8 3	5 3	5 5 8 3	5 3	5 5 8 3			Section 7 consultation conducted with existing program funds
2	2.1.1	Increase effectiveness and enforcement of state and federal take prohibitions	continuing	4	LE FF* ES*	NMFS* GSRMA*	75	75 180	75	75 180	75	75 180	75	75 180	75	75 180	Sec 7 consultation will be conducted under existing programs. Add. monitoring or law personnel may be necessary
2	1.1.1	Conduct and refine field investigations to locate important spawning, feeding, and developmental habitats	underway continuing	4	FF FRO-PC* GCFCO	NBS* GSRMA COE CCC UF	1 5 1	20 60 5 10 1	1 58 1	20 60 5 10 1	1 70 2	20 80 5 10 2	1 70 2	20 80 5 12 2	1 70 5	20 80 5 12 5	Tasks 1.1.1, 1.3.1, 2.5.1, and 1.5.1 can be conducted concurrently

TABLE 3. (continued). IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

PRIORITY	TASK #	TASK DESCRIPTION	TASK DURATION (YEARS)	RESPONSIBLE PARTY			ESTIMATED FISCAL YEAR COSTS ($000)										COMMENTS
				FWS		OTHER	FY 1		FY 2		FY 3		FY 4		FY 5		
				Region	Program		FWS	Other	FWS	Other	FWS	Other	FWS	Other	FWS	Other	
2	1.1.2	Characterize riverine, estuarine, and neritic areas that provide essential habitat	underway continuing	4	FRO-PC*	NBS* CCC GSRMA COE	5	15 2 28 5	20	15 2 28 5	70	15 3 40 5	70	15 3 40 5	10	15 3 40 5	Tasks 1.1.1 and 1.1.2 can be conducted concurrently
2	1.2	Conduct life history studies on the biological and ecological requirements of little known or inadequately sampled life stages	underway continuing	4	FRO-PC*	NBS* CCC GSRMA	5	25 2 28	20	25 2 28	20	25 3 40	40	25 3 40	40	25 3 40	Tasks 1.1.1 and 1.1.2, and 1.2 can be conducted concurrently
2	2.2.1	Identify potentially harmful chemical contaminants and water quality and quantity changes associated with surface water restrictions	3	4	ES-PC*	EPA GSRMA	25	10 40	15	10 100	75						Cost and time to complete year 2 efforts will be dependent on information collection in year 1.
2	2.2.2	Identify and eliminate potentially harmful point and non-point sources of chemical contaminants	4	4	ES-PC	EPA* GSRMA NRCS			20	10 28	25	15 40	25		25		
2	2.4.6	Seek optimum consistency between the purposes of federal and state authorized reservoirs, flood control, navigation, and hydropower projects and federal and state mandated restorations of fish populations	continuing	4	ES GCFCO	GSMFC* FERC COE NMFS				10		5		5		5	Most agency related wk., funded under existing programs

TABLE 3. (continued). IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

PRIORITY	TASK #	TASK DESCRIPTION	TASK DURATION (YEARS)	RESPONSIBLE PARTY			ESTIMATED FISCAL YEAR COSTS ($000)										COMMENTS
				FWS		OTHER	FY 1		FY 2		FY 3		FY 4		FY 5		
				Region	Program		FWS	Other	FWS	Other	FWS	Other	FWS	Other	FWS	Other	
2	2.4.1	Identify dam and lock sites that offer the greatest feasibility for successful restoration of and to essential habitats	1	4	ES-PC FRO-PC	GSMFC* COE GSRMA			5 2	15 10 20							
2	2.4.4	Identify potential modifications to specific navigation projects to minimize impacts which alter riverine habitats or modify thermal or substrate characteristics of those habitats.	underway continuing	4	ES FRO-PC GCFCO	FERC* COE* NMFS GSRMA GSMFC	5 5 5	10 10 2 8 5	5 5 5	10 10 2 8 5	2 2 2	5 5 2 4 2					Some funding under existing programs. Proj. mod. costs undetermined and may require Congress. author. & non-federal sponsor
2	4.3	Implement projects or actions which will achieve recovery plan objectives	underway continuing	4	FF FRO-PC	GSRMA* NGOs											Individual project funding ID elsewhere in schedule
2	4.2	Seek funding for Gulf sturgeon recovery activities	underway continuing	4	ES* GCFCO	NBS GSMFC GSRMA											Funded under existing programs
2	2.2.4	Identify and eliminate known and potential impacts to water quantity and quality associated with existing and proposed developments, agricultural uses, and water diversions in management units	continuing	4	ES	NBS EPA* GSRMA NRCS	2	2 2 8	10	5 20 8	75	5 20 8	75	5 20	75	20	Amount of effort will be determined by outcome of task 2.2.1
2	2.2.5	Assess the relationship between groundwater pumping and reduction of groundwater flows into management units, and quantify loss of riverine habitat related to reduced groundwater in-flows	2	4	ES	USGS* GADNR						252		125			Mostly funded under the Tri-state Comp Study- AL,GA,FL

TABLE 3. (continued). IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

PRIORITY	TASK #	TASK DESCRIPTION	TASK DURATION (YEARS)	RESPONSIBLE PARTY: FWS Region	RESPONSIBLE PARTY: FWS Program	RESPONSIBLE PARTY: OTHER	FY 1 FWS	FY 1 Other	FY 2 FWS	FY 2 Other	FY 3 FWS	FY 3 Other	FY 4 FWS	FY 4 Other	FY 5 FWS	FY 5 Other	COMMENTS
3	2.5.1	Evaluate the need to stock hatchery-produced Gulf sturgeon considering habitat suitablity and current population status	underway	4	FF FRO-PC ES-PC GCFCO	NBS GSRMA	1 1 1 1	5 8	1 3 1 1	10 8	1 5 2 1	10 4	1 10 2 1	10 4	1 10 2 1	10 13	Tasks 1.1.1, 1.3.1, 2.5.1, and 1.5.1 can be conducted concurrently
3	1.5.1	Conduct a Gulfwide genetic assessment to determine geographically distinct management units	underway	4	FF* FRO-PC GCFCO	NBS GSRMA NGOs	15 8 2	1 3 1	15 48 1	1 100 1							Majority of samples and analyses completed 1995. Will continue to completion.
3	2.2.3	Assess selected contaminant levels in Gulf sturgeon from management units	underway continuing	4	FF* ES*	EPA* GSRMA	15		30	10 20	30	10 20	10	5 20			Study on adult fish across FL panhandle completed 1994. Study on juvenile fish, Suwannee River completed 1995.
3	1.3.2	Develop population models	underway continuing	4	FF FRO-PC	NBS NMFS GSRMA NGOs	5 15	15 2 8 2	5 5	15 2 8 2	20						
3	4.1	Designate and fund a Gulf sturgeon recovery lead office	continuing	4	ES* FF		7 3		7 3		7 3		7 3		7 3	7 3	Majority of funding provided under other recovery actions
3	1.4.1	Continue culture of Gulf sturgeon	underway	4	WNFH WSRFC* FRO-PC	NBS LDWF ADNCR UF	3 2 1	2 3 3 5	23 25 10	2 3 3 5	23 25 10	2 5 5 5	23 25 10	2 5 5 10	23 25 10	2 5 5 10	

TABLE 3. (continued). IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

Priority	Task #	Task Description	Task Duration (Years)	Responsible Party: FWS Region	Responsible Party: FWS Program	Responsible Party: Other	FY 1 FWS	FY 1 Other	FY 2 FWS	FY 2 Other	FY 3 FWS	FY 3 Other	FY 4 FWS	FY 4 Other	FY 5 FWS	FY 5 Other	Comments
3	2.2.6	Conduct studies to determine the effects of known chemical contaminants in water from management units on Gulf sturgeon or a surrogate species	4	4	ES-PC* WNFH WSRFC	EPA NBS			75 5	10 5	75 5	10 5	75		75		WNFH & NBS may provide specimens for the studies
3	2.4.3	Operate and/or modify dams to restore the benefits of historical flow patterns and processes of sedimentation	underway continuing	4	ES FRO-PC GCFCO	FERC* COE* NMFS GSMFC											Some funding under existing programs. Project mod. costs underterm. May require Congress. authority & non-federal sponsor.
3	2.3.2	Identify, protect, and/or acquire appropriate land or aquatic habitats on an ecosystem approach	underway continuing	4	FF FRO-PC ES-PC* GCFCO RW	NBS NMFS GSRMA NGOs											ID conducted with other studies. Land acquis. & water rights costs undeterminable.
3	2.4.2	Evaluate, design, and provide means for Gulf sturgeon to bypass migration restrictions to essential habitats	continuing	4	ES FF	FERC* COE* NMFS				10 10		25 25		25 25		25 25	FWS & NMFS funded under exist. progs. Studies conducted or infrastructure funded by COE & FERC. May req. Congress. auth. & non-fed sponsor.
3	3.1	Coordinate research and recovery actions	continuing	4	ES* FF GCFCO	NBS GSMFC*	5	5	10 5 5	2 15	5	5	10 5 5	2 15	5	5	Funding for biennial workshops
3	2.5.2	Develop policy and guidelines for hatchery and culture operations related to stocking	2	4	FF FRO-PC* ES-PC GCFCO	NBS* GSRMA GSMFC UF			5 5 2 2	2 4 1 1					5 10 5 5	2 4 2 15	Conducting this effort will be dependent on the outcome of 2.5.1
3	3.2	Develop an effective communication program or network to obtain and disseminate information on recovery actions and research results	continuing	4	ES*	GSMFC CES			5	5 2	5	5 2	5	5 2	5	5 2	Funding for producing and distributing quarterly newsletters

Estimated Fiscal Year Costs ($000)

TABLE 3. (continued). IMPLEMENTATION SCHEDULE FOR GULF STURGEON RECOVERY ACTIONS

GULF STURGEON RECOVERY IMPLEMENTATION SCHEDULE

PRIORITY	TASK #	TASK DESCRIPTION	TASK DURATION (YEARS)	RESPONSIBLE PARTY			ESTIMATED FISCAL YEAR COSTS ($000)										COMMENTS
				FWS		OTHER	FY 1		FY 2		FY 3		FY 4		FY 5		
				Region	Program		FWS	Other	FWS	Other	FWS	Other	FWS	Other	FWS	Other	
3	3.3	Develop a non-scientific constituency and public information program directed toward enhancing recovery actions	underway continuing	4	FF* ES* GCFCO CES	GSMFC* NMFS GSRMA			5 5 8	10 5	5 5 8	10 5	5 5 8	5	2 2 8	5	
3	1.5.2	Assess the potential to develop genetic markers to differentiate wild and hatchery-produced Gulf sturgeon	ongoing	4	FF* ES	NMFS UF			25 25	10 10	25 25	10 10					Funding this task dependent on task 1.4.3 decision
3	1.4.2	Identify physical, chemical and biological parameters necessary to maintain growth, health, and survival of fish reared under artificial conditions	underway continuing	4	WNFH WSRFC*	NBS UF LDWF ADNCR	5 5	10 5 3 3	5 20	10 5 3 3	10 20	10 8 5 5	10 20	10 8 5 5	10 20	10 10 5 5	Continuation of this effort dependent on the outcome of 2.6.1.
3	1.4.3	ID and test non-genetic internal and external markers or techniques to differentiate wild and hatchery-produced Gulf sturgeon	2	4	FF FRO-PC*	NBS CCC GSRMA			25 5	5 2 4	25 5	5 2 4					Funding this task dependent on task 1.4.3 decision
3	4.4	Develop and implement a program to monitor levels and habitat conditions of known populations in the management units as well as newly discovered, introduced, or expanding populations	continuing	4	ES* FRO-PC	NBS CCC GSRMA	1 5	5 5 20	5 5	5 5 20	1 5	5 5 20	5 5	5 5 20	1 5	5 5 20	
3	5.1	Assess overall success of the recovery program and recommend action	continuing	4	ES*		2		2		2		2		2		

APPENDIX A

FISHERY MANAGEMENT JURISDICTIONS, LAWS AND POLICIES AFFECTING THE GULF STURGEON

APPENDIX A

FISHERY MANAGEMENT JURISDICTIONS, LAWS AND POLICIES AFFECTING THE STOCKS:

Gulf sturgeon may utilize both fresh water and marine habitats at different times of the year. Excursions into the territorial waters (Exclusive Economic Zone) of the United States may occur. This factor in its biology, together with its range, subject the subspecies to the regulatory jurisdictions of the federal government as well as the States of Alabama, Louisiana, Mississippi and Florida. Numerous state and federal legislative and regulatory actions may affect the stocks. The following is a partial list of some of the more important agencies and regulations that affect the Gulf sturgeon and its habitat. State agencies should be consulted for specific and current state laws and regulations.

Federal Management Institutions. Although some recreational and subsistence harvests of Gulf sturgeon have occurred at times, the primary fishery for the sturgeon has been commercial. Because Gulf sturgeon fisheries have occurred primarily in state waters, federal agencies historically have not directly managed the stocks; though, the federal government has maintained commercial fishery landing records on the subspecies for about the past 100 years. Nonetheless, a variety of federal agencies, through their administration of laws, regulations and policies, may influence Gulf sturgeon stocks.

Regional Fishery Management Councils. With the passage of the Magnuson Fishery Conservation and Management Act (MFCMA), the federal government assumed responsibility for fishery management within the Exclusive Economic Zone (EEZ). The EEZ is contiguous to the territorial sea, with an inner boundary at the outer boundary of each coastal state. The outer boundary continues out 200 miles. Management of the EEZ is to be based on fishery management plans developed by regional fishery management councils. Each council prepares plans, with respect to each fishery requiring management, within its geographical area of authority and amends such plans as necessary. Plans are implemented as federal regulation through the Department of Commerce (DOC).

Among the guidelines, under which the councils must operate, are standards which state that, to the extent practicable, an individual stock of fish shall be managed as a unit throughout its range and that management shall, where practicable, promote efficiency, minimize costs and avoid unnecessary duplication (MFCMA Section 301a).

The Gulf of Mexico Fishery Management Council has not developed, nor is it considering, a management plan for the Gulf sturgeon. Furthermore, no significant fishery for the subspecies exists in the EEZ of the U.S. Gulf of Mexico.

Department of Commerce, National Oceanic and Atmospheric Administration (NOAA).

National Marine Fisheries Service. The Secretary of Commerce, acting through the NMFS, has the ultimate authority to approve or disapprove all fishery management plans prepared by regional fishery management councils. Where a council fails to develop a plan, or to correct an unacceptable plan, the Secretary may do so. The NMFS also collects data and statistics on fisheries and fishermen, performs research, and conducts management authorized by international treaties. The NMFS has the authority to enforce the Magnuson Act and the Lacey Act and is the federal trustee for living and nonliving natural resources in coastal and marine areas under United States jurisdiction pursuant to the Endangered Species Act, Section 107(f) of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or "Superfund"), Section 311(f)(5) of the Clean Water Act (CWA), Executive Order 12580 of January 23, 1987, and Subpart G of the National Oil and Hazardous Substances Pollution Contingency Plan.

The NMFS exercises no management jurisdiction of the Gulf sturgeon, other than permitting scientific or incidental take under the Endangered Species Act and enforcement. The NMFS conducts some research and data collection programs and comments on all projects that affect marine fishery habitat under the Fish and Wildlife Coordination Act and Section 10 of the Rivers and Harbors Act.

The NMFS has entered into a Cooperative Agrrement with the Department of the Army to Restore and Create Fish Habitat. Under this agreement, the NMFS and the COE coordinate efforts to identify federal projects that could be modified to enhance fish habitat.

Office of Ocean and Coastal Resource Management (OCRM). The OCRM asserts its authority through the National Marine Sanctuaries Program pursuant to Title III of the Marine Protection, Research, and Sanctuaries Act (MPRSA). The OCRM Estuarine Sanctuary Program has designated Looe Key in Monroe County, Rookery Bay in Collier County, the Apalachicola River and Bay in Franklin County, Florida, and Weeks Bay in Baldwin County, Alabama, as estuarine sanctuaries.

The OCRM may influence fishery management for Gulf sturgeon indirectly through administration of the Coastal Zone Management Program and by setting standards and approving funding for state coastal zone management programs. Some states in the Gulf utilize a portion of these monies in their habitat protection and enhancement programs including reef maintenance and enhancement.

Department of the Interior (DOI).

National Park Service (NPS). The NPS under the DOI may regulate fishing activities within national park boundaries. Such regulations may affect Gulf sturgeon within specific parks. The NPS has authority to protect fishes and fish habitat primarily through the establishment of coastal and nearshore national parks and national monuments. Everglades National Park in Florida and the Mississippi District of Gulf·Islands National Seashore are two examples of national park areas where Gulf sturgeon may occur.

U.S. Fish and Wildlife Service. The authority of the FWS to affect the management of the Gulf sturgeon is based primarily on the Endangered Species Act and the Fish and Wildlife Coordination Act. The FWS is the lead agency in developing the recovery plan for the subspecies under the Endangered Species Act. Under the Fish and Wildlife Coordination Act, the FWS, in conjunction with the NMFS, reviews and comments on proposals to alter habitat. Dam construction, drainage projects, channel alteration, wetlands filling and marine construction are projects that can potentially affect the Gulf sturgeon. Further, the FWS may seek mitigation of fishery resource impairment due to federal water-related development. The FWS has the responsibility to focus efforts on nationally significant fishery resources. The FWS also facilitates restoration by rebuilding certain major, economically valuable, anadromous, endangered, threatened, and interjurisdictional (managed by two or more states) fishery resources to full, self-sustainable productivity. Because the Gulf sturgeon is a threatened and an anadromous species, the FWS has conducted studies on various aspects of the subspecies' biology.

Gulf sturgeon occur in the aquatic portions (riverine, estuarine, marine) of national wildlife refuges (NWR) such as Pine Island NWR, Island Bay NWR, Passage Key NWR, Pinellas NWR, Chassahowitzka NWR,Cedar Keys NWR, Lower Suwannee NWR, St. Marks NWR, St. Vincent NWR, Florida, Bon Secour NWR, Alabama, Bogue Chitto NWR, Louisiana and Mississippi, and Delta NWR, Breton Island NWR, Bayou Sauvage NWR, Lacassine NWR, Louisiana. Fish and wildlife populations and their harvest within refuges are usually managed by the respective state which the refuge is located. Special use permits are required for commercial fishing on national wildlife refuges.

<u>National Biological Service.</u> The National Biological Service (NBS) is the Department of Interior's newest bureau. The NBS was created November 11, 1993, by consolidating the biological research, inventory, monitoring, and information transfer programs of seven Interior bureaus: FWS, NPS, MMS, USGS, Bureau of Land Management, Bureau of Reclaimation, and Office of Surface Mining. The Southeastern Biological Service Center (Center), Gainesville, Florida, of NBS was formerly a research center for FWS. The Center has conducted research on Gulf sturgeon since 1987 and will continue work in this area as requested by FWS and other agencies.

<u>Environmental Protection Agency.</u> The EPA, through its administration of the Clean Water Act, National Pollutant Discharge Elimination System (NPDES), may provide protection to Gulf sturgeon habitat. Applications for permits to discharge pollutants may be disapproved or conditioned to protect fresh and estuarine aquatic resources.

U.S. Department of the Army, Corps of Engineers. Gulf sturgeon habitat may be influenced by the COE's regulatory responsibilities pursuant to the Section 404 of the Clean Water Act and Section 10 of the Rivers and Harbors Act. Under these laws, the COE may authorize proposals to dredge, fill and construct in navigable waters (Section 10) or to discharge dredged or fill material into wetland areas and waters of the United States (Section 404). Such proposals could affect Gulf sturgeon habitat. The COE is also responsible for planning, construction and maintenance of dams, navigation channels and other projects that may affect Gulf sturgeon habitat.

Treaties and Other International Agreements. There are no treaties or other international agreements that affect the Gulf sturgeon. No foreign fishing applications for Gulf sturgeon harvest have been submitted to the United States government.

Federal Laws, Regulations and Policies. The following Federal laws, regulations and policies may directly and indirectly influence the habitat, populations and ultimately the management of the Gulf sturgeon.

Anadromous Fish Conservation Act (AFCA). The AFCA authorizes the Secretary of the Interior to initiate cooperative programs with the states to conserve, develop and enhance the nation's anadromous fisheries. The Act authorizes construction, installation, maintenance and operation of structures to improve or facilitate feeding, spawning and free migration of anadromous fish.

Coastal Zone Management Act and Estuarine Areas Act. Congress passed policy on values of estuaries and coastal areas through these Acts. Comprehensive planning programs to be carried out at the state level, were established to enhance, protect, and utilize coastal resources. Federal activities must comply with the individual state programs. Habitat may be protected by planning and regulating development damage to sensitive coastal habitats.

Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). This act is also referred to as the "Superfund". It can provide funding for "clean-up" of important habitat areas affected by oil spills or other distinct pollution discharge events.

Endangered Species Act (ESA). The ESA provides for the protection of habitat necessary for the continued existence of species listed as threatened or endangered. Section 7 of the ESA requires consultation with the FWS or NMFS by a federal agency if an action authorized, funded or carried out by such agency may affect a listed species or its critical habitat (a legal, area-specific designation). Section 7 also prohibits any federal action that would jeopardize the continued existence of a listed species or its critical habitat. Section 9 of the ESA prohibits any person or entity from "taking" a listed species without a proper permit from the FWS or NMFS. Under the ESA, taking may include harassment or habitat degradation if such would interfere with feeding, reproduction or other essential life functions. The ESA also requires preparation of a recovery plan for each listed species outlining actions needed to allow the particular species to reach a population level at which it may be delisted.

Federal Power Act (FPA). The FPA regulates the construction and operation of hydroelectric power plants through a system of licenses and permits issued by the federal Energy Regulatory Commission (FERC) (formerly Federal Power Commission). The FWS, NMFS, state agencies and others may review proposed licenses and make recommendations with respect to the needs of instream flow for fish and wildlife downstream of dams as well as the impacts that reservoir establishment may have on fish and wildlife upstream of the dams. The Act also provides for construction of fish passage facilities during dam or diversion construction. Dams are likely major factors affecting anadromous fish populations in some Gulf streams.

Federal Water Pollution Control Act (FWPCA). Also called the "Clean Water Act", the FWPCA provides for the protection of water quality at the federal level. The law also provides for assessment of injury, destruction, or loss of natural resources caused by discharge of pollutants.

Of major significance is Section 404 of the Clean Water Act (CWA), which prohibits the discharge of dredged or fill material into navigable waters without a permit. Navigable waters are defined under the CWA to include all waters of the United States, including the territorial seas and wetlands adjacent to such waters. The permit program is administered by the COE. The Environmental Protection Agency (EPA) may approve delegation of Section 404 permit authority for certain waters (not including traditional navigable waters) to a state agency; however, it retains the authority to prohibit or deny a proposed discharge under Section 404(c) of the CWA. Recent attempts to revise Section 404 or change the legal definition of wetlands may affect the utility of the CWA in wetlands protection. Although of limited applicability to anadromous fish restoration, Section 404 may be important in protecting certain types of coastal habitats or in protecting water quality in certain streams. It may also be a consideration in approval of certain types of restoration projects.

The FWPCA also authorized programs to remove or limit the entry of various types of pollutants into the nation's waters. A point source permit system was established by the EPA and is now being administered at the state level in most states. This system, referred to as the National Pollutant Discharge Elimination System (NPDES), sets specific limits on discharge of various types of pollutants from point source outfalls. A non-point source control program focuses primarily on the reduction of agricultural siltation and chemical pollution resulting from rain runoff into the nation's streams. This control effort currently relies on the use of land management practices to reduce surface runoff through programs administered primarily by the Department of Agriculture.

Both chemical contamination and siltation may be major factors limiting populations of anadromous Gulf fish species. Efforts to achieve anadromous fish restoration in key river drainages should be aimed at assuring compliance with established point and non-point source reduction programs in these basins.

Federal Water Project Recreation Act. This Act requires that consideration be given to fish and wildlife enhancement in federal water projects.

Fish and Wildlife Act of 1956. This act provides assistance to states in the form of law enforcement training and cooperative law enforcement agreements. It also allows for disposal of property abandoned or forfeited in conjunction with convictions. Some equipment may be transferred to states. The act prohibits airborne hunting and fishing activities.

Fish and Wildlife Coordination Act (FWCA). The Fish and Wildlife Coordination Act (FWCA) is the primary law providing for consideration of fish and wildlife habitat values in conjunction with federal water development activities. Under this law the Secretaries of Interior and Commerce may investigate, report and advise on the effects federal water development projects may have on fish and wildlife habitat. Such reports and recommendations, which require concurrence of the state(s) involved, must accompany the construction agency's request for congressional authorization, although, the construction agency is not bound by the recommendations. Construction agencies may transfer funds to the FWS or NMFS to investigate and report on specific projects.

The FWCA also applies to water-related activities proposed by other organizations or individuals if those activities require a federal permit or license. The FWS and NMFS may review the proposed permit action and recommend to the permitting agencies to avoid or mitigate any potential adverse effects on fish and wildlife habitat.

Fish Restoration and Management Projects Act of 1950. Under this act, the DOI is authorized to provide funds to state fish and game agencies for fish restoration and management projects. Funds for protection of threatened fish communities that are located within state waters could be made available under the act.

Food and Agriculture Act of 1962. This Act established a Resource Conservation and Development Program for regionally-sponsored flood control and drainage projects that receive financial and technical assistance from the Soil Conservation Service. Though not as active a program as it once was, activities under this program may have relevance, both positive and negative, to anadromous fish habitat protection, restoration or enhancement.

Lacey Act of 1981, as amended. The Lacey Act prohibits import, export and interstate transport of illegally-taken fish and wildlife. As such, the Act provides for federal prosecution for violations of state fish and wildlife laws. The potential for federal convictions under this Act, with its more stringent penalties, has probably reduced interstate transport of illegally-possessed Gulf sturgeon.

Magnuson Fishery Conservation and Management Act. This Act provides for the conservation of habitats throughout the ranges of anadromous species within the Exclusive Economic Zone (EEZ). It mandates the preparation of fishery management plans for important fishery resources and sets national standards to be met by such plans. Each plan attempts to define, establish and maintain the optimum yield for a given fishery.

Marine Plastic Research and Control Act of 1987 and MARPOL Annex V. MARPOL Annex V is a product of the International Convention for the Prevention of Pollution from Ships, 1973/78. Regulations under this Act prohibit ocean discharge of plastics from ships; restrict discharge of other types of floating ship's garbage (packaging and dunnage) for up to 25 nautical miles from any land; restrict discharge of victual and other recomposable waste up to 12 nautical miles from land; and require ports and terminals to provide garbage reception facilities. The MPRCA of 1987 and 33 CFR, Part 151, Subpart A, implement MARPOL V in the United States.

Marine Protection, Research and Sanctuaries Act of 1972 (MPRSA), Titles I and III and the Shore Protection Act of 1988 (SPA). The MPRSA protects fish habitat through establishment and maintenance of marine sanctuaries. This Act and the SPA regulate ocean transportation and dumping of dredged materials, sewage sludge and other materials. Criteria for issuing permits include considering the effects dumping has on the marine environment, ecological systems and fisheries resources. Permits are issued by the Corps of Engineers.

National Environmental Policy Act (NEPA). The NEPA requires an environmental review process of all federal actions. This includes preparation of an environmental impact statement for major federal actions that may affect the quality of the human environment. Less rigorous environmental assessments are reviewed for most other actions while some actions are categorically excluded from formal review. These reviews provide an opportunity for the agency and the public to comment, on projects that may impact fish and wildlife habitat.

Oil Pollution Act. This Act provides a degree of protection to coastal fisheries habitat by regulating discharge of oil from United States registry ships. Under the Act, tankers cannot discharge oil within 50 nautical miles of land, and other ships must discharge as far as practicable from land.

Outer Continental Shelf (OCS) Lands Act Amendments of 1979. These Amendments provide for assessments of the effects oil and gas exploration, development and production have on biological resources. The law also provides a channel for comments on federal approval of leasing OCS areas for exploration and development. Oil and gas leasing activities could be of concern for coastal anadromous fish habitat and offshore winter habitat of the Gulf sturgeon.

River and Harbor Act of 1899. Section 10 of the River and Harbor Act requires a permit from the U.S. Army Corps of Engineers (COE) to place structures in navigable waters of the United States or modify a navigable stream by excavation or filling activities.

Water Resources Development Acts (WRDA). These legislative actions authorize the COE to study and/or construct individual water resource projects. Prior to 1974 such acts were known as the "Flood Control Act of (year)", the "River and Harbor Act of (year)" or commonly called the "Omnibus Bill." Beginning in 1974 these laws have been referred to as the "WRDA of (year)". Numerous projects may be authorized under these Acts in any given year. Under the FWCA, "Wildlife conservation shall receive equal consideration and be coordinated with other features of water-resource development programs . . ." and the FWS, NMFS and state fish and wildlife agencies may review, comment and make recommendations to the COE regarding these projects' impacts on fish and wildlife resources. These comments may address the avoidance, mitigation or compensation for habitat damages.

Of particular relevance to anadromous fish habitat restoration or enhancement is the WRDA of 1986. This Act authorized the COE to study and construct environmental enhancement projects in conjunction with existing federal water projects.

STATE MANAGEMENT INSTITUTIONS, LAWS, REGULATIONS AND POLICIES.

State management institutions, laws and regulations for the Gulf sturgeon are relatively consistent among the four Gulf States within the species' range. Each state delegates substantial authority to its administrative agencies for establishing management regulations. Brief narrative descriptions are presented below for each state institution. Important state laws, regulations and policies are also summarized. **To the greatest extent possible, these requirements are current to the date of publication.**

FLORIDA

Administrative Organization.

Florida Marine Fisheries Commission
2540 Executive Center Circle West, Suite 106
Tallahassee, FL 32301
Telephone: (904) 487-0554

The Florida Marine Fisheries Commission, a seven-member board appointed by the governor and confirmed by the senate, was created by the Florida legislature in 1983. This commission was delegated rule-making authority over marine life in the following areas of concern: gear specification; prohibited gear; bag limits; size limits; species that may not be sold; protected species; closed areas; seasons; quality control codes with the exception of specific exemptions for shellfish; and special considerations relating to oyster and clam relaying. All rules passed by the commission require approval by the governor and cabinet. The commission does not have authority over endangered species, license fees, penalty provisions or over regulation of fishing gear in residential saltwater canals.

Florida Department of Environmental Protection (FDEP)
Division of Marine Resources
3900 Commonwealth Boulevard
Tallahassee, Florida 32303
Telephone: (904) 488-6058

This agency is charged with the administration, supervision, development and conservation of marine natural resources in Florida. The Florida Department of Natural Resources was the predecessor marine resources agency until its merger with the Florida Department of Environmental Regulation July 1, 1993. The agency is headed by the Governor and Cabinet. The governor and cabinet serve as the seven-member board that approves or disapproves all rules and regulations promulgated by the FDEP. The administrative head of the FDEP is the Department Secretary. Within the FDEP the Division of Marine Resources, through Section 370.02(2), Florida Statutes, is empowered to conduct research directed toward management of marine and anadromous fisheries in the interest of all people of Florida. The Division of Law Enforcement is responsible for enforcement of all marine resource related laws and all rules and regulations of the department. The Division of Marine Resources has the responsibility of overseeing the management and research efforts on the Gulf sturgeon including issuance of collecting permits for the subspecies.

Florida Game and Fresh Water Fish Commission.
Division of Wildlife
620 South Merdian Street
Tallahassee, Florida 32399
Contact: Mr. Don A. Wood, Endangered Species Coordinator
Telephone: (904) 488-3831

This agency is charged with the administration, supervision, development and conservation of wildlife and fresh water aquatic life in Florida. The FGFC is a constitutionally autonomous agency and is overseen by a governor appointed five-member board. The administrative head of the FGFC is the executive director. Within the FGFC the Division of Wildlife Resources, in accordance with the Florida Endangered and Threatened Species Act of 1977, Section 372.072, Florida Statutes, and the Wildlife Code of the State of Florida, Title 39, Florida Administrative Code, Article IV, Sec. 9, Florida Constitution, is responsible for research and management of listed fresh water and upland species. These efforts include the administrative designation of all wildlife species (including marine and estuarine species), issuance of collection permits, and various types of research of listed upland and fresh water aquatic wildlife species. The Gulf sturgeon was listed as a species of special concern by the FGFC in 1987.

Florida has habitat protection and permitting programs and a federally-approved Coastal Zone Management (CZM) program.

Legislative Authorization. Chapter 370 of the Florida Statutes Annotated contains law regulating coastal fisheries. The legislature passes statutes for the management of fisheries resources as well as specific laws which are applicable within individual counties.

Reciprocal Agreement and Limited Entry Provisions. Not applicable, since any take of Gulf sturgeon is illegal in Florida.

Commercial Landings Data Reporting Requirements. Not applicable since all take of Gulf sturgeon is illegal in Florida.

Penalties for Violations. Penalties for violations of Florida statutes and regulations are prescribed in Section 370.021, Florida Statutes. Upon the arrest and conviction for violation of any of the regulations or laws, the license holder shall show just cause why his saltwater license should not be suspended or revoked.

Annual License Fees. Not applicable, since all take of Gulf sturgeon is illegal in Florida.

Laws and Regulations. It is illegal to take *Acipenser oxyrinchus* by any means statewide according to Rule No. 46-15.01 (1984) of the Florida Marine Fisheries Commission. (Most federal and state agencies have used the specific name *A. oxyrinchus* instead of the subspecific name *A. o. desotoi*.

ALABAMA

Administrative Organization.

Alabama Department of Conservation and Natural Resources (ADCNR)
Alabama Marine Resources Division (AMRD)
P.O. Box 189
Dauphin Island, Alabama 36528
Telephone: (205) 861-2882

Management authority of fishery resources in Alabama is held by the Commissioner of the Department of Conservation and Natural Resources. The Commissioner may promulgate rules or regulations designed for the protection, propagation and conservation of all seafood. He may prescribe the manner of taking, times when fishing may occur and designate areas where fish may or may not be caught; however, all regulations are to be directed toward the best interest of the seafood industry.[5]

Most regulations are promulgated through the Administrative Procedures Act approved by the Alabama Legislature in 1983; however, bag limits and seasons are not subject to this Act. The Administrative Procedures Act outlines a series of events that must precede the enactment of any regulations other than those of an emergency nature. Among this series of events are (a) the advertisement of the intent of the regulation, (b) a public hearing for the regulation, (c) a 35-day waiting period following the pubic hearing to address comments from the hearing and (d) a final review of the regulation by a joint house and senate review committee.

Alabama also has the Alabama Conservation Advisory Board (ACAB) that is endowed with the responsibility to provide advice on policies of the ADCNR. The board consists of the governor, the ADCNR commissioner and ten board members.

The AMRD has responsibility for enforcing state laws and regulations, for conducting marine biological research and for serving as the administrative arm of the commissioner with respect to marine resources. The division recommends regulations to the commissioner.

Alabama has a habitat protection and permitting program and a federally approved CZM program.

Legislative Authorization. Chapters 2 and 12 of Title 9, Code of Alabama, contain statutes that concern marine fisheries.

Reciprocal Agreement and Limited Entry Provisions. Not applicable since all take of Gulf sturgeon is illegal in Alabama.

Commercial Landings Data Reporting Requirements. Not applicable since all take of Gulf sturgeon is illegal in Alabama.

Penalties for Violations. Take of Gulf sturgeon is illegal in Alabama, any take is considered a Class C misdemeanor and punishable by fines up to $500.00 and three months in jail.

Annual License Fees. Not applicable since all take of Gulf sturgeon is illegal in Alabama.

Laws and Regulations. It is currently illegal to take Gulf sturgeon in freshwater or coastal waters in Alabama. Alabama has no official State list of threatened and endangered species. *Acipenser oxyrinchus* is considered a threatened species by the Symposium on Endangered and Threatened Plants and Animals of Alabama (Boshung 1976).

MISSISSIPPI

Administrative Organization.

Mississippi Department of Wildlife, Fisheries and Parks (MDWFP)
Bureau of Marine Resources (BMR)
2620 Beach Boulevard
Biloxi, Mississippi 39531
Telephone: (601) 385-5860

The MDWFP administers coastal fisheries and habitat protection programs through the BMR. Authority to promulgate regulations and policies is vested in the Mississippi Commission on Wildlife, Fisheries and Parks, the controlling body of the MDWFP. The commission consists of five members appointed by the governor. The commission has full power to "manage, control, supervise and direct any matters pertaining to all saltwater aquatic life not otherwise delegated to another agency" (Mississippi Code Annotated 49-15-11).

Mississippi has a habitat protection and permitting program and a federally approved CZM program.

Legislative Authority. Chapter 49-15 of the Mississippi Code of 1972 (Annotated) contains provisions for the management of marine fisheries resources.

Reciprocal Agreement and Limited Entry Provisions. Not applicable since it is illegal to take Gulf sturgeon anywhere in the State of Mississippi.

Commercial Landings Data Reporting Requirements. Not applicable since it is illegal to take Gulf sturgeon anywhere in the State of Mississippi.

Penalties for Violations. Any person, firm or corporation violating any of the provisions of Chapter 49-15 or any ordinance duly adopted by the commission, unless otherwise specifically provided for herein, shall, on conviction, be fined not less than $100, nor more than $500, for the first offense, unless the first offense is committed during a closed season, in which case the fine shall be not less than $500, nor more than $1,000; and not less than $500, nor more than $1,000, for the second offense when such offense is committed within a period of 3 years from the first offense; and not less than $2,000 nor more than $4,000, or imprisonment in the county jail for a period not exceeding 30 days for any third or subsequent offense when such offense is committed within a period of 3 years from the first offense and also upon conviction of such third or subsequent offense, it shall be the duty of the court to revoke the license of the convicted party and of the boat or vessel used in such offense, and no further license shall be issued to such person or for said boat to engage in catching or taking of any seafoods from the waters of the State of Mississippi for a period of 1 year following such conviction. Further, upon conviction of such third or subsequent offense committed within a period of 3 years from the first offense, it shall also be the duty of the court to order the forfeiture of any equipment or nets used in such offense. Provided, however, that equipment as used in this section shall not mean boats or vessels. Any person convicted and sentenced under this section shall not be considered for suspension or other reduction of sentence. Except as provided under subsection 5 of Section 49-15-45, any fines collected under this section shall be paid to the Mississippi Commission on Wildlife, Fisheries and Parks to be paid into the Seafood Fund.

Annual License Fees. Not applicable since it is illegal to take Gulf sturgeon anywhere in the State of Mississippi.

Laws and Regulations. *Acipenser oxyrinchus* was listed as an endangered species by the Mississippi Game and Fish Commission and the Rare and Endangered Species Committee (1975) and is protected by law. The subspecies is also listed as endangered by the Mississippi Natural Heritage Program, 1977, and as a Special Animal Species by the Mississippi Parks Commission, Bureau of Outdoor Recreation, Jackson, MS.

LOUISIANA

Administrative Organization.

Louisiana Department of Wildlife and Fisheries (LDWF)
P.O. Box 98000
Baton Rouge, Louisiana 70898
Telephone: (504) 765-3617

The LDWF is one of 21 major administrative units of the Louisiana government. A seven-member board, the Louisiana Wildlife and Fisheries Commission (LWFC) is appointed by the Governor. Six of the members serve overlapping terms of six years, and one serves a term concurrent with the Governor. The commission is a policy-making and budgetary-control board with no administrative functions. The legislature has sole authority to establish management programs and policies; however, the legislature has delegated certain authority and responsibility to the LDWF. The Secretary of the LDWF is the executive head and chief administrative officer of the department and is responsible for the administration, control and operation of the functions, programs and affairs of the department. The secretary is appointed by the Governor with consent of the Senate.

Within the administrative system, an Assistant Secretary is in charge of the Office of Fisheries. In this office a Marine Fisheries Division and an Inland Fisheries Division may have management jurisdiction over the Gulf sturgeon. The Enforcement Division, in the Office of the Secretary, is responsible for enforcing all fishery statutes and regulations.

The LDWF's Natural Heritage Program is responsible for administering the laws, rules, and regulations regarding threatened and endangered species (R.S. 56:1830). In addition, under a full authorities Section 6 agreement with the FWS, the take of threatened and endangered species may be authorized by permits issued by the Department.

Louisiana has habitat protection and permitting programs and a federally approved CZM program.

Legislative Authorization. Title 56 Louisiana Revised Statutes contains rules and regulations that govern marine fisheries in the state.

Reciprocal Agreement and Limited Entry Provisions. Not applicable, since take of Gulf sturgeon is illegal in Louisiana.

Commercial Landings Data Reporting Requirements. Not applicable, since take of Gulf sturgeon is illegal in Louisiana.

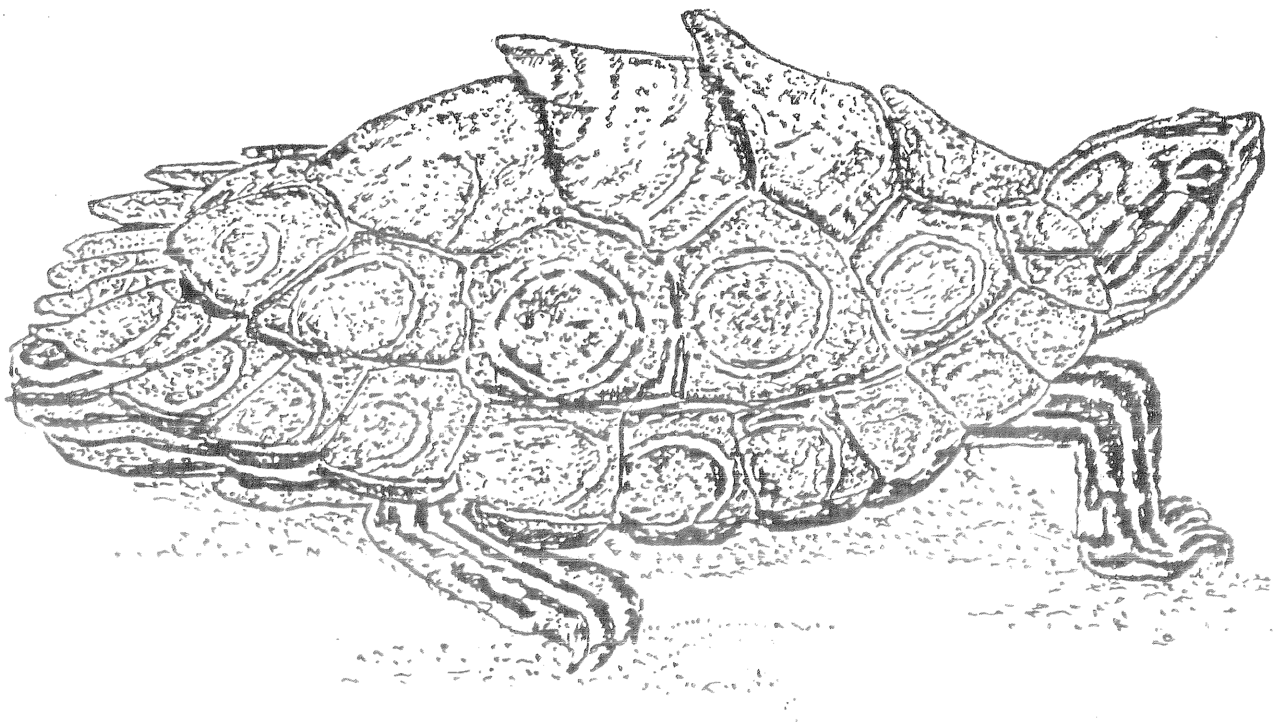
Penalties for Violations. The fine for each illegally caught fish is $2,500.00

Annual License Fees. Not applicable, since take of Gulf sturgeon is illegal in Louisiana.

Laws and Regulations. Louisiana law currently prohibits take of all sturgeon anywhere in the state. The Louisiana Division of Natural Heritage is responsible for listing of endangered and threatened species.

RINGED SAWBACK TURTLE

RECOVERY PLAN



U.S. Fish and Wildlife Service
Atlanta Georgia

A Recovery Plan For The Ringed Sawback Turtle

Graptemys oculifera

Prepared by

James H. Stewart

U.S. Fish and Wildlife Service

For

Southeast Region

U.S. Fish and Wildlife Service

Atlanta, Georgia

Approved [signature]
Regional Director, Southeast Region

Date: April 8, 1988

RECOVERY PLAN EXECUTIVE SUMMARY

1. Point or condition when the species can be considered recovered?

The primary objective of the recovery plan is to provide secure habitat for the ringed sawback turtle in two stretches of the Pearl River for a total protected area of 150 river miles. These reaches must be on opposite ends of Ross Barnett Reservoir at Jackson, and contain a minimum of 30 miles in either reach.

Delisting should occur on a rangewide basis when the two river reaches are protected, there is evidence of a stable or increasing population over a 10 year period, and a monitoring plan is developed and implemented to ensure a continuing stable population.

2. What must be done to reach recovery?

Determine the habitat requirements, including food sources for the various life stages of the ringed sawback turtle, and maintain at least 150 river miles of habitat that meets those requirements.

The primary steps are to characterize physical parameters of required habitat, determine reproductive requirements, food sources, population structure, and activity periods and behavior. On the basis of this information, identify and protect the two river reaches.

Attaining recovery depends upon protection of the required habitat. The areas where the ringed sawback turtle is common are known to some extent. The population status and trends and influencing factors are not known. Regulatory agencies must provide for habitat protection in areas identified as required habitat.

3. What management/maintenance needs have been identified to keep the species recovered?

The required habitat must be protected by the appropriate regulatory agencies. A monitoring plan to track population trends and protection success is a critical element.

Disclaimer

This is the completed ringed sawback turtle recovery plan. It has been approved by the U.S. Fish and Wildlife Service. It does not necessarily represent official positions or approvals of cooperating agencies, and it does not necessarily represent the views of all individuals who played a role in preparing this plan. This plan is subject to modification as dicated by new findings, changes in species status, and completion of tasks described in the plan. Goals and objectives will be attained and funds expended contingent upon appropriations, priorities, and other constraints.

ADDITIONAL COPIES ARE FOR SALE FROM:

Fish and Wildlife Reference Service
6011 Executive Blvd.
Rockville, Maryland 20852
301/770-3000
1-800-582-3421

TABLE OF CONTENTS

Part I: Introduction

Background

On December 23, 1986, the U.S. Fish and Wildlife Service published in the *Federal Register* a final rule indicating its determination that the ringed sawback turtle (*Graptemys oculifera*) is a threatened species under the Endangered Species Act of 1973, as amended. The ringed sawback turtle is restricted to the Pearl River and one tributary, the Bogue Chitto River, in Mississippi and Louisiana.

The ringed sawback turtle was described by Baur in 1890 as *Malacoclemmys oculifera* and renamed *Graptemys oculifera* in 1893. The type specimens were a group of turtles acquired for the United States National Museum by Gustave Kohn and reportedly came from Mandeville, Louisiana, and Pensacola, Florida (Cagle 1953). On the basis of a 1900 statement to this effect by George E. Beyer, then Curator of the Tulane Museum, Cagle said they were probably purchased in the French Quarter Market in New Orleans, Louisiana. Due to the absence of ringed sawback turtles from collections in southern Alabama and Florida, Cagle considered the Pensacola record to be erroneous, although Kohn had accepted the locality datum of the individual from whom the purchase was made. The Mandeville record probably is from the Pearl River, 26 miles to the east, since there is no suitable habitat near Mandeville.

Description

The ringed sawback turtle is a small turtle (adults 7.5 - 22 cm) having a yellow ring bordered inside and outside with dark olive-brown on each shield of the upper shell or carapace and a yellow undershell or plastron. The head has a large yellow spot behind the eye, two yellow stripes from the orbit backwards and a characteristic yellow stripe covering the whole lower jaw (Cagle 1953). Males are considerably smaller than females.

The only other member of the genus Graptemys in the Pearl River is the Alabama map turtle (G. pulchra). The ringed sawback turtle differs from the Alabama map turtle in the size of yellow markings on the head and the presence of yellow rings on each shield of the carapace. Closely related but distinct species occur in rivers to the east and west of the Pearl River.

Distribution

The ringed sawback turtle has been collected only from the Pearl and Bogue Chitto Rivers. In the Pearl River, it occurs from near coastal salt water influence upstream to Neshoba County, Mississippi (Cliburn 1971). Within the Pearl River, densities are greater above Ross Barnett Reservoir and below the river stretch impacted by the Jackson metropolitan area. In the latter, the population appears to decrease downstream of Bogalusa, Louisiana. In the Bogue Chitto River, this species has been collected as far upstream as Franklinton, Louisiana. The size of the Bogue Chitto River is a possible limiting factor to this species.

Description of the Habitat

The ringed sawback turtle is encountered most frequently in river stretches having a moderate current, numerous basking logs and sand beaches for nesting. The river must be wide enough to allow sun penetration for several hours. The factors influencing suitability of nesting sites for *Graptemys* appear to be sand particle size, elevation above the water level, cover quality and distance from the water's edge (Shealy 1976, Lahanas 1982).

Lahanas (1982) observed *G. nigrinoda* to nest in unvegetated and short grass situations with about equal frequency. The substrate was very fine sand. Nests were constructed from 4.3 to 193 m from shore with most nests within 35 m of shore. The great distances from shore may be a reflection of the large nesting area on Gravine Island. Nests were always located above mean water level with an average elevation of 4.2 m. Elevation selected for nesting appeared to be the result of how far an individual traveled inland, rather than an elevation preference.

Anderson (1958) found nests of *G. oculifera* on the landward margin of sand bars. Cagle (1953) collected one mature female *G. oculifera* that had moved to a clump of grass. Tracking her movements, Cagle found trial nests and a nesting attempt that was apparently abandoned due to interference by roots.

The most consistent characteristic associated with nesting sites of _G. pulchra_ was very fine sand. This sand is easily excavated, forms a fine crust when wetted and retains moisture beneath this crust. Nests of _G. pulchra_ were generally 3-15 m from the water's edge and at an elevation of 2-3 m above the existing water level. If gently sloping banks were present, _G. pulchra_ would travel greater distances to find the desired elevation (Shealy 1976).

Basking sites must be present and relatively safe. _G. pulchra_ were observed to use the tops of toppled trees for basking only when there was some water covering the trunk between the top and the river bank. When water levels dropped and the trunk was continuously exposed, the turtles ceased using the tree for basking (Shealy 1976).

Life History

Cagle (1954), in the descriptions of _G. flavimaculata_ and _G. nigrinoda_ concluded these two species and _G. oculifera_ formed a unique complex that has been referred to by other investigators as the "narrow-head" _Graptemys_. The most comprehensive study on the "narrow-head" complex is by Lahanas (1982). It is on this study that most of the following life history discussion is based.

The limited information available on _G. oculifera_ is Cagle's work in the early 1950s. Cagle (1953) concluded that _G. oculifera_ males matured at five years of age and that toe nails and the pre-anal tail length were not always conspicuously elongate. The smallest mature male was 6.52 cm (2.6 inches) plastron length. Cagle did not provide an age at which _G. oculifera_ females reached maturity but did record the smallest mature female at 12.8 cm (5.0 inches) plastron length. He concluded females grew more rapidly than males during their second year and that growth virtually ceased in both sexes at maturity.

Lahanas (1982) found female _G. nigrinoda_ grew at twice the rate of male _G. nigrinoda_ for the first five years. He collected immature female _G. nigrinoda_ that were 6 to 8 years old and 159 to 168 mm (6.2 - 6.5 inches) in plastron length. The smallest sexually mature females collected were 167 to 177 mm (6.5 - 7.0 inches) in plastron length, suggesting they were at least 9 years old. From these data he inferred that female _G. nigrinoda_ mature at 8 or 9 years of age and a plastron length of approximately 170 mm (6.7 inches). Shealy (1976) found _G. pulchra_ males matured in 3 or 4 years while females were 14 years old at maturity. Webb (1961) found male _G. ouachitensis_ in Lake Texoma, Oklahoma, were mature at 2 or 3 years while females were 6 or 7 years of age.

Lahanas (1982) concluded that _G. nigrinoda_ produced 3 or 4 clutches annually with an average clutch size of 5-6 eggs. Cagle (1953) collected a small nesting female _G. oculifera_ that had 3 eggs in the oviduct and 4 enlarged follicles. This turtle probably would have deposited two clutches totalling 7 eggs. Shealy (1976) autopsied a large female _G. pulchra_ that exhibited the potential production of 71 eggs in the season. Cagle (1952) found reproductive potential of up to 51 eggs with an average of 17 eggs in a season for _G. barbouri_. The narrow headed _Graptemys_ may have a lower reproductive potential than other species of _Graptemys_.

In _G. nigrinoda_, mating likely occurs in late spring and early summer but may occur at any time of the year (Lahanas 1982). _G. pulchra_ likely breeds in September to November with nesting from late April to late July. A single mating may be sufficient for several fertilizations since females can apparently store viable sperm for several months or possibly years. A female _G. oculifera_ collected by Cagle (1953) in April did not yet have eggs in the oviduct, while he observed one nesting in early June. Lahanas (1982) concluded that the nesting season for _G. nigrinoda_ extended from mid-May to early August.

In _Graptemys_, nesting activity may occur during the day or night but rarely both by the same species. _G. nigrinoda_ always nests after dark with the highest activity during the early hours of darkness (Lahanas 1982). _G. pulchra_ nests during the day (Shealy 1976). The nesting _G. oculifera_ observed by Cagle (1953) was during the day.

Graptemys' nests are about 15 cm (6 inches) deep with the eggs covered with packed sand to the top of the cavity. The egg incubation period for _G. oculifera_ is unknown. Under controlled conditions, Ewert (1979) artificially incubated _G. oculifera_ eggs in 62.8 days. Under similar conditions, Shealy (1976) incubated _G. pulchra_ eggs in 74-79 days.

Lahanas (1982) observed 9 clutches of G. nigrinoda incubating under natural conditions to require an average of 63-65 days. Hatchling turtles remain in the nest for up to several days after pipping to absorb the remaining egg yolk. Shealy (1976) determined the average time between nesting and emergence to be 95 days for G. pulchra. Lahanas (1982) observed G. nigrinoda remained in the nest for 2-5 days after pipping.

Nesting is generally on wide sand beaches (Lahanas 1982, Shealy 1976). Nest temperature is a determining factor in sex determination according to a study on three species of Graptemys (Bull 1985). In a study of G. geographica, G. ouachitensis, and G. pseudogeographica, only males were produced when nest temperatures were below 28^{o}C. If nest temperatures were above 30.5^{o}C, only females were produced. The critical time for nest temperature influence on sex determination was in the 4th to 7th weeks of incubation (Bull 1985).

Egg mortality is an important factor in reproductive success. Shealy (1976) found G. pulchra egg mortality exceeded 90 percent. Eighty-two percent of G. nigrinoda eggs were destroyed (Lahanas 1982). The mortality for G. nigrinoda could have been higher if the investigator had not been on site and disrupting the predatory activities. The effect of long periods of egg inundation under natural conditions is unknown.

Studies of other species of Graptemys indicate a diet of insects, snails, and clams (Cagle 1952, Webb 1961, Shealy 1976). Juveniles and small males of G. pulchra contained primarily insects while large females fed almost exclusively upon mussels by crushing the shell with their powerful jaws (Shealy 1976). Lahanas (1982) found G. nigrinoda used algae as a primary food and did not regard them as a mollusk specialist. Cagle (1953) found the stomachs of 10 G. oculifera contained only the fragments of insects. Fish and carrion may be an occasional and opportunistic food source.

A major factor in activity is water temperature (Shealy 1976). Although basking may occur during all months, a peak in activity occurs in March and April, continues through July and declines from July to October. Basking probably serves several functions with elevation of the body temperature as a primary function. The drying that occurs with basking also inhibits fungal and algal growth, ectoparasites and infections (Shealy 1976). In G. pulchra, basking did not occur on cloudy days when water temperature exceeded air temperature. This implies basking is primarily for thermoregulation (Shealy 1976). Turtles will quickly drop from basking sites if disturbed and may drop in response to another turtle plunging into the water.

Nocturnal activity of Graptemys is largely unknown. Individuals have been observed lingering close to the surface among tree branches and roots. Adults may feed at night and hide during the day when not basking.

Predation

Nest predation is the dominant factor inhibiting population growth in *G. pulchra* (Shealy 1976). He found at least 95 percent of all nests were destroyed by predators. Lahanas (1982) found 82 percent of *G. nigrinoda* destroyed by predators. Primary predators are the fish crow and raccoon. The fish crow will frequently follow a female to the nest site and excavate the eggs after laying. Raccoons apparently locate the nests by the odor of turtle urine. Most nests were destroyed by predators within 12-24 hours of laying. Cagle (1950) found the most important predators of *Pseudemys scripta* nests were skunks and raccoons.

Predation on hatchlings has not been observed. It is likely that large gars, herons, and alligator snapping turtles occasionally feed upon hatchlings (Shealy 1976). The only significant predator of adult turtles is man who shoots basking turtles and collects them for the commercial turtle trade.

Limiting Factors

Very little competition seems to occur among individuals or species. Because food is generally abundant if the habitat is satisfactory, the major limiting factor appears to be habitat availability. Competition for basking sites probably is not important at the population level in undisturbed habitat. The limitation of *G. oculifera* to the Pearl River system likely is from drainage isolation and the absence of overland migratory movements. The degree of adaptability to pond or lake situations has not been determined, but observations suggest G. oculifera marginally survives in such situations (McCoy and Vogt 1980). Nesting site requirements may be limiting factors. Basking sites probably are necessary for health, if not survival, and may be a limiting factor.

Reasons for Decline and Continuing Threats

The decline of the ringed sawback turtle is primarily due to habitat modification and water quality degradation. Construction of Ross Barnett Reservoir, modification of the west channel of the Pearl River to Bogalusa, Louisiana, and floodplain clearing at Jackson, Mississippi have impacted 21 percent of the historic range. Ross Barnett Reservoir modified 30 river miles (RM) to the exclusion of ringed sawback turtles. The channel and floodplain modifications at Bogalusa and Jackson have not eliminated this species but apparently have caused a decline in the population. Cliburn (1971) collected 12 G. oculifera from the Pearl River in the vicinity of the Highway 80 bridge at Jackson. Service biologists were unable to capture any G. oculifera on two occasions when using techniques similar to Cliburn's. Three surveys by Service biologists of basking turtles in the Pearl River at Jackson concluded the G. oculifera population was comprised almost completely of adults. The ringed sawback population in the vicinity of Bogalusa, Louisiana, and downstream has declined (pers. comm. R. Lohoefenor).

Projects planned or authorized will impact up to 28 percent of the remaining Pearl River habitat. Flood control studies on-going or planned for the Pearl River at Slidell, Louisiana and Pearlington, Morgantown, Monticello, Foxworth, Columbia, Carthage, and Leake County, Mississippi continue to threaten this turtle. Authorized channelization of 100 RM of the Bogue Chitto River would likely extirpate the ringed sawback turtle from this stream. Flood control studies on reaches of the Bogue Chitto River at Franklinton, Louisiana and Tylertown, Mississippi may lead to river modifications which would threaten this habitat.

Other threats include continued channelization in the drainage, which produces increased runoff and heavy siltation. Drainage ditches from agriculture fields may increase the amount of pesticides that reach the rivers. Sand and gravel dredging continues to impact reaches of ringed sawback turtle habitat.

Current Status and Population Trends

There are two vigorous population centers in the Pearl River, separated by Ross Barnett Reservoir and the Jackson metropolitan area. Information needed to evaluate current population trends within these centers is lacking. Much of the life history must be determined before we can evaluate trends and take protective action for recovery.

Part II: Recovery

A. Recovery Objective

The objective of this plan is to remove the ringed sawback turtle from the list of threatened species. The criteria for delisting the species are:

(1) Protection of a total of 150 miles of the turtle's habitat in two reaches of the Pearl River. There must be a minimum of 30 miles in either reach with the total protected area totalling 150 river miles.

(2) Evidence of a stable or increasing population over at least a ten year period in these two Pearl River reaches.

(3) An established, continuing plan of periodic monitoring of population trends and habitat to ensure a stable population in these river reaches.

B. Step-down Outline

1. Characterize physical parameters of habitat.

   1.1 Select and characterize five reaches with vigorous ringed sawback turtle populations.

1.2 Select and characterize five reaches that do not support vigorous ringed sawback turtle populations.

1.3 Compare data obtained in 1.1 and 1.2 to determine potentially limiting factors.

2. Determine reproductive requirements.

2.1 Determine nesting locations and prepare physical description of sites.

2.2 Determine nesting requirements.

2.3 Determine effects of environmental changes and of predation on reproductive success.

2.4 Determine where most of the successful reproduction occurs and the influencing factors.

3. Determine food sources.

3.1 Determine the food requirements at various life stages and seasons.

3.2 Determine physical requirements of the major prey species.

3.3 Determine how distribution and abundance of the major prey species correlates with the vigorous turtle populations.

4. Determine population structure.

4.1 Determine sex ratio, size, and age at maturity, and age structure.

4.2 Estimate number of ringed sawback turtles per mile in each of the study reaches.

5. Determine activity periods and behavior.

5.1 Determine seasonal activity.

5.2 Determine daily activity.

5.3 Determine if the species moves any distance during its lifetime and barriers to such movement, if any.

6. From the information gathered, determine and protect at least two river reaches critical to maintaining a stable population.

6.1 Protect these two river reaches from activities that would cause a decline of this species' population.

6.2 Develop and implement a monitoring plan to evaluate effectiveness of protective measures and to track population trends.

C. Narrative Outline

1. Characterize physical parameters of habitat. This section will seek to compare habitat parameters such as water depth, current, water chemistry, bottom composition, numbers of snags per mile, area of sandbars per mile, bank height, sandbar vegetation, average exposure of sandbars and snags to direct sunlight per day and any other factors that may be applicable, to determine the limiting factors for various reaches of the Pearl River.

1.1 Select and characterize five reaches with vigorous ringed sawback turtle populations. The ringed sawback turtle continues to exist in good numbers in an approximate 50 mile reach upstream of Ratliff's Ferry and in a 120 mile reach from near Georgetown, Mississippi downstream to the vicinity of Sandy Hook, Mississippi. Five reaches of at least 3 miles each will be selected to characterize the parameters in 1. These reaches will be in the vicinity of the Highway 35 bridge at Carthage, below the Highway 25 bridge, and near Monticello, Columbia, and Sandy Hook, Mississippi. Within each selected reach, at least 10 sample stations will be defined to aid in statistical comparisons. Each sample station will be at least 100 yards in length.

1.2 Select and characterize five reaches that do not support vigorous ringed sawback turtle populations. Populations of this turtle have apparently declined and/or were always low in the Pearl River from Ross Barnett Reservoir downstream to near Georgetown and in the Pearl River below Bogalusa. Five reaches of 3 miles each will be selected to characterize the parameters in 1. These reaches will be: between Ross Barnett Reservoir and Lakeland Avenue, Jackson; between the Jackson metropolitan area and Georgetown; downstream of Bogalusa; and near Walkiah Bluff Water Park above Picayune, Mississippi. Sample stations will be designated as in 1.1.

1.3 Compare data obtained in 1.1 and 1.2 to determine limiting factors. Compare the parameters to determine those common to all study areas in 1.1, but lacking in study areas 1.2.

2. Determine reproductive requirements. This section will determine the required and limiting factors to successful reproduction. Two study areas will be selected from those areas in 1.1 with one above and one below Ross Barnett Reservoir.

2.1 Determine nesting locations and prepare physical description of sites. In at least two study areas, determine the characteristics of nest cavity, substrate type, location of nest relative to water, vegetation, and debris, length of exposure per day of nest site to direct sun, temperature of nest substrate and other parameters necessary to determining the suitability of a nesting site.

2.2 Determine nesting requirements. Within the study areas selected in 2.1, determine dates of nesting, period of greatest nesting activity, extreme nesting dates, incubation period, clutch size, frequency of nesting, and description of eggs. Incubation period may be determined by artificial rearing but should be compared to natural incubation where possible. Clutch size can be determined by counting the number of eggs laid, X-ray of gravid females, and dissection of a small number of individuals. Determining frequency of nesting will require extensive mark-recapture and/or X-ray of females immediately after laying.

2.3 Determine effects of environmental changes and of predation on reproductive success. This task will determine the effects of high water on nesting and hatching. The length of time an egg or embryo can be submerged before dying is crucial in predicting certain environmental impacts to population trends. This will be crucial in evaluating the potential impact of a dry dam such as is authorized for the upper Pearl River. Because nest predation appears to be a limiting factor in other species of *Graptemys*, we must know the impact of nest predation on the ringed sawback turtle. This can be accomplished by marking and monitoring nests to determine the percent that successfully hatch and identifying predators by the tracks around destroyed nests.

2.4 Determine where most of the successful reproduction occurs and the influencing factors. This task will seek to determine if nest temperature influences sex determination and, if so, could bank clearing have enough effect on nest temperature to result in a skewed sex ratio. The relationship between female size and clutch size will be determined to evaluate the contribution of large females to population trends. Larger turtles are more likely targets of wanton shooting and in *Graptemys* these often are females. Determining where successful reproduction occurs will permit an evaluation of the importance of habitat quality, predator density, and the interaction of these factors.

3. Determine food sources. This task will seek to determine preferred and available food sources for various life stages of the ringed sawback turtle.

3.1 Determine the food requirements at various life stages and seasons. Juvenile and small male *Graptemys* apparently utilize insects as a food source. Larger males and adult females may also consume mollusks as food. This task will seek to determine the groups or species utilized as food by examining fecal samples from captured turtles and from stomach content analysis of any turtles that may be sacrificed for other purposes or from stomach pumping of live turtles. Stomach contents of *G. pulchra* that occur in the same habitat will be examined to determine what a closely associated species consumes and if there is competition for the food source. Determination of diet will be accomplished for two seasons in the two study areas selected in 2.1 to evaluate seasonal variations and dietary variation between the study areas.

3.2 Determine physical requirements of the major prey species. Once the major prey species have been identified in 3.1, this task will seek to determine those factors that influence their abundance and availability. This will include correlating peak insect population levels with turtle hatchling emergence and the impact of habitat modification on various life stages of the insect prey. The required substrate and reproductive requirements of mollusk prey species will be determined. This may include the determination of molluscan fish host to evaluate project impacts on the turtle's food supply.

3.3 Determine how distribution and abundance of the major prey species correlates with the vigorous turtle populations. Shealy (1976) found a direct correlation between the presence and abundance of mollusks and the presence and abundance of *G. pulchra*. This task will determine if the same is true for *G. oculifera* by examining reaches of the Pearl River for prey organisms relative to *G. oculifera* populations.

4. Determine population structure. This task will develop data to evaluate population trends and possibly identify limiting factors for this species in areas of low population levels.

4.1 Determine sex ratio, size and age at maturity, age structure. Nest temperatures influence sex determination in several other species of *Graptemys* and probably do in *G. oculifera*. If early nesting is lost due to flooding or some other factor, the increased temperatures of late nesting could skew the sex ratio. Determining and comparing all these factors for each of the study areas will provide a base for determining population trends and possibly identify limiting factors for reaches of the Pearl River that have low population levels of this turtle.

4.2 Estimate number of ringed sawback turtles per mile in each of the study reaches. This task will provide baseline data to determine population trends. Basking ringed sawback turtles will be counted on at least three occasions at each sample station each year for three years. Turtles will be captured within selected reaches to provide hands-on verification of the basking counts. Captured turtles will be marked by notching or drilling certain carapace marginals for recapture studies to provide an additional estimate of population size and to provide data on survivorship and growth rates.

5. Determine activity periods and behavior. This task will seek to provide data on activity and behavior by age and sex classes, by seasons, and on daily activity.

5.1 Determine seasonal activity. Studies have shown that feeding activity ceases in the late fall with turtles entering a period of low activity. This task will seek to identify if and when this occurs in *G. oculifera*. Where the turtles go during this period of low activity will be compared by age and sex classes.

5.2 Determine daily activity. Daily activity will be determined by age and sex classes under varying environmental conditions. Shealy (1976) found juveniles of *G. pulchra* to have a strong basking drive and that they would bask under unfavorable environmental conditions. Does this happen with *G. oculifera* and, if so, what impact does it have on the population structure? Peak feeding hours will be determined for juveniles and adults.

5.3 Determine if the species moves any distance during its lifetime and barriers to such movement, if any. This task will determine if river reaches that support low population levels are dependent upon emigration from other river reaches and if there are barriers to such movement. Homing tendencies will be evaluated by relocating some turtles and determining if they return to the point of capture. This task will require considerable mark and recapture work.

6. From the information gathered, determine and protect at least two river reaches critical to maintaining a stable population. Baseline data from this plan will be used to identify two river reaches with a minimum of 30 miles in one reach and totalling at least 150 river miles in the two reaches. Necessary actions to protect and plans to monitor these areas will be developed and implemented. Immediate and interim protection will be provided by the authority of Section 7 of the Endangered Species Act.

6.1 <u>Protect these two river reaches from activities that would cause a decline of this species' population</u>. This task will develop the actions necessary to protect the identified habitat and seek to protect it by implementing those actions. These actions may include anything from regulations or legislation restricting habitat modification in the designated reaches to acquisition of key areas.

6.2 <u>Develop and implement a monitoring plan to evaluate effectiveness of protective measures and to track population trends</u>. Using the baseline data gathered in this plan, develop a monitoring plan to evaluate the protective measures, track population trends, and take corrective action as necessary. Population monitoring including age classes, sex ratio, nesting success, food availability, and number of turtles per mile will be conducted at three year intervals in the protected river reaches.

The same factors will be monitored in the river reaches selected in 1.2 at five year intervals to evaluate trends in the areas of low population.

Literature Cited

Anderson, P.K. 1958. The photic responses and water-approach behavior of hatchling turtles. Copeia 1958:211-215.

Bull, J.J. 1985. Sex ratio and nest temperature in turtles: comparing field and laboratory data. Ecology 66(4):1115-1122.

Cagle, F.R. 1950. The life history of the slider turtle Pseudemys scripta troosti. Ecol. Monog. 20:31-54.

_____. 1952. The status of the turtles Graptemys pulchra Baur and G. barbouri Carr and Marchand, with notes of their life history. Copeia 1952:223-224.

_____. 1953. The status of the turtle Graptemys oculifera (Baur). Zoologica 83:137-144.

_____. 1954. Two new species of the Genus Graptemys. Tulane Stud. in Zool. I(11):165-186.

Cliburn, J.W. 1971. The ranges of four species of Graptemys in Mississippi. J. Mississippi Acad. Sci. 16:16-19.

Ewert, M.A. 1979. The embryo and its egg development and natural history. In: Turtles. Perspectives and research. M. Harless and H. Morlock (eds.). John Wiley & Sons, N.Y. 695 pp.

Lahanas, P.N. 1982. Aspects of the life-history of the southern black-knobbed sawback Graptemys nigrinoda delticola Folkerts and Mount. A thesis submitted to Auburn University. 275 pp.

McCoy, C.J. and R.C. Voyt. 1980. Distribution and population status of the ringed sawback Graptemys oculifera (Baur) in Mississippi and Louisiana. A status survey report for the U.S. Fish and Wildlife Service.

Shealy, R.M. 1976. The natural history of the Alabama map turtle, Graptemys pulchra Baur, in Alabama. Bull. Florida State Mus., Biol. Sci. 21:47-111.

Webb, R.G. 1961. Observations on the life histories of turtles (Genus Pseudemys and Graptemys) in Lake Texoma, Oklahoma. Amer. Midl. Nat. 65:193-214

PART III

KEY TO IMPLEMENTATION SCHEDULE COLUMNS 1 & 4

General Category (Column 1):

Information Gathering - I or R (research)

1. Population status
2. Habitat status
3. Habitat requirements
4. Management techniques
5. Taxonomic studies
6. Demographic studies
7. Propagation
8. Migration
9. Predation
10. Competition
11. Disease
12. Environmental contaminant
13. Reintroduction
14. Other information

Acquisition - A

1. Lease
2. Easement
3. Management agreement
4. Exchange
5. Withdrawal
6. Fee title
7. Other

Other - O

1. Information and education
2. Law enforcement
3. Regulations
4. Administration

Management - M

1. Propagation
2. Reintroduction
3. Habitat maintenance and manipulation
4. Predator and competitor control
5. Depredation control
6. Disease control
7. Other management

Priority (Column 4):

1 - An action that must be taken to prevent extinction or to prevent the species from declining irreversibly in the foreseeable future.

2 - An action that must be taken to prevent a significant decline in species population/habitat quality or some other significant negative impact short of extinction.

3 - All other actions necessary to provide for full recovery of the species.

Implementation Schedule

General Category	Plan Task	Task Number	Priority	Task Duration	Responsible Agency: FWS Region	Responsible Agency: FWS Division	Responsible Agency: Other	Estimated Fiscal Year Costs: FY 1	Estimated Fiscal Year Costs: FY 2	Estimated Fiscal Year Costs: FY 3	Comments/Notes
I3	Characterize physical parameters of habitat	1	2	1 yr.	4	FWE	MSDWC COE	5,000			All funding estimates are for FWS funds only.
I3	Determine nesting locations and physical description of sites	2.1	2	2	4	FWE	MSDWC COE	5,000	5,000		
I3	Determine nesting requirements and parameters	2.2	2	1	4	FWE	MSDWC COE		5,000		
I3	Determine effects of environmental changes and of predation on reproductive success	2.3	2	3	4	FWE	MSDWC COE		5,000	5,000	
I3	Determine where most of the successful reproduction occurs and the influencing factors	2.4	2	1	4	FWE	MSDWC COE			5,000	
I3,5	Determine the food organisms at various life stages and seasons	3.1	2	2	4	FWE	MSDWC COE	5,000	5,000		
I3	Determine physical requirements of the major prey species	3.2	3	1	4	FWE	MSDWC COE		5,000		
I1	Determine how distribution and abundance of major prey species correlates with vigorous turtle populations	3.3	3	1	4	FWE	MSDWC COE			5,000	

Implementation Schedule

General Category	Plan Task	Task Number	Priority	Task Duration	Responsible Agency FWS Region	Responsible Agency FWS Division	Responsible Agency Other	Estimated Fiscal Year Costs FY 1	Estimated Fiscal Year Costs FY 2	Estimated Fiscal Year Costs FY 3	Comments/Notes
I1,6	Determine sex ratio, size and age at maturity, age structure, survival rates by sex and age classes	4.1	2	3	4	FWE	MSDWC COE	8,000	8,000	8,000	
I1	Estimate number of ringed sawback turtles in each of study reaches	4.2	2	3	4	FWE	MSDWC COE	5,000	5,000	5,000	
I6,8 14	Determine activity periods and behavior	5	3	1	4	FWE	MSDWC COE			5,000	
A3,03	Protect two river reaches	6.1	3	continu-ous	4	FWE LE	MSDWC COE	1,000	1,000	1,000	
04	Develop and implement a monitoring plan	6.2	3	continu-ous	4	FWE	MSDWC COE	2,000	2,000	2,000	

FWE = Division of Endangered Species, Fish and Wildlife Enhancement
LE = Law Enforcement
MSDWC = Mississippi Dept. of Wildlife Conservation
COE = U.S. Army Corps of Engineers

Species Status Assessment Report
for the
Northern long-eared bat
(*Myotis septentrionalis*)

Version 1.2

Photo by: Jill Utrup, USFWS

August 2022
U.S. Fish and Wildlife Service
Great Lakes Region
Bloomington, MN

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[1] This SSA was developed in tandem with SSA analyses and reports for the tricolored bat (TCB; *Perimyotis subflavus*) and little brown bat (LBB; *Myotis lucifugus*).

EXECUTIVE SUMMARY

This report summarizes the results of a species status assessment (SSA) that assessed the northern long-eared bat's (NLEB; *Myotis septentrionalis*) viability over time. Although this SSA is its own separate report, it was developed in tandem with SSA analyses and reports for the tricolored bat (*Perimyotis subflavus*) and the little brown bat (*Myotis lucifugus*).

NLEB, a wide-ranging bat species, found in 37 states and 8 provinces in North America, typically overwinters in caves or mines and spends the remainder of the year in forested habitats. As its name suggests, NLEB is distinguished by its long ears, particularly as compared to other bats in genus *Myotis*.

In conducting our status assessment, we first considered what NLEB needs to ensure viability. We then considered factors that are currently influencing those viability needs or expected to in the future. Based on the species' viability needs and current influences on those needs, we evaluated NLEB's current condition. Lastly, we projected plausible future scenarios for NLEB based on its current condition and expected future influences on viability.

For survival and reproduction at the individual level, the NLEB requires access to food and water resources when not hibernating, along with suitable habitat throughout its annual life cycle. During the spring, summer and fall seasons, NLEB requires suitable foraging, roosting, traveling (between summer and winter habitat) and swarming habitat with appropriate conditions for maternity colony members; during the winter, NLEB requires habitat with suitable conditions for prolonged bouts of torpor. For NLEB populations to be healthy, they require a population size and growth rate sufficient to withstand natural environmental fluctuations, habitat of sufficient quantity and quality to support all life stages, gene flow among populations, and a matrix of interconnected habitats that support spring migration, summer maternity colony formation, fall swarming, and winter hibernation.

At the species level, NLEB viability requires having a sufficient number and distribution of healthy populations to ensure NLEB can withstand annual environmental and demographic variation (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). Resiliency is best measured by the number, distribution, and health of populations across the species' range. Redundancy can be measured through the duplication and distribution of resilient populations across the species' range relative to potential catastrophic events. Representation can be measured by the number and distribution of healthy populations across areas of unique adaptive diversity. For NLEB, we identified five representation units (RPUs): Eastern Hardwoods, Southeast, Midwest, Subarctic, and East Coast.

Although there are countless stressors affecting NLEB, the primary factor influencing the viability of the NLEB is white-nose syndrome (WNS), a disease of bats caused by a fungal pathogen. Other primary factors that influence NLEB's viability include: wind energy mortality, effects from climate change, and habitat loss.

- WNS has been the foremost stressor on NLEB for more than a decade. The fungus that causes the disease, *Pseudogymnoascus destructans* (*Pd*), invades the skin of bats and infection leads to increases in the frequency and duration of arousals during hibernation and eventual depletion of fat reserves needed to survive winter, and often results in mortality. WNS has caused estimated NLEB population declines of 97–100% across 79% of the species' range.

- Wind energy-related mortality of NLEB, is also proving to be a consequential stressor at local and regional levels, especially in combination with impacts from WNS. Most bat mortality at wind energy projects is caused by direct collisions with moving turbine blades. Wind energy mortality may occur over 49% of the NLEB range.
- Climate change variables, such as changes in temperature and precipitation, may influence NLEB resource needs, such as suitable roosting habitat for all seasons, foraging habitat, and prey availability. Although there may be some benefit to NLEB from a changing climate, overall negative impacts are anticipated, especially at local levels.
- Habitat loss may include loss of suitable roosting or foraging habitat, resulting in longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality. Loss of or modification of winter roosts (i.e., making hibernaculum no longer suitable) can result in impacts to individuals or at the population level.

In evaluating current and future conditions of the NLEB, we used the best available data. Winter hibernacula counts provide the most consistent, long-term, reliable trend data, and provide the most direct measure of WNS impacts, even for species such as NLEB that may be undercounted (due to their proclivity to roost in crevices). Although the availability and quality of summer data vary temporally and spatially, this data offered additional support (to winter data results) in evaluating population trends since *Pd* arrival. We relied upon the data derived from North American Bat Monitoring Program (NABat) analyses for all available winter (NABat 2021) and summer data (NABat 2020).

Available evidence, including both winter and summer data, indicates NLEB abundance has and will continue to decline substantially over the next 10 years under current demographic conditions. Winter abundance (from known hibernacula) has declined rangewide (49%) and across most RPUs (0–90%). In addition, the number of extant winter colonies declined rangewide (81%) and across all RPUs (40–88%). By 2030, rangewide abundance declines by 95% and the spatial extent declines by 75%. There has also been a noticeable shift towards smaller colony sizes, with a 96–100% decline in the number of large hibernacula (≥100 individuals). Declining trends in abundance and occurrence are also evident across much of NLEB's summer range. Rangewide summer occupancy declined by 80% from 2010–2019. Data collected from mobile acoustic transects found a 79% decline in rangewide relative abundance from 2009–2019 and summer mist-net captures declined by 43–77% compared to pre-WNS capture rates. To assess NLEB's future viability, we determined how WNS occurrence and wind energy capacity is likely to change into the future. We described two scenarios that bound our uncertainty on WNS spread and wind energy capacity. The first scenario included WNS spread under the Hefley et al. (2020, entire) model and lower wind energy capacity (low impact scenario) and the second scenario included WNS spread under Wiens et al. (2022, pp. 215–248) model and higher wind energy capacity (high impact scenario).

Using these scenarios, we projected the species' abundance and distribution. Under these future scenarios, NLEB declines worsen precipitously. Rangewide abundance declines 95% by 2030 and 99% by 2040. The number of extant winter colonies decline to only 9 (99% decline) by 2030 and 0 by 2050. Colony size also declines, with the number of large hibernacula (≥100 bats) declining 89% between 2020 and 2030. NLEB's winter spatial extent also declines by 75% by 2030 and by 100% by 2060. There are no areas within the species range where similar declines were not observed, with all RPUs experiencing declines in abundance, number of extant winter colonies, and spatial extant. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. We expect that these impacts will result in further reduction in the species' resiliency, representation, and redundancy.

Unquestionably, WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future. As is the case for all species status assessments, we do not have perfect information (see Appendix 1) on NLEB's occurrence, but the best available data suggest that bats at unknown hibernacula will undergo similar declines observed at known winter colonies. Wind energy related mortality is projected to be a more impactful influence in the future as annual mortality is projected to increase between 202 and 2,926 individuals by 2050 under the future low and high build-out scenarios, respectively. Although there may be some offsetting of effects under current climate conditions, increasing negative impacts are anticipated in the future. Increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase, leading to increased NLEB mortality and reduced reproductive success. Although we consider habitat loss pervasive across the NLEB range, impacts to NLEB and its habitat are often realized at the individual or colony level. Also, loss of hibernation sites (or modifications such that the site is no longer suitable) can result in impacts to winter colonies.

In conclusion, multiple data types and analyses indicate downward trends in NLEB population abundance and distribution over the last 14 years and consequently, we found no evidence to suggest that this downward trend will change in the future. NLEB abundance (winter and summer), number of occupied hibernacula, spatial extent, probability of persistence, and summer habitat occupancy across the range and within all RPUs are decreasing. Since the arrival of WNS, NLEB abundance steeply declined. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, NLEB's ability to recover from these low abundances is limited given their low reproduction output (1 pup per year). Therefore, NLEB's resiliency is greatly compromised in its current condition and is projected to decline under future scenarios. Additionally, because NLEB's abundance and spatial extent are projected to decline dramatically, NLEB will also become more vulnerable to catastrophic events. NLEB's representation has also been reduced. The steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reduced NLEB adaptive capacity. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit NLEB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen NLEB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further WNS spread and additional wind energy development, NLEB's viability is likely to rapidly decline over the next 10 years. Further, given the projected low abundance and the few number and restricted distribution of winter colonies, NLEB's currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

Contents

ABBREVIATION AND ACRONYMS:

%Sp – Percent Species Composition
AC – Adaptive Capacity
AEO – Annual Energy Outlook
AWEA – American Wind Energy Association
AWWI – American Wind Wildlife Institute
Bfat – Bat Fatality
BWEC – Bat Wind Energy Association
C – Celsius
CanWEA – Canadian Wind Energy Association
CC – Climate Change
CE – Catastrophic Event
CER – Canadian Energy Regulator
CI – Confidence Interval
CONUS – Continental United States
CWTD – Canada Wind Turbine Database
DFW – Department of Fish and Wildlife
ESA – Endangered Species Act
F – Fahrenheit
GRTS – Generalized Random-Tessellation Stratified
Hibs – Hibernacula
IUCN – International Union for Conservation of Nature
km – Kilometers
LBB – Little brown bat (*Myotis lucifugus*)
MAST – Mean Annual Surface Temperature
mi – Miles
MLRC – Multi-Resolution Land Characteristics
MW – Megawatts
MYLU – *Myotis lucifugus*
MYSE – *Myotis septentrionalis*
N – Abundance
NLCD – National Land Cover Database
NLEB – Northern long-eared bat (*Myotis septentrionalis*)
NPS – National Park Service
NREL – National Renewable Energy Laboratory
Pd – Pseudogymnoascus destructans
PESU – *Perimyotis subflavus*
pPg – Probability of Population Growth
RPA – Resources Planning Act
RPU – Representation Unit
SSA – Species Status Assessment
TCB – Tricolored bat (*Perimyotis subflavus*)
USDOE – U.S. Department of Energy
USEIA – U.S. Energy Information Administration
USFS – U.S. Forest Service

USFWS – U.S. Fish and Wildlife Service
USGS – U.S. Geological Survey
USWTDB – U.S. Wind Turbine Database
WNS – White-Nose Syndrome
YOA – Year of Arrival
YSA – Years since Arrival
λ (Lambda) – Population Growth Rate
λavg – Average Population Growth Rate
λtot – Total Population Growth Rate

CHAPTER 1 – INTRODUCTION

Background

This report summarizes the results of a species status assessment (SSA) conducted for northern long-eared bat (NLEB, *Myotis septentrionalis*). It delivers the best available scientific and commercial information available on the NLEB in a transparent and defensible peer reviewed report for immediate and future Endangered Species Act (ESA) related decisions. Therefore, while the report is not a decisional document, it does serve as a synthesis of the best available information on the biological status, helpful in promoting the current and future conservation of the species. For this reason, after reviewing this document relative to all relevant laws, regulations, and policies, the U.S. Fish and Wildlife Service (USFWS) plans to utilize the results of this report to make and publish a listing determination in the *Federal Register*.

This chapter describes the analytical framework and methods used to assess NLEB's viability over time. Chapter 2 summarizes the ecological requirements for survival and reproduction at the individual, population, and species levels. Chapter 3 summarizes the historical condition of NLEB. Chapter 4 describes the key drivers that led to NLEB's current condition and the anticipated plausible change in the primary drivers (referred to as influences) over time. Chapter 5 summarizes the current condition assuming no change in influences. Chapter 6 describes the species' future conditions given the plausible projections of the key influences. Lastly, Chapter 7 synthesizes the above analyses and describes how the consequent change in the number, health, and distribution of populations influence NLEB viability over time as well as the sources of uncertainty and the implications of this uncertainty. Appendices 1–5 provide further information on uncertainty and sensitivity, supplemental methodology information, supplemental results, supplemental threat background information, and supplemental data.

Analytical Framework

Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308–311). Meaning, to sustain populations over time, a species must have a sufficient number of populations distributed throughout its geographic range to withstand:

(1) environmental and demographic stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

Viability is a measure of the likelihood of sustaining populations over time. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is generally better able to adapt to future changes and to tolerate catastrophes, environmental stochasticity, and stressors, and thus, typically has high viability.

Resiliency is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature, rainfall), periodic disturbances within the normal range of variation (e.g., fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

Resiliency is multi-faceted. First, it requires having healthy populations demographically (robust survival, reproductive, and growth rates), genetically (large effective population size, high heterozygosity, and gene flow between populations), and physically (good body condition). Second, resiliency also requires having healthy populations distributed across heterogeneous environmental conditions (referred to as spatial heterogeneity; this includes factors such as temperature, precipitation, elevation, and aspect). Spatial heterogeneity is particularly important for species prone to spatial synchrony (regionally correlated fluctuations among populations). Populations can fluctuate in synchrony over broad geographical areas (Kindvall 1996, pp. 207, 212; Oliver et al. 2010, pp. 480–482) because environmental stochasticity can operate at regional scales (Hanski and Gilpin 1997, p. 372). Spatial heterogeneity induces asynchronous fluctuations among populations, thereby guarding against concurrent population declines. Lastly, resiliency often requires connectivity among populations to maintain robust population-level heterozygosity via gene flow among populations and to foster demographic rescue following population decline or extinction due to stochastic events.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health (Mangal and Tier 1993, p. 1083). For all species, a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000, pp. 307, 309–310; Groves et al. 2002, p. 506). Reducing the risk of extinction due to a single or series of catastrophic events requires having multiple populations widely distributed across the species' range, with connectivity among groups of locally adapted populations to facilitate demographic rescue following population decline or extinction. Redundancy provides a margin of safety to reduce the risk of losing substantial portions of genetic diversity or the entire species to a single or series of catastrophic events.

Representation is the ability of a species to adapt to both near-term and long-term novel or extraordinary changes in the conditions of its environment, both physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (novel pathogens, competitors, predators, etc.). This ability to adapt to changing and novel conditions-- referred to as adaptive capacity--is essential for viability as environmental conditions are continuously changing (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either 1) moving to new, suitable environments or 2) by altering (via plasticity or genetic change) their physical or behavioral traits (phenotypes) to match the new environmental conditions (Nicotra et al. 2015, p. 1270; Beever et al. 2016, p. 132). Maintaining a species' *ability to disperse* and colonize new environments fosters adaptive capacity by allowing species to move from areas of unsuitable conditions to regions with more favorable conditions. It also fosters adaptive capacity by increasing genetic diversity via gene flow, which is, as discussed below, important for evolutionary adaptation (Hendry et al. 2011, p. 173; Ofori et al. 2017, p. 1). Thus, maintaining natural levels of connectivity among populations is important for preserving a species' adaptive capacity (Nicotra et al. 2015, p. 1272).

Maintaining a species' *ability to adapt* to novel and extraordinary conditions requires preserving the breadth of genetic variation. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either genetic change or plasticity (see Text Box 1.1). For adaptation to occur, whether through plasticity or evolutionary adaptation, there must be genetic variation upon which selection can act (Hendry et al. 2011, pp. 164–165; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 326). Without genetic variation, the species cannot adapt and is more prone to extinction (Spielman et al. 2004, p. 15263; also see Text Box 1.1).

> *Text Box. 1.1.* **Species Adaptation**. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either *genetic change* or *plasticity* (Chevin et al. 2010, p. 2-3; Hendry et al. 2011, p. 162; Nicotra et al. 2015, p.1270). *Genetic change*, referred to as evolutionary adaptation or potential, involves a change in phenotypes via an underlying genetic change (specifically, a change in allele frequency) in response to novel environmental cues (Nicotra et al. 2015, p. 1271; Ofori et al. 2017, p. 2). *Plasticity*, unlike evolutionary adaptation, involves a change in phenotypes (phenotypic plasticity) without undergoing changes in the genetic makeup (Nicotra et al. 2015, p. 1271-1272). Plasticity is an important mechanism for species to adapt both in immediate and future time frames. In the immediate time frame, plasticity directly acts to allow species to persist despite novel changes in the environment. In the longer time frame, plasticity contributes to a species' adaptive capacity by buying time for adaptive evolution to occur through genetic changes (referred to as genetic assimilation, see Ghalambor et al. 2007, p. 395; Nicotra et al. 2015, p. 1271). Not all genetic and plastic induced changes are adaptive; changes must lead to improved fitness to be adaptive (Nicotra et al. 2015, p. 1271-1272). Importantly, however, adaptive traits can vary over space and time; what is adaptive in one location may not be adaptive in another, and similarly, what is adaptive today may not be under future conditions and vice versa (Nicotra et al. 2015, p. 1271-1272). Thus, maintaining the full breadth of variation in both plastic traits and genetic diversity is important for preserving a species' adaptive capacity.

Genetic variation that is adaptive is difficult to identify for a species and represents a significant challenge even when there is genetic information available. To denote variation as 'adaptive' we need to identify which loci are under selection, which traits those loci control, how those traits relate to fitness, and what the species' evolutionary response to selection on those traits will be over time (Hendry et al. 2011, p. 162–163; Lankau et al. 2011, p. 316; Teplitsky et al. 2014, p. 190). Although new genomic techniques are making it easier to obtain this type of information (see Funk et al. 2019), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation. One of the easiest proxies to measure is variation in biological traits (also described as phenotypic variation). Phenotypic variation, which on its own can be a mechanism for adapting to novel changes, can be due to underlying adaptive genetic variation (Crandall et al. 2000, p. 291; Forsman 2014, p. 304; Nicotra et al. 2015, p. 3). A second proxy for adaptive genetic variation is neutral genetic variation, which is usually the type of genetic data first reported in species-specific genetic studies (see Text Box 1.2). A third, and more distant, proxy for adaptive genetic variation is disjunct or peripheral populations (Ruckelhaus et al. 2002, p. 322). These populations can be exposed to the extremes in habitat/ecological/climate conditions and thus harbor unique and potentially adaptive traits. Similarly, populations that occur across steep environmental gradients can be indicators of underlying adaptive genetic diversity because local adaptation is driven by environmental conditions, which are continually changing at different rates and scales (Sgro et al. 2011, pp. 330, 333).

Text Box. 1.2. ***Genetic diversity***. Genetic variation can be partitioned into two types: adaptive and neutral genetic diversity. Both types are important for preserving the adaptive capacity of a species (Moritz 2002, p. 243), but in different ways. Genetic variation under selection underlies traits that are locally adaptive and that determine fitness (Holderegger et al. 2006, pp. 801, 803; Lankau et al. 2011, p. 316); thus, it is the variation that underpins adaptive evolution (Sgro et al. 2011, p. 328). This type of genetic variation is referred to as adaptive genetic diversity and determines the capacity for populations to exhibit an adaptive evolutionary response to changing environmental conditions. Conversely, neutral genetic variation refers to regions of the genome that have no known direct effect on fitness (i.e., selectively neutral) and change over time due to non-deterministic processes like mutation and genetic drift (Sgro et al. 2011, p. 328). Although, by definition, neutral genetic variation is not under selection, it contributes to the adaptive capacity of a species in a couple of ways. First, neutral genetic variation that is statistically neutral in one environment may be under selection--and thus adaptive--in a different environment (Nicotra et al. 2015, p. 1271-1272). Second, neutral markers can allow us to infer evolutionary lineages, which is important because distinct evolutionary lineages may harbor locally adaptive traits (Hendry et al. 2011, p. 167), and hence, serve as an indicator of underlying adaptive genetic variation. Thus, maintaining the full breadth of neutral and adaptive genetic diversity is important for preserving a species' adaptive capacity.

Lastly, preserving a species' adaptive capacity requires maintaining the processes that allow for evolution to occur; namely, natural selection and gene flow (Crandall et al. 2000, pp. 290–291; Zackay 2007, p. 1; Sgro et al. 2011, p. 327). Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population via differential survival or reproduction (Hendry et al. 2011, p. 169). To preserve natural selection as a functional evolutionary force, it is necessary to maintain populations across an array of environments (Shaffer and Stein 2000, p. 308; Hoffmann and Sgro 2011, p. 484; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 332). Gene flow serves as an evolutionary process by introducing new alleles (variant forms of genes) into a population, thereby, increasing the gene pool size (genetic diversity). Maintaining the natural network of genetic connections between populations will foster and preserve the effectiveness of gene flow as an evolutionary process (Crandall et al. 2000, p. 293). Preserving genetic connections among populations along with maintaining large effective population sizes will minimize the loss of genetic variation due to genetic drift (Crandall et al. 2000, p. 293). Maintaining large population abundance also fosters adaptive capacity as the rate of evolutionary adaptation is faster in populations with high diversity, which is correlated with population size (Ofori et al. 2017, p.2).

General Methods

Below we describe our methods for assessing NLEB viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and 5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability and extinction risk under both current and future conditions (Figure 1.1). We briefly explain these steps below and provide further details in Appendix 2. Because of the difficulty of delineating populations, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. Henceforth, the terms populations, winter colonies, and hibernacula are used interchangeably.

As is the case for all species status assessments, we do not have perfect information. Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios and making reasonable assumptions based on the best available data. The key uncertainties and how we addressed these uncertainties are described in Appendix 1.

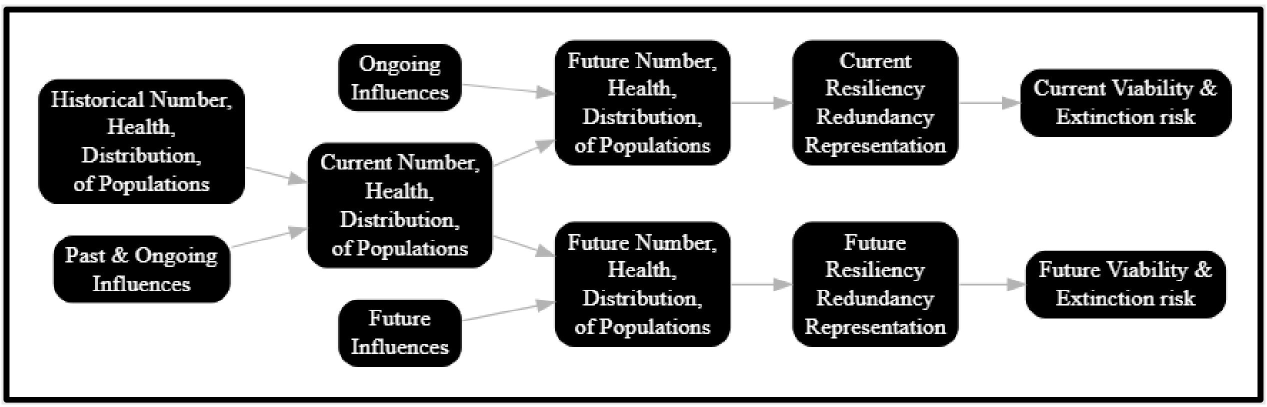

Figure 1.1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time given current and future conditions.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, State and other) across the range to garner all relevant and available data. The majority of these data were collected by State agencies and are now maintained in the North American Bat Monitoring Program (NABat) database, unless otherwise requested by the data contributor or data was not in a format compatible with NABat. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (NABat 2021). Winter counts are conducted as internal surveys of caves, mines, tunnels, culverts and other accessible subterranean habitats. Winter counts are conducted in mid to late winter when bats are expected to be less likely to move between hibernacula and prior to spring emergence. Colony counts in hibernacula provide the best estimate of species abundance consistently available for NLEB. Colony count data represent the largest amount (geographic and in amount of survey) of abundance data throughout the range of the species. Because not all hibernacula are known and accessible, we assume that hibernacula for which data are available are representative of all known and unknown hibernacula for the species. Additionally, to provide a non-model approach, we calculated historical abundances by summing the observed counts within each year. To account for missing data, we applied the last observed count. We refer to this third approach as "constant interpolation."

We measured population health as abundance within hibernacula (*N*) and population trend (λ). To estimate historical *N* and λ, we relied upon analyses completed by Wiens et al. (2022. pp. 231–233). Using a linear mixed effects model (henceforth, status and trends model), Wiens et al. (2022) estimated the yearly population abundance (*N*) from 1990 to 2020. From these yearly abundances, λ was estimated over time for each hibernaculum. For sites with insufficient data points, λ values were applied from the nearest neighbor (see Appendix 2). To capture uncertainty in the year of arrival of *Pseudogymnoascus destructans* (*Pd*), we calculated yearly abundance trajectories under two different *Pd*-occurrence models (Wiens et al. 2022, pp. 226–229 and Hefley et al. 2020, entire). Additionally, to provide a non-model approach, we calculated historical abundances by summing the observed counts within each year. To account for missing data, we applied the last observed count. We refer to this third approach as "constant interpolation."

Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, p. 215–251) as described above. Additionally, because bats occupying a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led analyses of available summer capture records and acoustic records to garner insights on population trends at regional scales (see *Summer Data Analyses* subsection below).

Step 3. Identify the Primary Drivers (Influences)

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) drivers of population numbers. We identified white-nose syndrome (WNS), wind related mortality, habitat loss, and climate change as the primary drivers in NLEB abundance.

We qualitatively assessed the scope, severity, and impact of the four stressors using an approach adapted from Master et al. (2012, pp. 28–35) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. The criteria used to assign levels are shown in Figure 1.2.

SCOPE (% of range)	SEVERITY (% of population decline)			
	Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure 1.2. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012).

For WNS and wind related impacts, we quantitatively modeled the current and future severity of these stressors. We used an existing demographic population model (BatTool, Erickson et al. 2014) to estimate the impacts (severity) from WNS and wind related mortality (described below).

To assess the impact of WNS and wind related mortality into the future, we used published data, expert knowledge, and professional judgment to form plausible future scenarios. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible “lower” and “upper” impact scenarios. The future scenarios are described in Chapter 4.

To calculate the impact of WNS, Wiens et al. (2022, pp. 231–247) derived the yearly effects of WNS, referred to as “WNS impacts schedule” from winter counts at sites upon WNS arrival (see Appendix 2 for further detail). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and thus, we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts to 40 years (i.e., the time throughout which WNS will affect survival in the population). However, to understand the sensitivity of the results to the duration of disease dynamic and to fully capture the uncertainty, we also incorporated a shorter disease dynamic duration. Based on current data (i.e., data from caves documented with WNS in 2008 continue to show continued impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect a population after *Pd* arrives. Thus, our lower impact scenario assumes a 15-year impact duration (i.e., no further WNS impacts after year 15 since *Pd* arrival) and high impact scenario assumes a 40-year impact duration (i.e., the last and least severe WNS disease stage carries through to 2060) (see Appendix 5 for further detail).

To calculate the impact from wind related mortality, we estimated species-specific wind fatality rates as:

$$\text{NLEB } \textit{per MW fat rate} = \textit{Bfat} * \textit{\%Sp}$$

Where *Bfat* is the all-bat fatality rate per megawatt (MW) and *%Sp* is the species-specific percent composition of fatalities reported (see Appendix 2 for further details of how *Bfat* and *%Sp* were calculated).

Step 4. Project the Number, Health, and Distribution of Populations Under Current and Future Influences

To project future abundance and trend given current and future state conditions for WNS and wind, we used the population model, BatTool (updated with NLEB-specific demographic values). In sum, the BatTool projects hibernaculum abundance over time given starting abundance (*N*), trend (λ), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind mortality as specified by the wind capacity scenarios. Starting abundance (*N*) and trend (λ) were derived from Step 2 above. We projected abundance through 2060 to capture the colony response to the 2050 wind energy build-out. Given the species' generation time is 5–7 years, 10 years is sufficient to discern the impacts of the annual mortality levels associated with the 2050 wind capacity build-out.

Using these projected abundance estimates, we calculated various hibernaculum-level and Representation Unit (RPU) metrics to describe the species' historical, current, and future condition (number, health, and distribution of populations) given current and future influences. The results are summarized in chapters 3, 4, and 6. RPUs are further described in Chapter 2.

Step 5. Assess the Current and Future Viability

We evaluated how the change in the number, health, and distribution of populations from historical to present to future influences NLEB's ability to withstand stochastic events, catastrophes, and novel changes in its environment, i.e., the 3Rs over time. Specifically, we used the change in the abundance and distribution of winter colonies over time--to evaluate NLEB's resiliency to stochasticity, disturbances, and stressors. To assess redundancy, we qualitatively assessed how the current and projected abundance and distribution of colonies affect the risk of catastrophic losses due to extreme weather eventsand epizootics.. To assess NLEB's ability to adapt to novel changes in its physical and biological environment, we characterized NLEB adaptability relative to 12 recognized core adaptive capacity attributes (Thurman et al. 2020, entire) and assessed the likelihood of maintaining colonies across the breadth of adaptive diversity given geographic-specific influences and vulnerability to catastrophic events (Appendix 2).

Summary of NABat Data Sources

Our analyses relied on existing information and upon the data and analyses conducted by NABat. Wiens et al. (2022, entire) provided estimates of past, current, and future abundance based on available winter count data (NABat 2021; accessed February 10, 2021). Deeley and Ford (2022, entire), Stratton and Irvine (2022, entire), and Whitby et al. (2022, entire), provided estimates of population trend since *Pd* arrival based on available summer data (NABat 2020; accessed November 18, 2020). Udell et al. (2022, entire) estimated hibernaculum-specific wind energy mortality estimates. How we used these data are briefly described in Table 1.1, with more detail in Appendix 2. A conceptual model of the BatTool is provided in Figure 1.3. Using Wiens et al. (2022, entire) data, we calculated summary statistics at rangewide and RPU scales over time. For ease of reading, we do not cite the source of the data within the text of Chapters 3–7. In several cases, contributed data could not be utilized in these range-wide analyses due to incompatibility with the database structure of NABat or infeasibility of transferring data files, e.g., New York State Department of Environmental Conservation acoustic data. In these cases, we reviewed any data summaries and analyses provided by the contributing partner and assessed them alongside analyses from NABat.

Table 1.1. NABat analyses used in the SSA analyses. Steps refer to the 5 steps of our analytical approach.

Citation	Data/Analyses	Step in Analytical Process	Chapter
Cheng et al. 2021	Impacts of WNS	Step 3: past WNS impacts	Chapter 4
Cheng et al. 2022	Winter colony count analysis	Step 3: past WNS impacts	Chapter 4
Deeley and Ford 2022	Rangewide analysis of summer capture rates from 1999–2019	Step 2 - Current conditions	Chapter 5
Stratton and Irvine 2022	Rangewide change in occupancy from 2010 – 2019 based on summer acoustic & mist-net data	Step 2 - Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Whitby et al. 2022	Rangewide analysis of relative abundance based on summer mobile acoustic data from 2009 – 2019	Step 2 - Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Udell et al. 2022	Estimated wind related bat mortality & allocation to known hibernacula	Step 3. Define future scenarios for wind energy mortality	Chapter 4
Wiens et al. 2022 pp. 231–247	Status & trends linear effects model using winter colony count data	Steps 1 & 2 Historical & current abundance (*N*) and population trend (λ) over time Step 3 past WNS impacts, construct WNS impacts schedule	Chapter 3 Chapters 4, 5
Hefley et al. 2020	*Pd*-occurrence model 2	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future low impact scenario for *Pd*-spread	NA Chapter 4
Wiens et al. 2022, pp. 226–229	*Pd*-occurrence model 1	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future high impact scenario for *Pd*-spread	NA Chapter 4
Wiens et al. 2022, pp. 236–247	Future projections of N via BatTool	Step 4. Project abundance over time	Chapters 5, 6

Figure 1.3. A conceptual diagram showing where the NABat data sources are used in our analytical process.

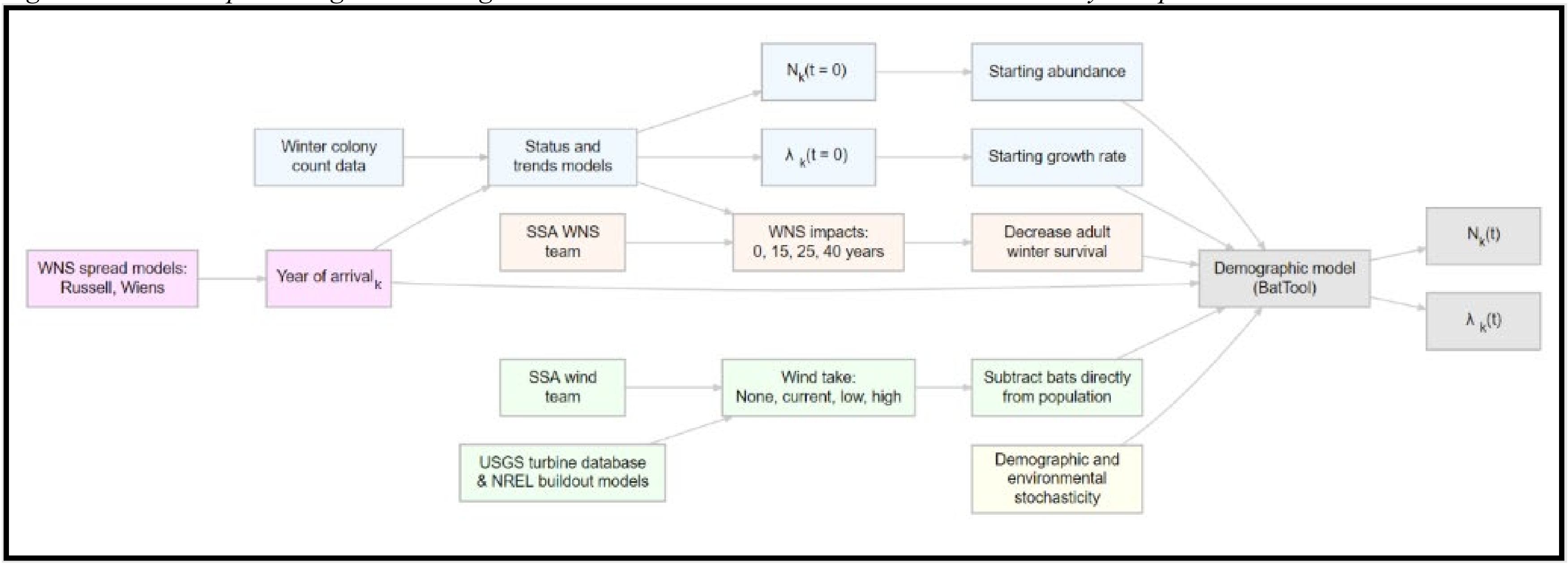

Figure 1.3. BatTool conceptual model. Top (blue boxes): raw data (winter colony) feeds into the status and trends model, which outputs current colony size (N) and population trend (λ) values to input into the BatTool. Middle (pink boxes): 2 Pd occurrence models give Pd year of arrival, which is used in both the status and trends model and BatTool. Middle (peach boxes): SSA core team derived WNS annual impacts schedule, which feeds into the BatTool as decreases in adult winter survival. Bottom (green boxes): SSA core team calculated species-specific bat fatality per MW and USGS projected allocation of this mortality are used to project colony specific mortality over time, which feeds into the BatTool as direct loss of adult females. Far right boxes (gray boxes): projected abundance (N) over time is the output, which is used to calculate colony and RPU level statistics, e.g., λ, number of extant sites, etc.

CHAPTER 2 – SPECIES ECOLOGY AND NEEDS

Taxonomy and Genetics

NLEB belongs to the order *Chiroptera*, family *Vespertilionidae*, subfamily *Vespertilioninae*, genus *Myotis*, and subgenus *Myotis* (Caceres and Barclay 2000, p. 1). The NLEB was first considered a subspecies of Keen's long-eared myotis (*Myotis keenii*) (Fitch and Schump 1979, p. 1), but was recognized as a distinct species by van Zyll de Jong in 1979 (1979, p. 993), based on geographic separation and difference in morphology (as cited in Nagorsen and Brigham 1993, p. 87; Caceres and Pybus 1997 p. 1; Whitaker and Hamilton 1998, p. 99;Caceres and Barclay 2000, p. 1; Simmons 2005, p. 516; Whitaker and Mumford 2009, p. 207), and more recently genetically by Platt et al. (2018, p. 239). The NLEB is currently considered a monotypic species, with no subspecies described for this species (van Zyll de Jong 1985, p. 94; Nagorsen and Brigham 1993, p. 90;; Caceres and Barclay 2000, p. 1; Whitaker and Mumford 2009, p. 214; USFWS 2015, p. 17975).

Although there have been few wide-ranging genetic studies on this species, information collected to date indicates the species to be panmictic (random mating within a population). Johnson et al. (2014, entire) assessed nuclear genetic diversity at one site in New York and several sites in West Virginia, and found little evidence of population structure in NLEBs at watershed or regional scales. In addition, studies conducted in Ohio, Nova Scotia and Quebec, Canada, and Kentucky showed variation in NLEB haplotypes at local levels; however, these studies also indicated relatively low levels of overall genetic differentiation between groups and high levels of diversity overall (Arnold 2007, p. 157, Johnson et al. 2015, p. 12; Olivera-Hyde et al. 2020, p.729).

This species has been recognized by different common names, such as: Keen's bat (Whitaker and Hamilton 1998, p. 99), northern myotis (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207), and the northern bat (Foster and Kurta 1999, p. 660). For purposes of this SSA, we recognize it as a listable entity under the ESA (USFWS 2015, p. 17975).

Species Description

NLEB's adult body weight averages 5 to 8 grams (g) (0.2 to 0.3 ounces), with females tending to be slightly larger than males (Caceres and Pybus 1997, p. 3). Average body length ranges from 77 to 95 millimeters (mm) (3.0 to 3.7 inches [in]), tail length between 35 and 42 mm (1.3 to 1.6 in), forearm length between 34 and 38 mm (1.3 to 1.5 in), and wingspread between 228 and 258 mm (8.9 to 10.2 in) (Barbour and Davis 1969, p. 76; Caceres and Barclay 2000, p. 1). Pelage (fur) colors include medium to dark brown on its back; dark brown, but not black, ears and wing membranes; and tawny to pale-brown fur on the ventral side (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). As indicated by its common name, the NLEB is distinguished from other *Myotis* species by its relatively long ears (average 17 mm (0.7 in); Whitaker and Mumford 2009, p. 207) that, when laid forward, extend beyond the nose up to 5 mm (0.2 in; Caceres and Barclay 2000, p. 1; Figure 2.1). The tragus (projection of skin in front of the external ear) is long (average 9 mm [0.4 in]; Whitaker and Mumford 2009, p. 207), pointed, and symmetrical (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). There is an occasional tendency for the NLEB to exhibit a slight keel on the calcar (spur of cartilage arising from inner side of ankle; Nagorsen and Brigham 1993, p. 87). This can add some uncertainty in distinguishing NLEBs from other sympatric *Myotis* species (Lacki 2013, in litt.). Within its range, the NLEB can be confused with the little brown bat (*Myotis lucifugus*) or the western long-eared myotis (*Myotis evotis*). The NLEB can be distinguished from the little brown bat by its longer ears, tapered and symmetrical tragus, slightly longer tail, and less glossy pelage (Caceres and Barclay 2000, p. 1; Kurta 2013, in litt.). The NLEB can be distinguished from the western long-eared myotis by its darker pelage and paler membranes (Caceres and Barclay 2000, p. 1).

Figure 2.1. Hibernating NLEB. Photo credit: Al Hicks, New York Department of Environmental Conservation (retired).

Species Distribution

NLEB's range includes much of the eastern and north-central U.S., and all Canadian provinces west to the southern Yukon Territory and eastern British Columbia (Nagorsen and Brigham 1993, p. 89; Caceres and Pybus 1997, p. 1; Environment Yukon 2011, p. 10) (Figure 2.2[2]). In the U.S., the species' range reaches from Maine west to Montana, south to eastern Kansas, eastern Oklahoma, Arkansas, and east to South Carolina (Whitaker and Hamilton 1998, p. 99; Caceres and Barclay 2000, p. 2; Simmons 2005, p. 516; Amelon and Burhans 2006, pp. 71–72). The species' range includes all or portions of the following 37 states and the District of Columbia: Alabama, Arkansas, Connecticut, Delaware, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Montana, Nebraska, New Hampshire, New Jersey, New York, North Carolina, North Dakota, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming.

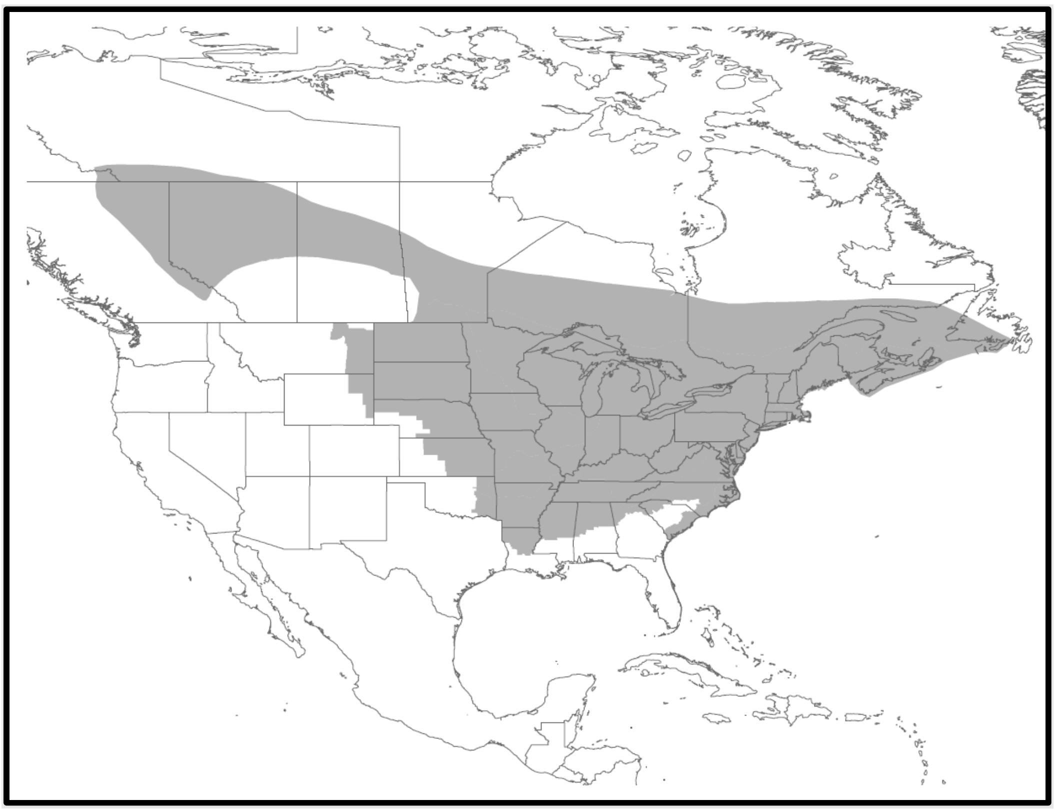

Figure 2.2. Range map for NLEB

[2] The range map was developed using the USFWS's NLEB range map for the U.S. in combination with IUCN's map for Canada (https://www.iucnredlist.org/species/14201/22064312). USFWS maintains a range map using known locations of NLEB. The range boundary is updated as new information is received and can be found here: https://www.fws.gov/Midwest/Endangered/mammals/nleb/nlebRangeMap.html.

Individual Needs and Ecology

Below we describe the life history and ecological needs for NLEB individuals to survive and reproduce; ecological needs are summarized in Table 2.1. The generalized annual life history is summarized for NLEB in Figure 2.3.

Swarming (Fall)

The swarming season occurs between the summer and winter seasons (Lowe 2012, p. 50) and the purpose of swarming behavior may include: introduction of juveniles to potential hibernacula, copulation, and stopping over sites on migratory pathways between summer and winter regions (Kurta et al. 1997, p. 479; Parsons et al. 2003, p. 64; Lowe 2012, p. 51; Randall and Broders 2014, pp. 109–110). During this period, heightened activity and congregation of transient bats around caves and mines is observed, followed later by increased sexual activity and bouts of torpor prior to winter hibernation (Davis and Hitchcock 1965, pp. 304–306; Fenton 1969, p. 601; Parsons et al. 2003, pp. 63–64). For the NLEB, the swarming period may occur between July and early October, depending on latitude within the species' range (Hall and Brenner 1968, p. 780; Fenton 1969, p. 598; Caire et al. 1979, p. 405; Kurta et al. 1997, p. 479; Lowe 2012, p. 86;). The NLEB may investigate several cave or mine openings during the transient portion of the swarming period, and some individuals may use these areas as temporary daytime roosts or may roost in forest habitat adjacent to these sites (Kurta et al. 1997, pp. 479, 483; Lowe 2012, p. 51). Many of the caves and mines associated with swarming are also used as hibernacula for several species of bats, including the NLEB (Fenton 1969, p. 599; Whitaker and Rissler 1992, p. 132; Kurta et al. 1997, p. 484; Glover and Altringham 2008, p. 1498; Randall and Broders 2014, p. 109).

Winter Hibernation

NLEBs are thought to predominantly overwinter in hibernacula that include caves and abandoned mines. These hibernacula have relatively constant, cooler temperatures (0 to 9 degrees Celsius [°C] or 32 to 48 degrees Fahrenheit [°F]) (Raesly and Gates 1987, p. 18; Caceres and Pybus 1997, p. 2; Brack 2007, p. 744), with high humidity and no strong currents (Fitch and Shump 1979, p. 2; van Zyll de Jong 1985, p. 94; Raesly and Gates 1987, p. 118; Caceres and Pybus 1997, p. 2). NLEBs are typically found roosting singly or in small numbers in cave or mine walls or ceilings, often in small crevices or cracks, sometimes with only the nose and ears visible and thus are easily overlooked during surveys (Griffin 1940a, pp. 181–182; Barbour and Davis 1969, p. 77; Caire et al. 1979, p. 405; van Zyll de Jong 1985, p. 9; Caceres and Pybus 1997, p. 2; Whitaker and Mumford 2009, pp. 209–210).

NLEBs have also been observed overwintering in other types of habitat that have similar conditions (e.g., temperature, humidity levels, air flow) to cave or mine hibernacula. The species may use these alternate hibernacula in areas where caves or mines are not present (Griffin 1945, p. 22). NLEBs have been found using the following alternative hibernacula: abandoned railroad tunnels (USFWS 2015, p. 17977), the entrance of a storm sewer in central Minnesota (Goehring 1954, p. 435), a hydroelectric dam facility in Michigan (Kurta et al. 1997, p. 478), an aqueduct in

Massachusetts (Massachusetts Department of Fish and Game 2012, unpublished data), a dry well in Massachusetts (Griffin 1945, p. 22). More recently, NLEBs were found in a crawl space within a dwelling in Massachusetts (Dowling and O'Dell 2018, p. 376) and a rock crevice in Nebraska (White et al. 2020, p. 114). Further, Girder et al. (2016, p. 11) found NLEB to be present and active year round on the coastal plain of North Carolina, where there is no known non-cavernicolous (cave-like) hibernacula; therefore, it is possible this population was not (traditionally) hibernating. Also, in coastal North Carolina, NLEB were observed to be active the majority of the winter, and although torpor was observed, time spent in torpor was very short with the longest torpor bout (i.e., hibernation period) for each bat averaging 6.8 days (Jordan 2020, p. 672).

Summer Roosting

Roosting habitat–NLEBs typically roost singly or in maternity colonies underneath bark or more often in cavities or crevices of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 662; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 222; Timpone et al. 2010, p. 119). Males' and non-reproductive females' summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, p. 77; Amelon and Burhans 2006, p. 72). Studies have documented the NLEB's selection of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 668; Lacki and Schwierjohann 2001, p. 484; Menzel et al. 2002, p. 107; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 224; Timpone et al. 2010, p. 118). NLEBs are flexible in tree species selection and while they may select for certain tree species regionally, likely are not dependent on certain species of trees for roosts throughout their range; rather, many tree species that form suitable cavities or retain bark will be used by the bats opportunistically (Foster and Kurta 1999, p. 668; Silvis et al. 2016, p. 12; Hyzy 2020, p. 62). Carter and Feldhamer (2005, p. 265) hypothesized that structural complexity of habitat or available roosting resources are more important factors than the actual tree species. Further, Silvis et al. (2012, p. 7) found forest successional patterns, stand and tree structure to be more crucial than tree species in creating and maintaining suitable long-term roosting opportunities. To a lesser extent, NLEBs have also been observed roosting in colonies in human-made structures, such as in buildings, in barns, on utility poles, behind window shutters, in bridges, and in bat houses (Mumford and Cope 1964, p. 72; Barbour and Davis 1969, p. 77; Cope and Humphrey 1972, p. 9; Burke 1999, pp. 77–78; Sparks et al. 2004, p. 94; Amelon and Burhans 2006, p. 72; Whitaker and Mumford 2009, p. 209; Timpone et al. 2010, p. 119; Bohrman and Fecske 2013, pp. 37, 74; ; Feldhamer et al. 2003, p. 109; Sasse et al. 2014, p. 172; USFWS 2015, p. 17984; Dowling and O'Dell 2018, p. 376). It has been hypothesized that use of human-made structures may occur in areas with fewer suitable roost trees (Henderson and Broders 2008, p. 960; Dowling and O'Dell 2018, p. 376). In north-central West Virginia, NLEBs were found to more readily use artificial roosts as distance from large forests (greater than 200 hectares [494 acres]) increased, suggesting that artificial roosts are less likely to be selected when there is greater availability of suitable roost trees (De La Cruz et al. 2018, p. 496).

Roosting behavior–Maternity colonies, consisting of females and young, are generally small, numbering from about 30 (Whitaker and Mumford 2009, p. 212) to 60 individuals (Caceres and Barclay 2000, p. 3); however, larger colonies of up to 100 adult females have been observed (Whitaker and Mumford 2009, p. 212). Most studies have found that the number of individuals roosting together in a given roost typically decreases from pregnancy to post-lactation (Foster and Kurta 1999, p. 667; Lacki and Schwierjohann 2001, p. 485; Garroway and Broders 2007, p. 962; Perry and Thill 2007, p. 224; Johnson et al. 2012, p. 227). NLEBs exhibit fission-fusion behavior (Garroway and Broders 2007, p. 961), where members frequently coalesce to form a group (fusion), but composition of the group is in flux, with individuals frequently departing to be solitary or to form smaller groups (fission) before returning to the main spatially discrete unit or network (Barclay and Kurta 2007, p. 44). As part of this behavior, NLEBs switch tree roosts often (Sasse and Pekins 1996, p. 95), typically every 2 to 3 days (Foster and Kurta 1999, p. 665; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 261; Timpone et al. 2010, p. 119). Patriquin et al. (2016, p. 55) found that NLEB roost switching and use varies regionally in response to differences in ambient conditions (e.g., precipitation, temperature). Adult females give birth to a single pup (Barbour and Davis 1969, p. 104). Birthing within the colony tends to be synchronous, with the majority of births occurring around the same time (Krochmal and Sparks 2007, p. 654). Parturition (birth) may occur as early as late May or early June (Easterla 1968, p. 770; Caire et al. 1979, p. 406; Whitaker and Mumford 2009, p. 213) and may occur as late as mid-July (Whitaker and Mumford 2009, p. 213). Juvenile volancy (flight) often occurs by 21 days after birth (Kunz 1971, p. 480; Krochmal and Sparks 2007, p. 651) and has been documented as early as 18 days after birth (Krochmal and Sparks 2007, p. 651).

Foraging (Spring, Summer, Fall)

Diet–NLEBs are nocturnal foragers and use hawking (catching insects in flight) and gleaning (picking insects from surfaces) behaviors in conjunction with passive acoustic cues (Nagorsen and Brigham 1993, p. 88; Ratcliffe and Dawson 2003, p. 851). The NLEB has a diverse diet including moths, flies, leafhoppers, caddisflies, and beetles (Griffith and Gates 1985, p. 452; Nagorsen and Brigham 1993, p. 88; Brack and Whitaker 2001, p. 207), with diet composition differing geographically and seasonally (Brack and Whitaker 2001, p. 208). The most common insects found in the diets of NLEBs are lepidopterans (moths) and coleopterans (beetles) (Brack and Whitaker 2001, p. 207; Lee and McCracken 2004, pp. 595–596; Feldhamer et al. 2009, p. 45; Dodd et al. 2012, p. 1122), with arachnids also being a common prey item (Feldhamer et al. 2009, p. 45).

Foraging behavior–Most foraging occurs above the understory, 1 to 3 m (3 to 10 ft) above the ground, but under the canopy (Nagorsen and Brigham 1993, p. 88) on forested hillsides and ridges, rather than along riparian areas (LaVal et al. 1977, p. 594; Brack and Whitaker 2001, p. 207). This coincides with data indicating that mature forests are an important habitat type for foraging NLEBs (Caceres and Pybus 1997, p. 2; White et al. 2017, p. 8). Foraging also takes place over small forest clearings and water, and along roads (van Zyll de Jong 1985, p. 94). NLEB seem to prefer intact mixed-type forests with small gaps (i.e., forest trails, small roads, or forest-covered creeks) in forest with sparse or medium vegetation for forage and travel rather than fragmented habitat or areas that have been clear cut (USFWS 2015, p. 17992). Foraging patterns indicate a peak activity period within 5 hours after sunset followed by a secondary peak within 8 hours after sunset (Kunz 1973, pp. 18–19). Brack and Whitaker (2001, p. 207) did not find significant differences in the overall diet of NLEBs between morning (3 a.m. to dawn) and evening (dusk to midnight) feedings; however there were some differences in the consumption of particular prey orders between morning and evening feedings. Additionally, no significant differences existed in dietary diversity values between age classes or sex groups (Brack and Whitaker 2001, p. 208).

Staging (Spring)

Spring staging for the NLEB is the time period between winter hibernation and spring migration to summer habitat (Whitaker and Hamilton 1998, p. 80). During this time, bats begin to gradually emerge from hibernation, exit the hibernacula to feed, but re-enter the same or alternative hibernacula to resume daily bouts of torpor (state of mental or physical inactivity) (Whitaker and Hamilton 1998, p. 80). The staging period for the NLEB is likely short in duration (Caire et al. 1979, p. 405; Whitaker and Hamilton 1998, p. 80). In Missouri, Caire et al. (1979, p. 405) found that NLEBs moved into the staging period in mid-March through early May. Sasse et al. (2014, p. 172) found pregnant NLEB using a mine in late April and May in Arkansas. In Michigan, Kurta et al. (1997, p. 478) determined that by early May, two-thirds of the *Myotis* species, including the NLEB, had dispersed to summer habitat. Variation in timing (onset and duration) of staging for Indiana bats (*Myotis sodalis*) was based on latitude and weather (USFWS 2007, pp. 39–40, 42); similarly, timing of staging for NLEBs is likely based on these same factors.

Migration (Spring and Fall)

While information is lacking, short regional migratory movements between seasonal habitats (summer roosts and winter hibernacula) of 56 kilometer (km) (35 mi) to 89 km (55 mi) have been documented (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88). The spring migration period typically runs from mid-March to mid-May (Easterla 1968, p. 770; Caire et al. 1979, p. 404; Whitaker and Mumford 2009, p. 207); fall migration typically occurs between mid-August and mid-October.

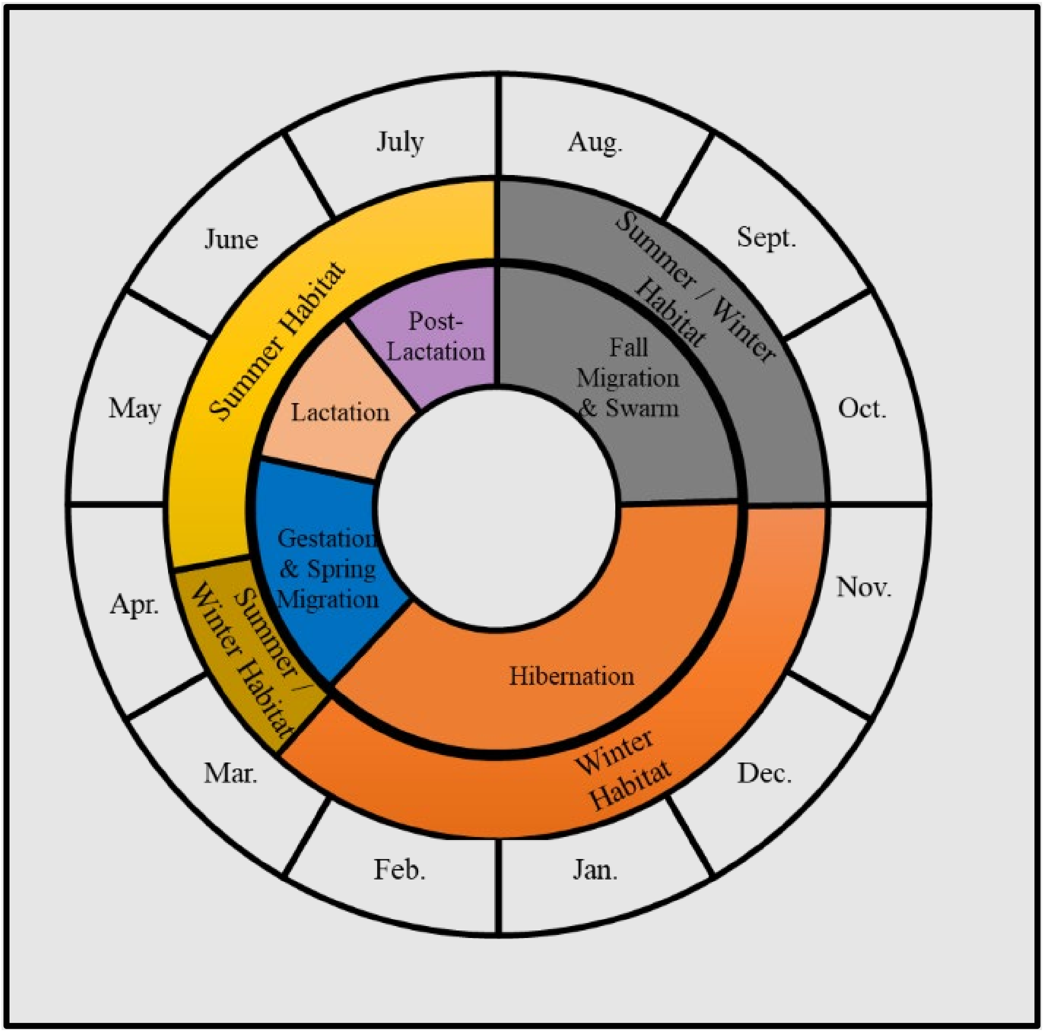

Figure 2.3. Generalized annual life history diagram for NLEB (adapted from Silvis et al. 2016, p. 1).

Table 2.1. The ecological requisites for survival and reproductive success of individuals.

LIFE STAGE	SEASON			
	Spring	**Summer**	**Fall**	**Winter**
Pups		Roosting habitat with suitable conditions for lactating females, and for pups to stay warm and protected from predators while adults are foraging.		
Juveniles		Other maternity colony members (colony dynamics, thermoregulation); Suitable roosting and foraging habitat near abundant food and water resources.	Suitable roosting and foraging habitat near abundant food and water resources.	Habitat with suitable conditions for prolonged bouts of torpor and shortened periods of arousal.
All Adults	Suitable roosting and foraging habitat near abundant food and water resources. Habitat connectivity and open air space for safe migration between winter and summer habitats.	Summer roosts and foraging habitat near abundant food and water resources.	Suitable roosting and foraging habitat near abundant food and water resources; Cave and/or mine entrances (or other similar locations, e.g., culvert, tunnel) for conspecifics to swarm and mate; Habitat connectivity and open air space for safe migration between winter and summer habitats.	Habitat with suitable conditions for prolonged bouts of torpor and shortened periods of arousal.
Reproductive Females		Other maternity colony members (colony dynamics); Network of suitable roosts (i.e.,		

LIFE STAGE	SEASON			
		multiple summer roosts in close proximity) near conspecifics and foraging habitat near abundant food and water resources.		

Population-level Needs

To be self-sustaining, a population must be demographically, genetically, and physically healthy (see Redford et al. 2011, entire). Demographically healthy means having robust survival, reproductive, and growth rates. Genetically healthy populations have large effective population sizes (*Ne*), high heterozygosity, and gene flow between populations. Physically healthy means individuals have good body condition. The population-level ecological requirements of a healthy NLEB population are discussed further below and summarized in (Figure 2.4 and Table 2.2).

Similar to other temperate bat species, NLEB hibernation conditions, prey availability, summer roosting habitat, and connectivity between habitats influence population growth rates and reproduction rates (Figure 2.4). For NLEB populations to be demographically healthy, their growth rate (lambda, or λ) must be sufficient to withstand natural environmental fluctuations. For a population to remain stable (or increasing) over time, λ must be greater than or equal to one. Although variations to summer and winter habitat conditions may result in lower demographic health of a population, NLEB does not generally experience extreme variation in demographics year-to-year due to their selection of summer and winter habitat with narrow microclimate conditions (see *Individual-Level Ecology and Requirements*). During favorable hibernation and summer habitat conditions, NLEB survival and therefore reproductive rates are greater (increasing λ); conversely, when environmental conditions are unfavorable, survival and reproductive rates are lower (decreasing λ).

To support a strong growth rate, NLEB populations benefit from large population sizes and sufficient quality and quantity of habitat to accommodate all life stages. Large effective population size is crucial in maintaining genetic health along with and withstanding environmental variability. Habitat requirements for NLEB are described under *Individual-level Ecology and Needs*. The necessary quantity of habitat is likely to vary among populations, but will likely hinge on the availability of roosting habitat in the summer and suitable hibernacula in the winter. Research has found the minimum summer roost area (i.e., area encompassing all known roost locations) for individual female NLEB ranges between 5.4 hectares and 26 hectares (13 acres and 65 acres), but most studies found the summer roost area to be leaning toward the smaller end of the range (Owen et al. 2003, p. 353; Broders et al. 2006, p. 1177; Badin 2014, p. 75).

To support all life stages, NLEB populations require a matrix of interconnected habitats that support spring migration, summer maternity colony formation and foraging, fall swarming, and winter hibernation. For these populations, movement among habitats is needed to maintain genetic diversity and to allow recolonization in the event of local extirpation. NLEB may migrate short distances between seasonal habitats (summer roosts and winter hibernacula) between 56 km (35 mi) and 89 km (55 mi), as previously mentioned (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88). There is evidence that NLEBs have an affinity for less fragmented habitat (interior forest) (Broders et al,. 2006, p. 1181; Henderson et al. 2008, p. 1825). Therefore, increased fragmentation may negatively impact connectivity between summer and winter habitat and between roosting and foraging habitat.

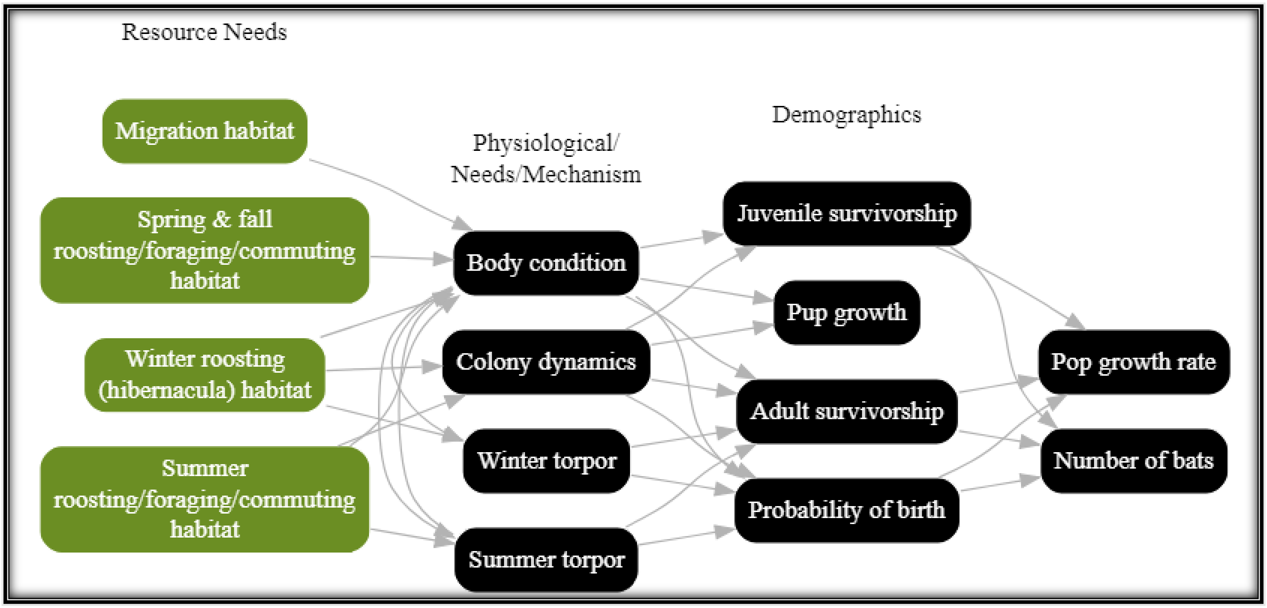

Figure 2.4. Conceptual model showing the connections between resource needs and the physiological needs and demographic rates of a NLEB population (population-level resiliency).

Table 2.2. Population level requirements for a healthy population.

Parameter	**Requirements**
Population growth rate, λ	At a minimum, λ must be ≥ 1 for a population to remain stable over time.
Population size, N	Sufficiently large N to allow for essential colony dynamics and to be resilient to environmental fluctuations.
Winter roosting habitat	Safe and stable winter roosting sites with suitable microclimates.
Migration habitat	Safe space to migrate between spring/fall habitat and winter roost sites.
Spring and fall roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support bats as they exit hibernation (lowest body condition) or as they enter into hibernation (need to put on body fat).
Summer roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support maternity colonies.

Species-level Needs

The ecological requisites at the species level include having a sufficient number and distribution of healthy populations to ensure NLEB can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). We describe NLEB's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

NLEB's ability to withstand stochastic events requires maintaining healthy populations across spatially heterogeneous conditions. Healthy populations-- demographically, genetically, or physically robust--are more likely to withstand and recover from environmental and demographic variability and stochastic perturbations. The greater the number of healthy populations, the more likely NLEB will withstand perturbations and natural variation, and hence, have greater resiliency. Additionally, occupying a diversity of environmental conditions and being widely distributed helps guard against populations fluctuating in synchrony (i.e., being exposed to adverse conditions concurrently). Asynchronous dynamics among populations minimizes the chances of concurrent losses, and thus, provides species' resiliency. Lastly, maintaining the natural patterns and levels of connectivity between populations also contributes to NLEB resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

NLEB's ability to withstand catastrophic events requires having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events. In addition to guarding against population extirpation, redundancy is important to protect against losses in NLEB's adaptive capacity. Multiple, widely distributed populations within areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events, such as extreme winter events, epizootics, and hurricanes.

Representation

NLEB's ability to withstand ongoing and future novel changes is influenced by its capacity to adapt (referred to as adaptive capacity). NLEB may adapt to novel changes by either moving to new, suitable environments or by altering (via plasticity or genetic change) its physical or behavioral traits to match the new environmental conditions. There are multiple intrinsic factors that limit the species ability to adapt to a rapidly changing environment (see Appendix 2-B). Below we describe NLEB's ability to colonize new areas and to alter its physical traits.

NLEB's capacity to colonize new areas (or track suitable conditions) is a function of its physical capability and behavioral tendencies to disperse. NLEB exhibits capabilities (e.g., flight) and behavior (e.g., fission-fusion) that allows for colonization of new areas. NLEB switch summer roosts for a variety of reasons, including temperature, precipitation, predation, parasitism, sociality, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264; Patriquin et al. 2016, p. 55). In addition, although to a lesser extent, NLEB has been found using human-made structures for summer roosts (see *Individual-level Ecology and Needs*). It has been suggested that use of human-made structures may occur in areas with fewer suitable roost trees or lower proximity to larger patches of habitat (Henderson and Broders 2008, p. 960; De La Cruz et al. 2018, p. 496; Dowling and O'Dell 2018, p. 376). Therefore, NLEB has the ability to inhabit new summer roosting habitat at the local level provided that suitable habitat (see *Individual-level Ecology and Needs)* is in close proximity. However, the species may lack the capacity for rapid, large shifts in response to broad-scale novel changes to summer habitat. Maintaining suitable habitat within local home-ranges and beyond is needed to allow for any capacity to shift their range to track suitable conditions. With regard to NLEB's ability to colonize new winter hibernacula, although the species is capable of arousing from torpor and moving between hibernacula during the winter (Griffin 1940a, p. 185; Whitaker and Rissler 1992, p. 131; Caceres and Barclay 2000, pp. 2–3), arousal and movement come at a high energetic cost (Thomas and Geiser 1997, p. 585). NLEB's high degree of site fidelity for a hibernaculum (Pearson 1962, p. 30) also limit their capabilities to inhabit new hibernacula at a broad-scale.

NLEB's capacity to alter its physical or behavioral traits (phenotypes) to match the new environmental conditions is driven by the breadth of adaptive genetic variation. Thus, maintaining populations across the breadth of variation preserves NLEB's capacity to adapt to ongoing and future changes. In addition to preserving the breadth of variation, it is also necessary to maintain the key evolutionary processes through which adaptation occurs, namely, natural selection, gene flow, and genetic drift. Maintaining healthy NLEB populations across a diversity of environments and climatic conditions as well as keeping natural networks of genetic connections between populations allows for such adaptation, via natural selection or gene flow; and preserving large effective population abundances, ensures genetic drift does not act unduly upon the species (see Chapter 1 for further explanation).

For reasons explained in Chapter 1, we rely on proxies to identify species' adaptive genetic variation. We identified and delineated the genetic variation across NLEB's range into geographical representation units using the following proxies: variation in biological traits, neutral genetic diversity, peripheral populations, habitat niche diversity, and steep environmental gradients. These representation units (RPUs) are described below and displayed in Figure 2.5. Bailey's Eco-Divisions (Bailey 2016, entire) were overlayed on these proxies to identify approximate boundaries due to the associated climatic differences (i.e., precipitation levels, patterns and temperatures) that may be influential in driving the species' adaptive ability. By establishing these RPUs (a combination of proxies and Bailey's Eco-Divisions) the underlying adaptive variation of NLEB (at a broad scale) is preserved.

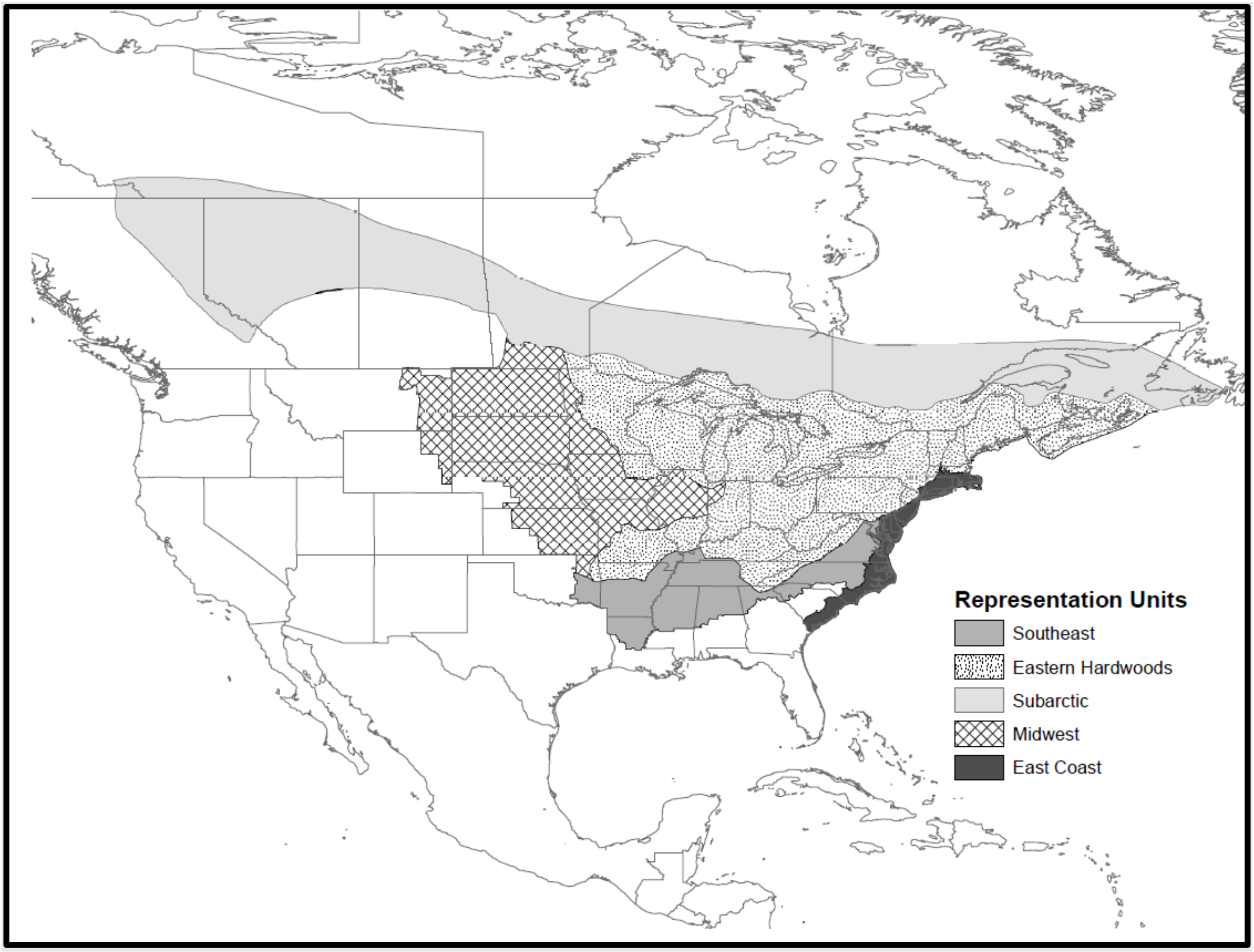

Figure 2.5. Range of NLEB organized into five Representation Units.

1. *Southeast RPU:* In general, NLEB have shorter hibernation periods in this unit (in comparison to the Eastern hardwoods and Subarctic units). Hibernation period correlates with average minimum temperatures and other climatic features, and thus, we used the minimum average temperature zones, specifically zones 6 and 7 in combination with Bailey's Ecoregions "Hot continental" and "Subtropical" divisions to circumscribe variation in hibernation periods.

2. *Eastern Hardwoods RPU:* The Eastern hardwoods Unit was established based on differences in hibernation duration and landcover. NLEB have longer hibernation periods in the Eastern hardwoods unit (in comparison with the Southeast unit). The northern border of this unit was separated from the Subarctic unit based on minimum average temperature zone lines, specifically zones 2 and 3 in combination with Bailey's Ecoregions "Warm continental" and "Subarctic" divisions.

3. *Subarctic RPU:* The Subarctic unit was established based on assumed longer hibernation periods relative to the Eastern hardwoods and Southeastern units. Unlike for the Eastern hardwoods and Southeast units, data on hibernation duration is lacking for the Subarctic unit. However, given hibernation is influenced by minimum winter temperatures, we

assume longer hibernation periods in northern portions of the species' range. The line that was established between the Eastern hardwoods and Subarctic units is described above under the Eastern hardwoods unit description.

4. *Midwest RPU:* The Midwest unit was established based primarily on markedly different landcover than other units, with limited or fragmented forested habitat prevailing throughout much of this unit. Unlike the other units, the Midwest Unit is largely non-forested landcover (e.g., grassland/pasture, cultivated crops, and pasture/hay; Appendix 4-D, NLCD 2016).

5. *East Coast RPU:* The Coastal unit was established based on observed NLEB atypical behavior (e.g., year-round activity, use of non-cavernicolous hibernacula). Southern coastal populations have been observed with similar activity levels year-round in areas with no known nearby traditional hibernacula (i.e., caves or mines; Girder et al. 2016, p. 11; Jordan 2020, p. 672). Further, northern coastal populations have been observed using alternate summer roosting habitat (e.g., human dwellings) and non-cavernicolous hibernacula (e.g., house crawl spaces, Dowling and O'Dell 2018, p. 376).

Table 2.3. Species-level ecology: Requisites for long-term viability (ability to maintain self-sustaining populations over a biologically meaningful timeframe).

3 Rs	**Requisites Long-Term Viability**	**Description**
Resiliency (populations able to withstand stochastic events)	Demographic, physically, and genetically healthy populations across a diversity of environmental conditions	Self-sustaining populations are demographically, genetically, and physiologically robust, have sufficient quantity of suitable habitat
Redundancy (number & distribution of populations to withstand catastrophic events)	Multiple and sufficient distribution of populations within areas of unique variation, i.e., Representation units	Sufficient number and distribution to guard against population losses and losses in species adaptive diversity, i.e., reduce covariance among populations; spread out geographically but also ecologically
Representation (genetic & ecological diversity to maintain adaptive potential)	Maintain adaptive diversity of the species	Populations maintained across breadth of behavioral, physiological, ecological, and environment diversity
	Maintain evolutionary processes	Maintain evolutionary drivers--gene flow, natural selection--to mimic historical patterns

CHAPTER 3 – HISTORICAL CONDITION

This chapter describes the number, health, and distribution of NLEB populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in NLEB viability over time (Figure 3.1).

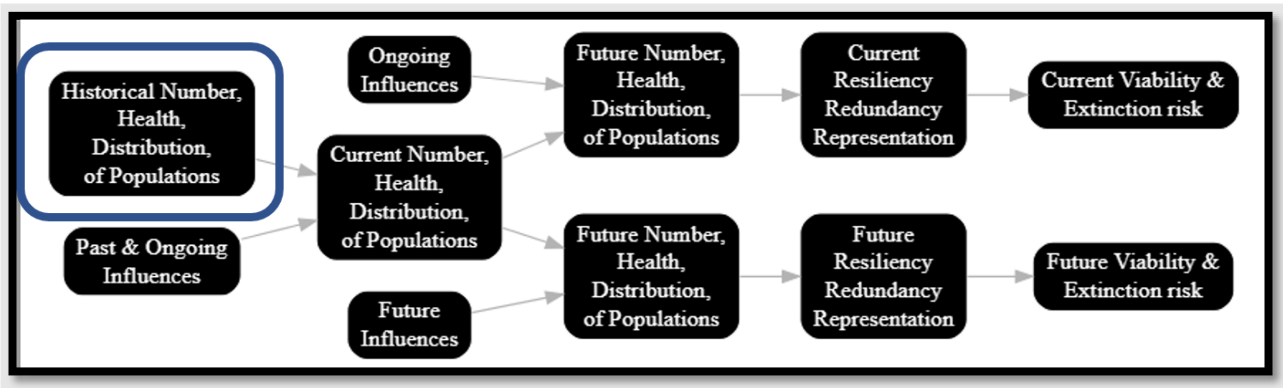

Figure 3.1. Highlighting (blue rectangle) the current step in our analytical framework.

Prior to 2006 (i.e., before WNS was first documented; see Chapter 4), NLEB was abundant and widespread throughout much of its range (despite having low winter detectability) with 737 occupied hibernacula, a maximum count of 38,181 individuals and its range being spread across >1.2 billion acres in 29 states and 3 Canadian provinces (Figure 3.2, Table A-3A1)[3]. NLEB numbers vary temporally and spatially, but abundance and occurrence on the landscape were stable (Cheng et al. 2022, p. 204; Wiens et al. 2022, p. 233). Winter colony sizes ranged from small (less than 100) to large (greater than 100), although the vast majority of individuals included in our dataset occupied a small subset of hibernacula; for example, in 2000, 16.6% (n = 66) of the known winter colonies contained 90% of total winter abundance.

Historically, the core of NLEB's range was centered in the Eastern Hardwoods RPU. This RPU encompasses approximately 90% of the total number of known hibernacula and 78% of the known winter abundance. The Southeast RPU contained 7% of the sites and 1% of total abundance, while the Subarctic RPU comprised 1% of the sites and 14% of the abundance. The Midwest and East Coast RPUs comprised 1% of the sites and 3% and 4% of the abundance, respectively (Table A-3A2).

The summer range for NLEB encompasses 37 states and 8 Canadian provinces (Figure 2.2). In this SSA, we have records of occurrences (i.e., NLEB acoustic calls, mist-net captures, and hibernacula records) from 37 states, the District of Columbia and 7 provinces (Figure 3.3).

[3] Hibernacula count numbers, number of hibernacula, and spatial range only represent NLEB available (i.e., usable format, provided within certain timeframe) winter records submitted to NABat (NABat 2021) for use in this SSA; we acknowledge historical NLEB abundance, number of hibernacula, and spatial range were likely higher.

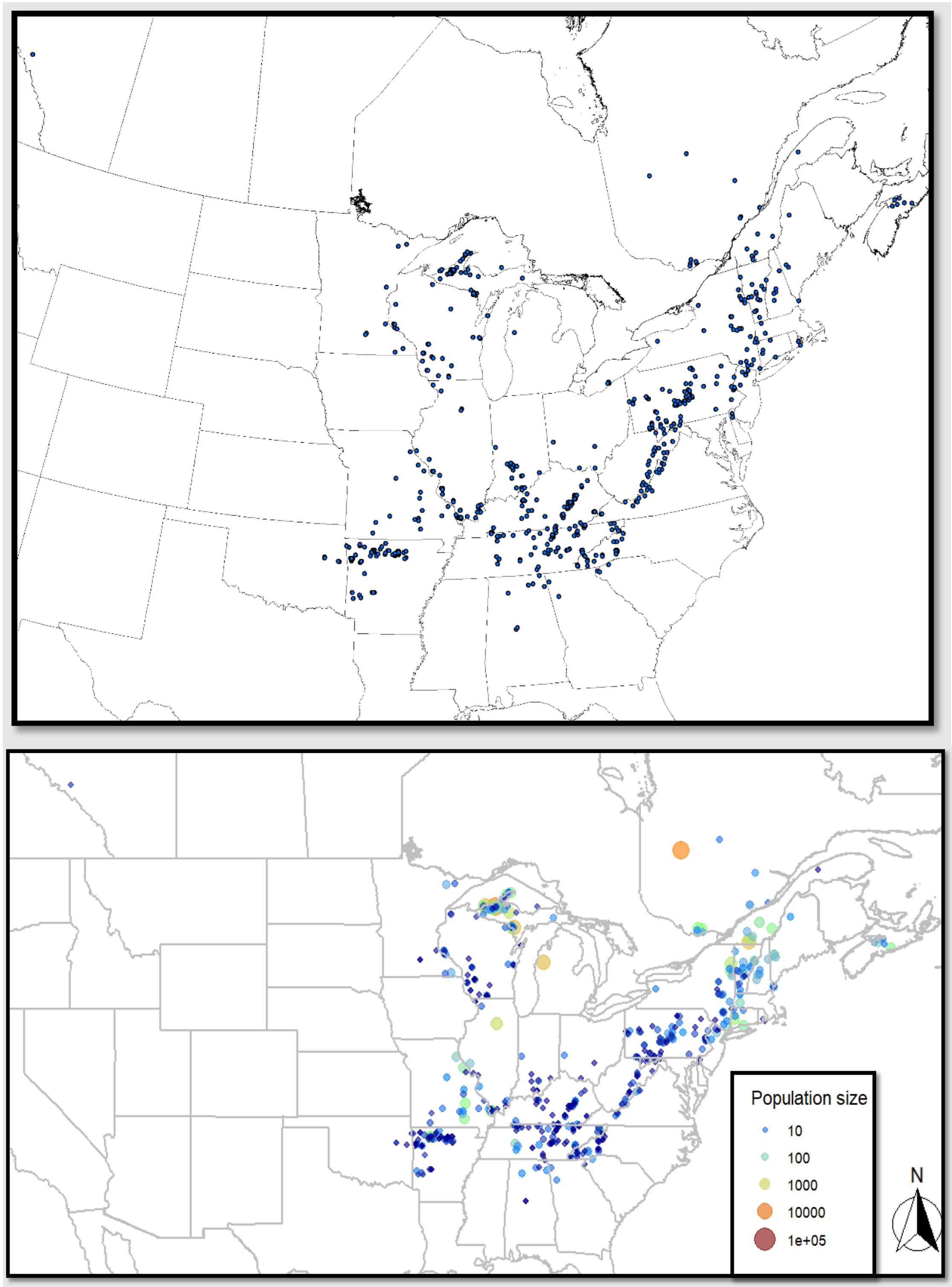

Figure 3.2. All known historical hibernacula (top figure) and winter abundances at hibernacula in 2000 (bottom figure). Point color and size corresponds to maximum colony count size at a hibernaculum.

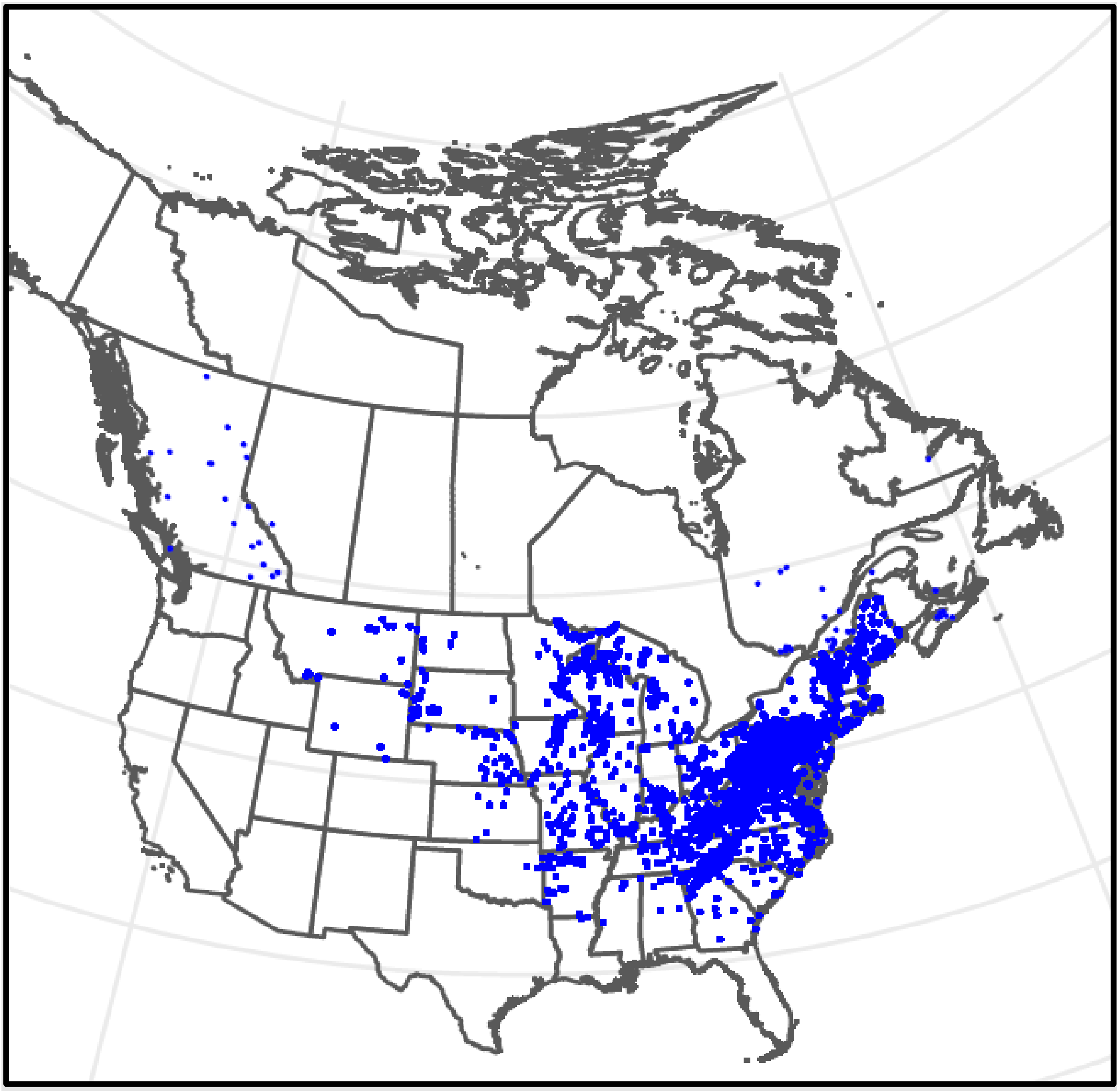

Figure 3.3. Documented range of NLEB as known from available acoustic calls, captures, and hibernacula records (records indicated by blue dots) in the U.S. and Canada. (Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center. Disclaimer: Provisional information is subject to revision). This map shows data provided to the SSA and does not replace the accepted species range (Figure 2.2).

CHAPTER 4 – PRIMARY INFLUENCES ON VIABILITY

Recognizing there are myriad influences operating on NLEB, this chapter describes the primary threats that have most likely led to its current condition: WNS, wind related mortality, effects from climate change, and habitat loss (Figures 4.1 and 4.2). We similarly describe the primary past and ongoing conservation efforts that may be ameliorating these threats. Lastly, for WNS and wind related mortality we describe the plausible future condition for each threat. To capture the uncertainty in our future projections, we identified the lowest plausible and highest plausible state for each primary threat. These lower and upper impact states for each threat were then combined to create composite plausible “low impact” and “high impact” scenarios. For climate change and habitat loss, we provide qualitative assessments.

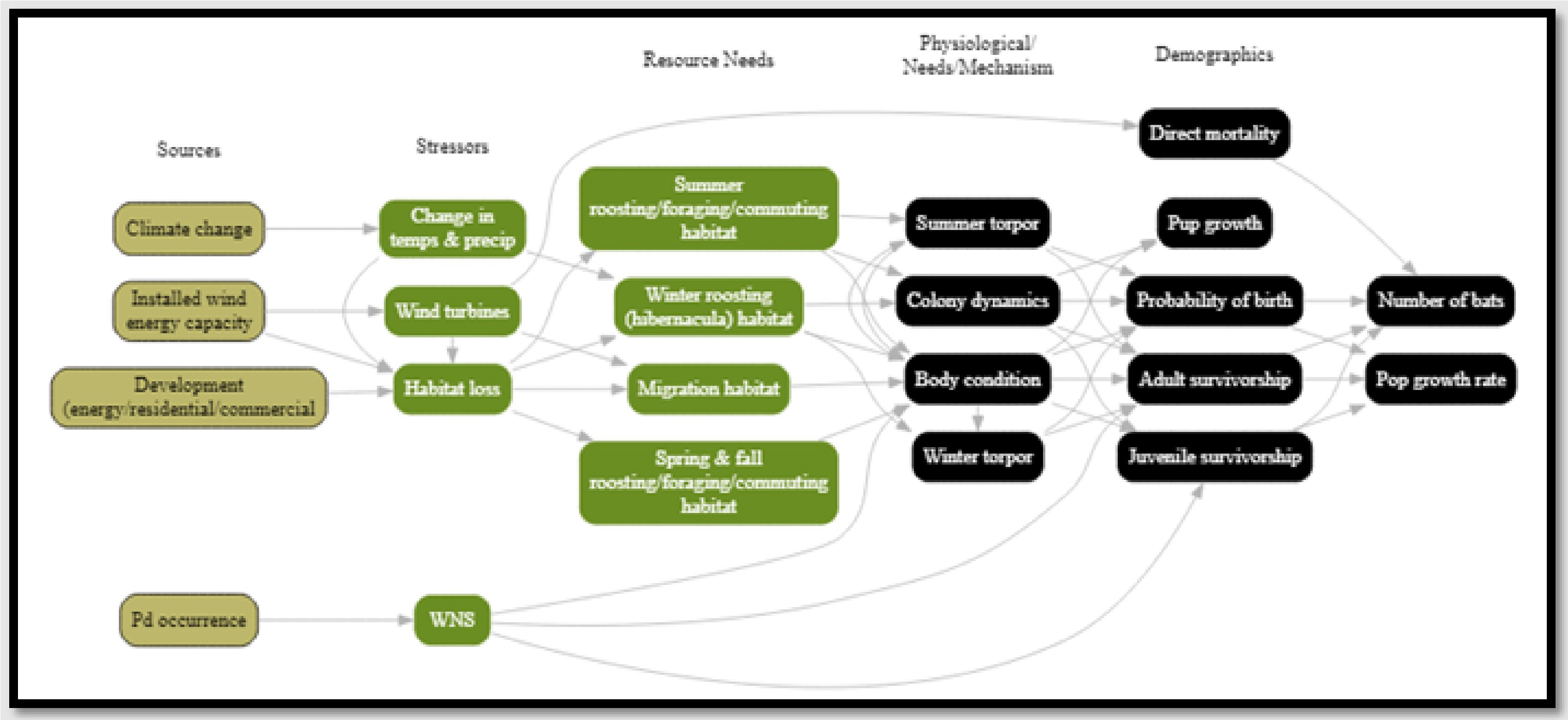

Figure 4.1. Visual diagram showing relationships between the primary threats and population needs.

Current Threat Conditions

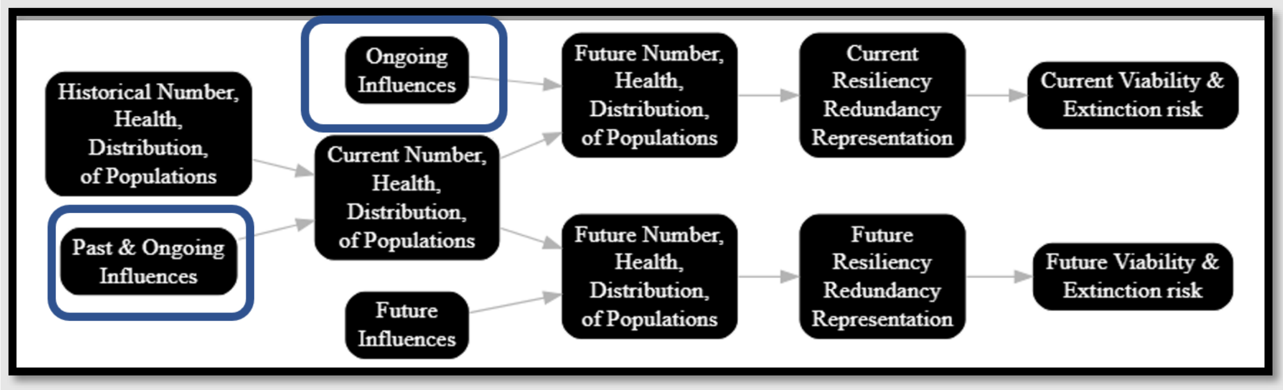

Figure 4.2. Highlighting (blue rectangle) the current step in our analytical framework.

White-nose Syndrome

For over a decade, WNS has been the foremost stressor on NLEB. WNS is a disease of bats that is caused by the fungal pathogen *Pd* (Blehert et al. 2009, entire; Turner and Reeder 2009, entire; Lorch et al. 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2016, entire; Puechmaille and Willis et al. 2017, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first discovered in eastern New York in 2007 (with photographs showing presence since 2006) (Meteyer et al. 2009, p. 411), and since then has spread to 39 states and 7 provinces in North America (Figure 4.3). *Pd* invades the skin of bats, initiating a cascade of physiological and behavioral processes that often lead to mortality (Warnecke et al. 2013, p. 3; Verant et al. 2014, pp. 3–6). Infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves needed to survive winter (Reeder et al. 2012, p. 5; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). Bats that do not succumb to starvation in hibernacula often seek riskier roosting locations near entrances to roosts or emerge from roosts altogether, where they face exposure to winter conditions and scarce prey resources on the landscape (Langwig et al. 2012, p. 2). The weeks following emergence from hibernation also mark a critical period because prey availability is still limited, energetic costs of healing from WNS are high, and the potential for immune reconstitution inflammatory syndrome that can lead directly to mortality or impact reproductive success (Reichard and Kunz 2009, p. 461; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8). As of May 2021, WNS has been confirmed in 12 species in North America, including NLEB, and numerous other species in Europe and Asia (www.whitenosesyndrome.org, accessed May 13, 2021; Hoyt et al. 2021, Suppl. material).

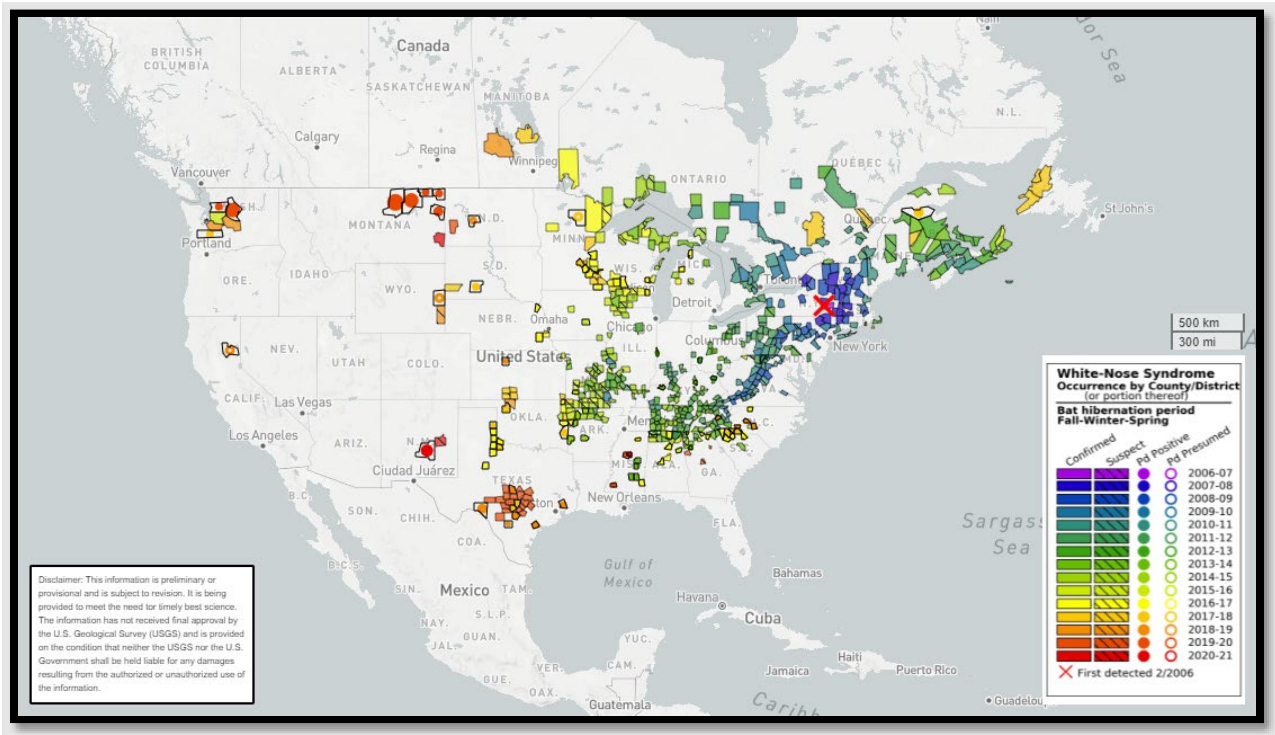

Figure 4.3. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles). Pd and WNS occurrence records generally reflect locations of winter roosts and are not representative of the summer distribution of affected bats (www.whitenosesyndrome.org, accessed May 13, 2021).

The fungal pathogen is spread primarily via bat-bat and bat-environment-bat movement and interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). With the arrival of *Pd* (year 0) to a new location, WNS progresses through "stages" similarly to many emerging infectious diseases: pre-invasion, invasion, epidemic, and establishment (Langwig et al. 2015, p. 196; Cheng et al. 2021, entire). During *invasion* (years 0–1), the fungus arrives on a few bats and spreads through the colony as a result of swarming and roosting interactions until most individuals are exposed to the pathogen. Such interactions may occur in hibernacula or at nearby roosts where conspecifics engage in mating activity (Neubaum and Siemers, 2021, p. 2). As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7), and 8 years after its arrival, *Pd* is determined to be *endemic* in a population (Langwig et al. 2015, p. 196; Cheng et al. 2021, entire).

The effect of WNS on NLEB has been extreme, such that most summer and winter colonies experienced severe declines following the arrival of WNS. Just 4 years after the discovery of WNS, for example, Turner et al. (2011, pp. 18–19) estimated that NLEB experienced a 98% decline in winter counts across 42 sites in Vermont, New York and Pennsylvania. Similarly, Frick et al. (2015, p. 5) estimated the arrival of WNS led to a 10–fold decrease in NLEB colony size. Most recently, Cheng et al. (2021, entire) used data from 27 states and 2 provinces to conclude WNS caused estimated population declines of 97–100% across 79% of NLEB's range. Although variation exists among sites, the arrival of *Pd* caused marked decrease in population abundance during invasion, epidemic, and established stages of the disease (Figure 4.4), with few exceptions (Figure 4.5). These analyses were extended to include additional data and years by Cheng et al. (2022, p. 212; Figure 4.4, Figure 4.5).

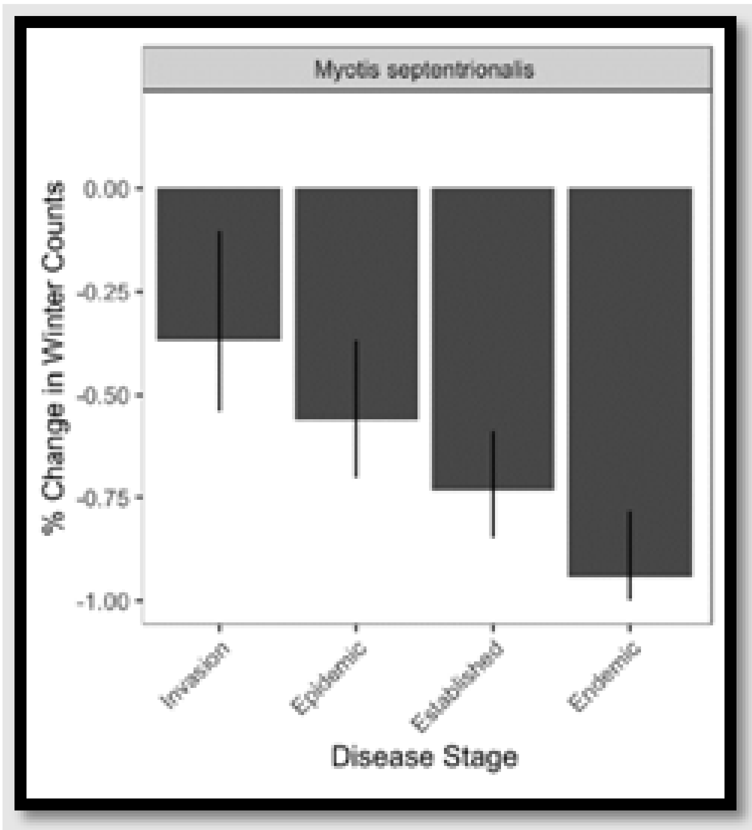

Figure 4.4. Percent change in winter colony counts by disease stage relative to predicted median count prior to arrival of Pd (with 95% credible interval) (Cheng et al. 2022, p. 212).

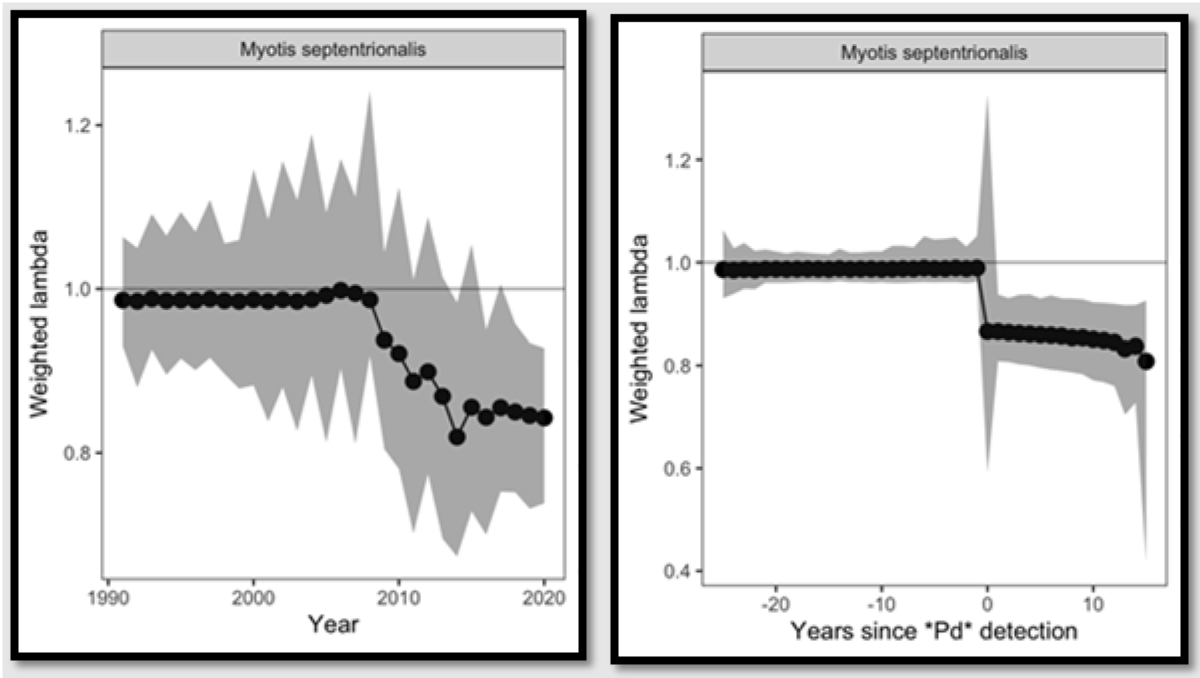

Figure 4.5. Estimated weighted lambda (function of growth rate and colony size) by year (left) and by year since arrival of Pd (right) (Cheng et al. 2022, p. 211).

Building off work of Cheng et al. (2022, entire), Wiens et al. (2022, entire) used available data from hibernacula surveys to estimate the annual impacts of WNS relative to the year of arrival of *Pd*, adding additional analysis of an endemic stage. Their analysis applied two models of *Pd* spread to interpolate WNS occurrence to all documented hibernacula. The analysis predicted *Pd* is present at 99–100% of documented hibernacula for NLEB (Appendix 2-A). Although variation exists among sites, an overwhelming majority of hibernating colonies of NLEB have developed WNS and experienced serious impacts within 2–3 years after the arrival of *Pd* (Cheng et al. 2021, entire; Wiens et al. 2022, pp. 231–247) (Figure 4.5).

A variety of factors may contribute to the differences observed amongst hibernacula. Year-round temperature profiles may affect the environmental reservoir of *Pd*, thus reducing the source of reinfection when bats return to the locations each fall, which would be more likely to delay than preclude infection (Hoyt et al. 2020, pp. 7257–7258). However, it is important to acknowledge that bats likely encounter multiple subterranean environments during swarming activity, during which they can encounter reservoirs of *Pd* (Neubaum and Siemers, 2021, pp. 3–4). Over winter temperature and climate may also affect the physiology of hibernating bats in these sites or offer foraging opportunities that make it possible for them to avoid more serious infections, but these mechanisms have not been tested. Regardless, the vast majority of NLEB colonies exposed to *Pd* have developed and will continue to develop WNS and experience impacts from the disease (Cheng et al. 2021, entire; Wiens et al. 2022, pp. 231–247) (Figure 4.6).

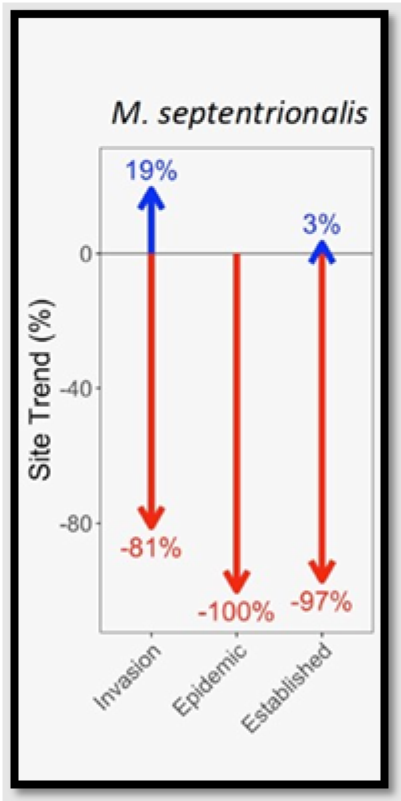

Figure 4.6. Percentage of accessible winter colonies with increasing (blue) and decreasing (red), colony trend relative to WNS pre-arrival stage for invasion, epidemic, and established stages (Cheng et al. 2021, entire; appendix S3).

There are multiple national and international efforts underway in attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts. See Appendix 4-A for more information regarding WNS impacts.

Wind Related Mortality

Wind related mortality, overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and RPU levels. Wind power is a rapidly growing portion of North America's energy portfolio in part due to changes in State energy goals (NCSL 2021, web) and recent technological advancements (Berkeley Lab 2020, web) and declining costs (Wiser et al. 2021, entire), allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association (AWEA) 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often have tens or hundreds of turbines installed in a given area, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of October 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[4] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat mortalities continue to be documented at wind power installations across North America and Europe. We describe mechanisms leading to bat fatalities in Appendix 4-B.

Bat fatality varies across facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise the majority of all known bat fatalities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other non-listed bat species. However, there is notable spatial overlap between NLEB occurrences and wind facilities (Figure 4.7) along with NLEB mortality documented. At the 2020 installed MW capacity, we estimated 122 NLEB are killed annually at wind facilities (Table 4.1). Analyses using data from Wiens et al. (2022, pp. 236–247) and analyses by Whitby et al. (2022, entire) suggest that the impact of wind related mortality is discernible in the ongoing decline of NLEB. Based on data from Wiens et al. (2022, pp. 236–247) comparing a no wind baseline scenario to current and future wind scenarios, the projected abundance decreases 24–33% by 2030 under the current wind scenario and up to 83% by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, entire) found a decline in the predicted relative abundance of NLEB as wind energy risk index increased. To reduce bat fatalities, some facilities "feather" turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein and Straw, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed, or at a higher threshold, typically referred to as curtailment. The effectiveness of feathering below various cut-in speeds differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with feathering below wind speeds of 4.0–6.5 meters per second (m/s) (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for NLEB, however, has not been documented. Hereafter, we refer to feathering below the manufacturer's cut-in speed or higher wind speeds collectively as curtailment.

[4]Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, p. 32 and 69).

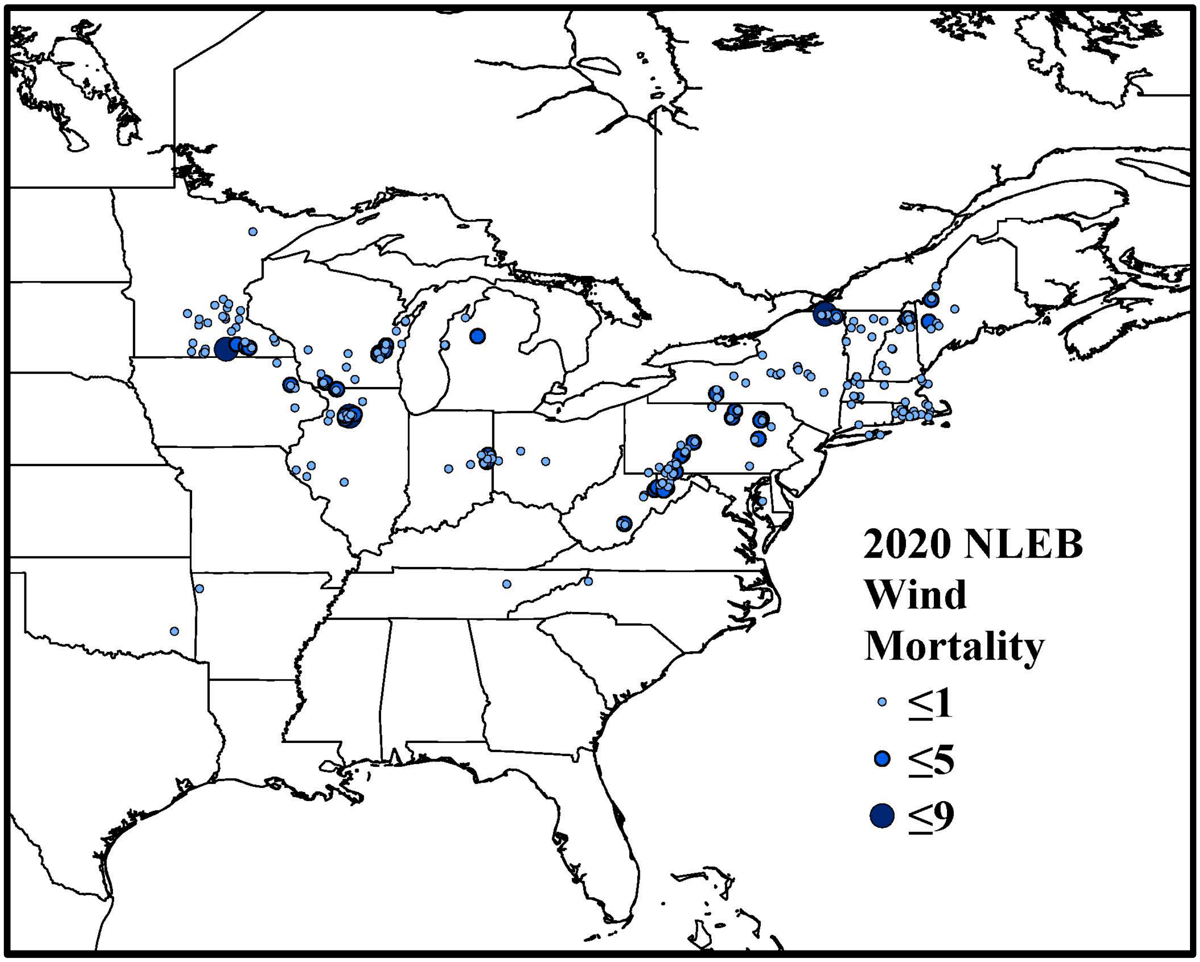

Figure 4.7. Estimated total annual NLEB mortality at wind facilities in 2020. Mortality is shown at U.S. wind turbines as summed by 11x11-km NREL grid cell within the migratory range of extant NLEB hibernacula. Note that because MW were summed by Province centroid in Canada (and none were within the migratory range of hibernacula), the only NLEB mortality that was allocated to Canadian hibernacula (Quebec) was that occurring at U.S. turbines within the migratory range. See Udell et al. 2022, pp. 265–266 and Appendix 2 for details on the wind mortality analysis.

Table 4.1. Estimated annual NLEB mortality from wind facilities allocated to hibernacula by USFWS Region (Figure A-2A6) and Canada, based on installed MW capacity in October 2020 (Udell et al. 2022, pp. 265–266).

Location	Mean Annual Mortality (n)	Lower CI	Upper CI
Region 2	0	0	0
Region 3	59	19	72
Region 4	1	0	1
Region 5	58	17	72
Quebec	4	1	5
Total	**122**	**38**	**150**

There are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. To date, operational strategies (e.g., feathering turbine blades when bats are most likely to be active) are the only broadly proven and accepted measures to reduce the severity of impacts. See Appendix 4-B for more information.

Climate Change

There is growing concern about impacts to bat populations in response to climate change (Jones et al. 2009, entire; Jones and Rebelo 2013, entire, O'Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats, including changes in hibernation, mortality from extreme drought, cold, or excessive rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed and discussed potential impacts of climate change, including effects to bat foraging, roosting, reproduction, and biogeography. Climate change is also likely to influence disease dynamics as temperature, humidity, phenology and other factors affect the interactions between *Pd* and hibernating bats (Hayman et al. 2016, p. 5; McClure et al. 2020, p. 2; Hoyt et al. 2021, p. 8). However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). Climate change may impact these bats in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid (*Adelges tsugae*), southern pine beetle (*Dendroctonus frontalis*)) (Haavik 2019).

While there are a number of changing climatic variables, our analysis focused solely on changes in temperature and precipitation. These variables influence NLEB resource needs, such as suitable roosting habitat (all seasons), foraging habitat, and prey availability (Figure 4.1). Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., annual average temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986 to 2016 relative to 1901 to 1960 (Hayhoe et al. 2018, p. 86). Temperatures increased during that time at a regional scale as well, with the largest changes (average increases of more than 1.5 degrees F (0.8 degrees C)) in Alaska, the Northwest, the Southwest, and the Northern Great Plains and the least change in the Southeast (increase of 0.46 degrees F (0.26 degrees C); Vose et al. 2017, pp. 186–187; Hayhoe et al. 2018, p. 86). Annual average precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the West, Southwest and Southeast (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events across the U.S. have increased more than the increases in average precipitation (Hayhoe et al. 2018, p. 88).

NLEB risk of exposure to changes in the climate is rangewide. However, the magnitude, direction, and seasonality of climate variable changes is not consistent rangewide. In addition, the resiliency of populations and inherent differences (e.g., genetics, summer roost microclimates) among populations may result in differing ability for NLEB to respond to the same types of changes across the range. Therefore, the overall impact of climate change for such a wide-ranging species is challenging to describe. Although there may be some benefit to NLEB from a changing climate, overall negative impacts are anticipated. Although we lack species-specific observations for NLEB, observed impacts to date for other insectivorous bats, such as the little brown bat, include reduced reproduction due to drought conditions leading to decreased availability of drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years (drought) in the Northeast (Frick et al. 2010, pp. 131–133). While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During the anticipated heavier precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795). Bats are likely to reduce their foraging bouts during heavy rain events and reduced reproduction has been observed during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Responses will vary throughout NLEB range based on the extent of annual temperature rise in the future. For additional information on climate change see Appendix 4-C.

Habitat Loss

Roosting/Foraging/Commuting Habitat Loss

As discussed in Chapter 2, NLEB require suitable habitat for roosting and foraging, and commuting between those habitats during spring, summer, and fall. Forest is a primary component of roosting, foraging, and commuting habitat. Wetlands and water features are important foraging and drinking water sources. Loss of these habitats influences survival and reproduction of NLEB colonies.

We reviewed changes in various NLCD landcover classes within each RPU from 2006 to 2016 in the continental U.S. Overall, forest landcover was fairly stable in all RPUs with slight annual increases (27,000 to 50,000 acres/year) in all but Midwest RPU (loss of 23,000 acres/year). However, deciduous forest landcover decreased across all RPUs by 1.4 million acres for an average loss of 140,000 acres per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 1.4 million acres. See Appendix 4-D for additional information.

These changes in landcover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality. While temporary or permanent habitat loss may occur throughout all states within the species' range, impacts to NLEB typically occur at a more local-scale (i.e., individuals and potentially colonies).

Impacts to the NLEB from loss of habitat vary depending on the timing, location, and extent of the removal.

Impacts from forest habitat removal may range from minor (e.g., removal of a small portion of foraging habitat in unfragmented forested area with a robust NLEB population) to significant (e.g., removal of roosting habitat in highly fragmented landscape with small, disconnected population). Adverse impacts are more likely in areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), as there is a higher probability of removing roosts or causing loss of connectivity between roosting and foraging habitat. There are a variety of conservation measures that can either serve to reduce effects from habitat loss or help maintain or enhance habitat. See Appendix 4-D for examples.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, NLEB require hibernation sites with specific microclimates and NLEB exhibit high interannual fidelity to their hibernacula. Therefore, the complete loss of or modification of winter roosts (such that the site is no longer suitable) can result in impacts to individuals or at the population level. In addition, disturbance within hibernacula can render a site unsuitable or can pose harm to individuals using the site.

Modifications to bat hibernacula (e.g., erecting physical barriers to control cave and mine access, intentional or accidental filling or sealing of entries, or creation of new openings) can alter the ability of bats to access the site (Spanjer and Fenton 2005, p. 1110) or affect the airflow and alter microclimate of the subterranean habitat, and thus the ability of the cave or mine to support hibernating bats, such as NLEB. These well-documented effects on cave-hibernating bat species were discussed in the USFWS's *Indiana Bat Draft Recovery Plan* (USFWS 2007, pp. 71–74). In addition to altering the thermal or humidity regime and ability of the site to support hibernating bats, bats present during any excavation or filling can be crushed or suffocated. Sources of these stressors include fill from adjacent activities, mining, and intentional closures of abandoned mines or cave openings to restrict access.

Human entry or other disturbance to hibernating bats results in additional arousals from hibernation which require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable. This is even more important for sites where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Willis and Boyles 2012, p. 96).

There are many conservation efforts and protections (e.g., bat-friendly gates, closure of caves during hibernation) in place that attempt to reduce the risk of modifications to hibernacula and disturbance to overwintering bats. See Appendix 4-D for more information.

Conservation Efforts

Conservation efforts associated with reducing the effects of WNS, wind related mortality, and habitat loss are mentioned above and discussed further within associated appendices. In addition to those efforts, below we highlight the regulatory protections afforded to NLEB in parts of its range.

Federal, State, Provincial Protection

NLEB was listed as threatened under the Endangered Species Act on April 2, 2015 (USFWS 2015, entire). We also developed a final 4(d) rule, which published in the *Federal Register* on January 14, 2016 (USFWS 2016, entire). The 4(d) rule specifically defines the "take" prohibitions. NLEB was listed as endangered on Schedule 1 of Canada's Species at Risk Act in 2014. This provided the NLEB protection from being killed, harmed, harassed, captured, or taken in Canada. Environment and Climate Change Canada finalized a recovery strategy for NLEB in 2018 (Environment and Climate Change Canada 2018, entire).

In addition, NLEB receives varying degrees of protection through state laws as it is designated as Endangered in Arkansas, Connecticut, Delaware, Indiana, Maine, Massachusetts, Missouri, New Hampshire, Vermont; Threatened in Georgia, Illinois, Louisiana, Maryland, New York, Ohio, Pennsylvania, Tennessee, Virginia, and Wisconsin; and Special Concern in Alabama, Iowa, Michigan, Minnesota, Mississippi, Oklahoma, South Carolina, South Dakota, West Virginia, and Wyoming.

Synopsis of Current Threat Conditions

To provide a comparative assessment of the primary influences, we summarize the scope, severity, and impact of each of the four influences using criteria defined by Master et al. (2012, pp. 28–35; Table 4.2). Currently, WNS is the greatest threat to NLEB, with WNS related population declines occurring over 78% (pervasive in scope) of NLEB's range of an estimated 97–100% (extreme severity; Cheng et al. 2021, entire). Wind mortality, although large in scope (occurring over 49% of range) has a "medium" level impact to NLEB due to a moderate to serious severity based on differences in the two models (current population-level decline of 24–33% (Table A-3D1)). A "medium" impact level for wind mortality was decided on in part due to mortality rates being kept constant for projections in the model and as declines increase, presumably so will exposure to wind mortality, which reduces overall impact. While confidence in impact to NLEB from WNS and wind were "moderate to high" due to availability of quantitative data, our confidence analysis of the impact of habitat loss and climate change remain "moderate to low" due to minimal quantitative data. Both habitat loss and climate change are pervasive, occurring across the species' range, while severity of population level declines are predicted to be slight. Conservation efforts, such as protection of winter hibernacula from disturbance and habitat protections for NLEB and other listed species, may provide some benefit to NLEB populations. Lastly, habitat loss (e.g., disturbance to or loss of maternity colony, tree removal) and climate change (e.g., precipitation levels, rising temperatures) are anticipated to vary regionally, but have more localized impacts on the species viability.

Table 4.2 Assessment of current impact to NLEB from primary threats (adapted from Master et al. 2012). See Chapter 1 for definitions of the criteria (Figure 1.2).

Criteria	**WNS**	**Wind Mortality**	**Habitat Loss**	**Climate Change**
Scope	Pervasive	Large	Pervasive	Pervasive
Severity	Extreme	Moderate	Slight	Slight
Impact	**Very High**	**Medium**	**Low**	**Low**
Confidence level	High	Moderate	Moderate	Low

Future Threat Conditions

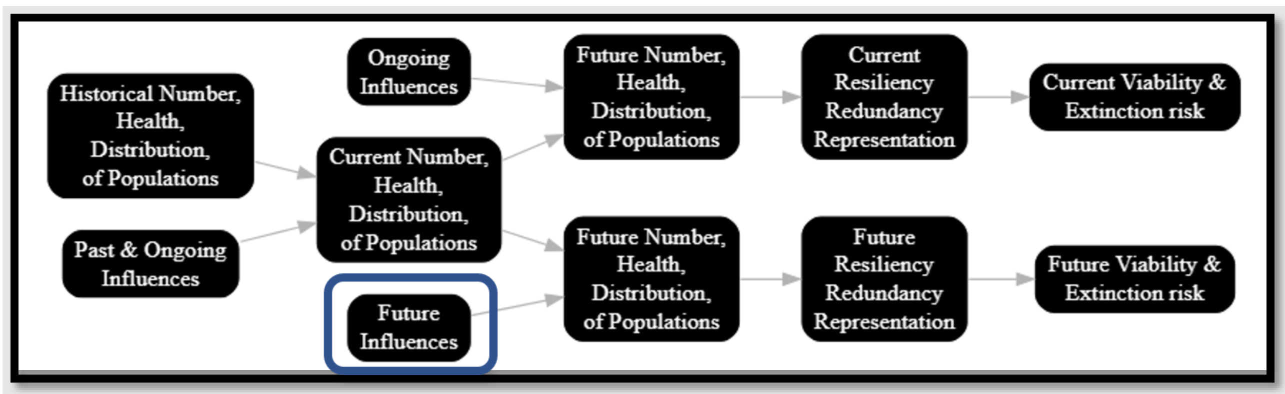

Figure 4.8. Highlighting (blue rectangle) the current step in our analytical framework.

To assess how NLEB will respond to foreseeable changes in *Pd* and wind energy capacity, we identified the plausible future state of these influences (Figure 4.8). We developed realistic lower and upper bounds for both and combined them to create composite plausible “high impact” and “low impact” scenarios. The composite future scenarios for WNS and wind mortality is summarized in Table 4.3. These scenarios and their underlying rationales are described below, along with the future projected conditions for habitat loss and climate change. We provide further rationale for our low and high impact scenarios in Appendix 5.

Table 4.3. NLEB composite plausible future scenarios.

Plausible Scenario	WNS Spread	WNS Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition	*Pd* rate
Low impact	*Pd* occurrence model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	U.S. - combined, Canada - regional-specific	No
High impact	*Pd* occurrence model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	U.S. - combined, Canada - regional-specific	No

White-nose Syndrome

To project future impacts of WNS, we relied on 1) predicted current and future occurrence of *Pd* on the landscape using two different models (hereafter, "*Pd* occurrence models") and 2) the WNS impacts schedule. For the latter, we assumed winter colonies that are exposed to *Pd* in the future will respond similarly to those currently exposed (i.e., colonies exposed in the future will follow the same WNS impacts schedule) (see Chapter 1, *Step 3. Identify the Primary Drivers (Influences)* and Appendix 5 for more detail).

To project future spread of WNS, we relied upon two *Pd* occurrence models, *Pd* occurrence model 1 (derived by Wiens et al. 2022, pp. 226–229) and *Pd* occurrence model 2 (derived by Hefley et al. 2020, entire); both models are briefly described in Appendix 2. For a low impact scenario, we used *Pd* occurrence model 1 for predicted year of arrival (YOA) and assumed that the WNS impacts schedule continues for 15 years after arrival of *Pd*, after which the colonies return to pre-WNS survival rates for the remainder of the simulation (i.e., no WNS impacts applied after 15 years since *Pd* arrival). Return to pre-WNS growth rates at YOA 15 is the earliest year we can reasonably assume (given data show impacts continue occurring 14-years since the first detection in New York). For the high impact scenario, we used *Pd* occurrence model 2 for predicted YOA and assumed that WNS impacts continue through 2060 (i.e., after YOA 0 to 6, survival rates remain in the endemic phase).

Wind Related Mortality

To project future installed wind capacity, we relied upon National Renewable Energy Laboratory's (NREL; Cole et al. 2020) and Canadian Energy Regulator's (CER) (CER 2020) projections for the U.S. and Canada, respectively (Figure 4.9). Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario* (Figure 4.10). Our high impact scenario (i.e., higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Figure 4.11). For both scenarios, we calculated NLEB fatalities per MW using the species composition approach (see

Chapter 1 methods and Appendix 2-A for additional detail). The annual mortality associated with the future low and high impact scenarios by Year 2050 is provided in Table 4.4.

We selected NREL's scenarios per consultation with the U.S. Department of Energy's (USDOE) Wind Energy Technology Office (P. Gilman 2020, Program Manager, personal communication). The NREL scenarios model future deployment levels based on projected trends in electricity demand, technology cost trajectories, and existing Federal and state energy policies (Cole et al. 2020, p. iii; see Appendix 5 for details). NREL's 2020 (Cole et al. 2020) report presents 45 power sector scenarios that consider present day through 2050. We chose the *High Wind Cost* and *Low Wind Cost* scenarios as reasonable lower and upper bounds of future wind build-out, respectively. NREL agreed that use of the *High Wind Cost* and *Low Wind Cost* scenarios provides a reasonable range of future wind build-out (W. Cole 2020, personal communication).

CER's *Canada's Energy Future* report is published annually and provides up-to-date projections for wind build-out in Canada. CER uses economic and energy models to project future scenarios based on assumptions about *trends in* "*technology, energy and climate policies, energy markets, human behavior and the structure of the economy*" (CER 2019, p. 1). Annual wind build-out projections are produced at the province/territory level and data are continually refined based on current trends. We chose the *Reference Scenario* as our lower-impact scenario (i.e., lower wind build-out) and the *Evolving Scenario* as our higher-impact scenario (i.e., higher wind build-out; see Appendix 5 for details).

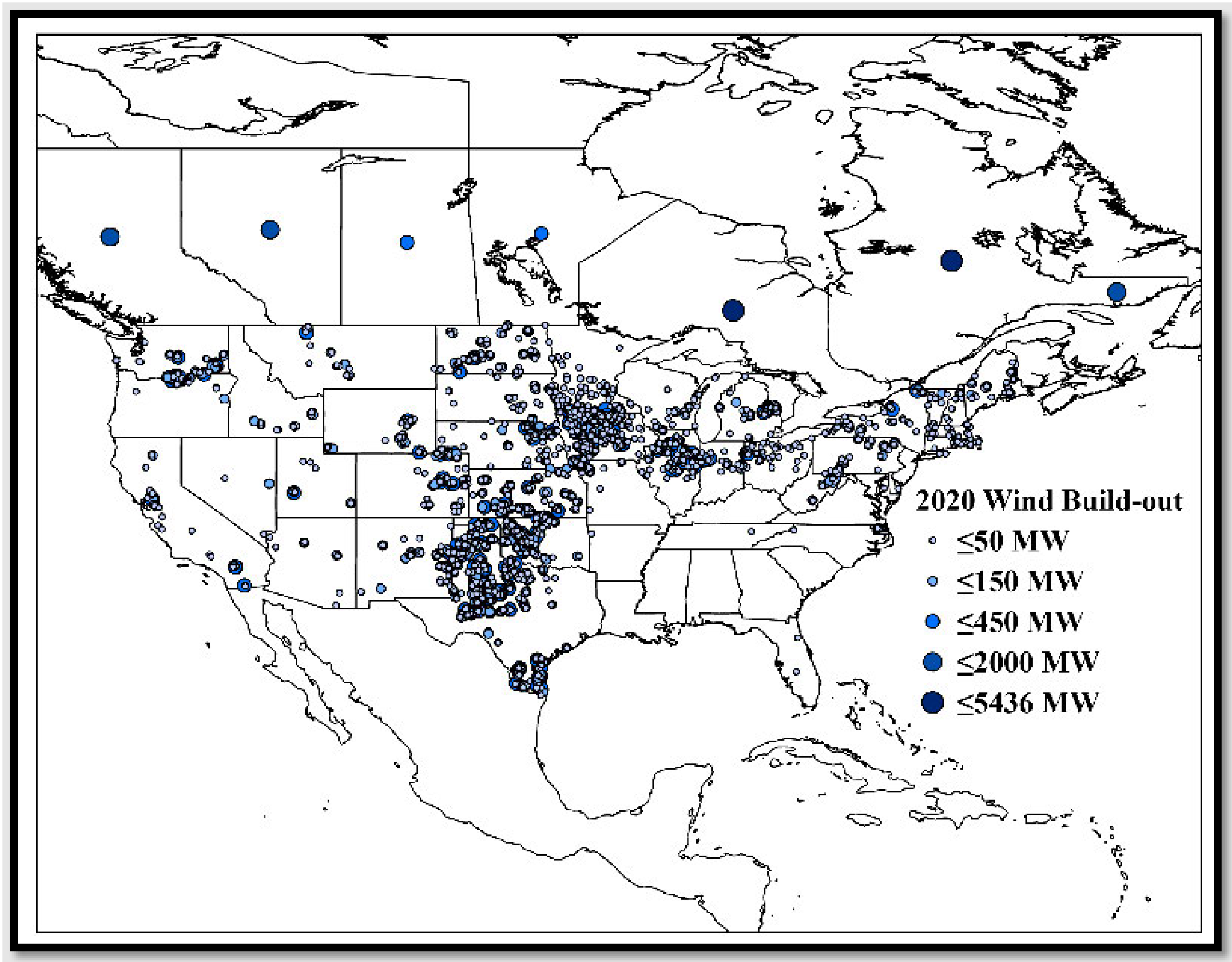

Figure 4.9. Wind build-out as of October 2020 for the U.S. and Canada (Udell et al. 2022, entire). U.S. capacity is summed by 11x11-km NREL grid cell and Canadian capacity by province.

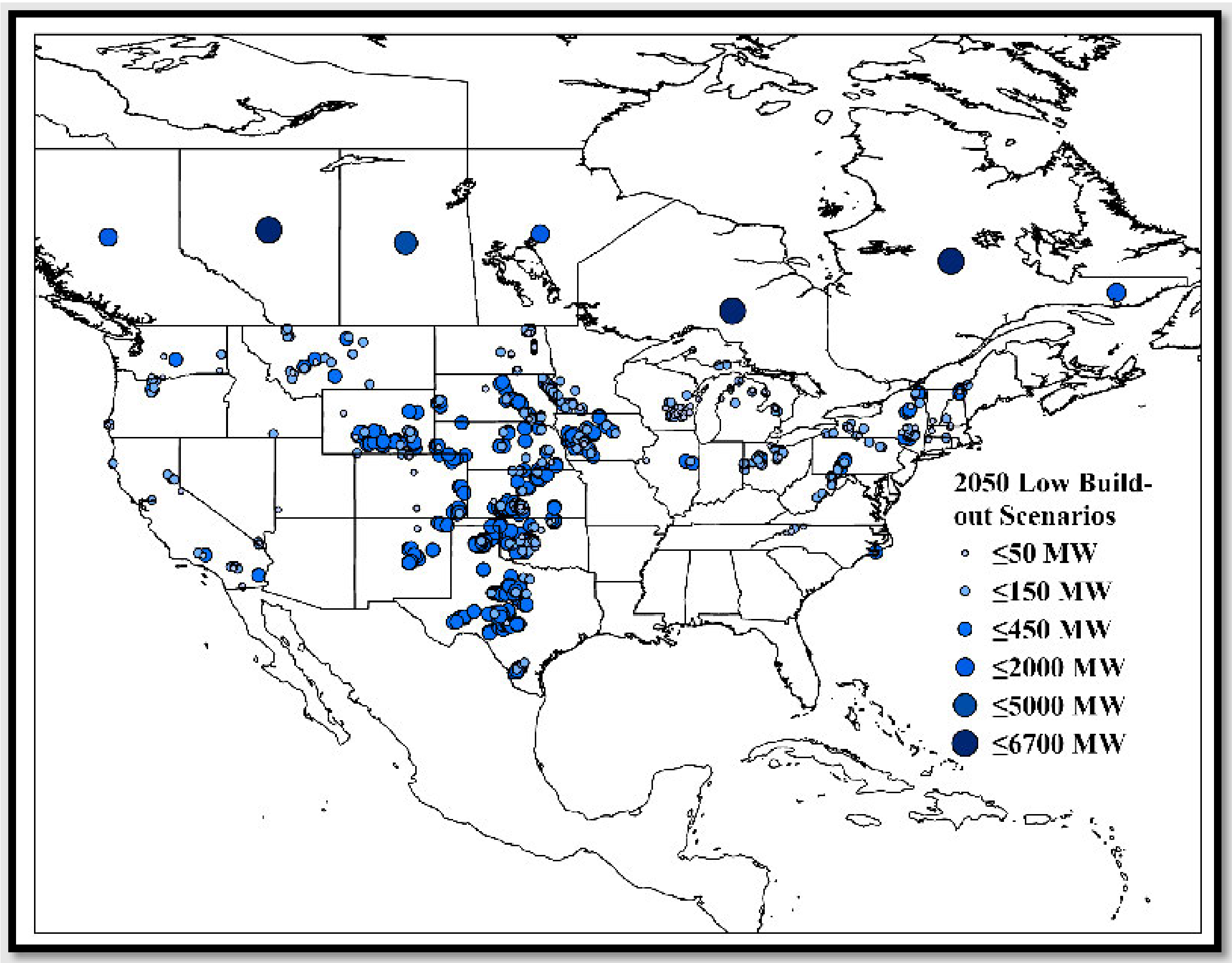

Figure 4.10. Projected wind build-out for the year 2050 per low build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km NREL grid cell and Canadian future capacity by province.

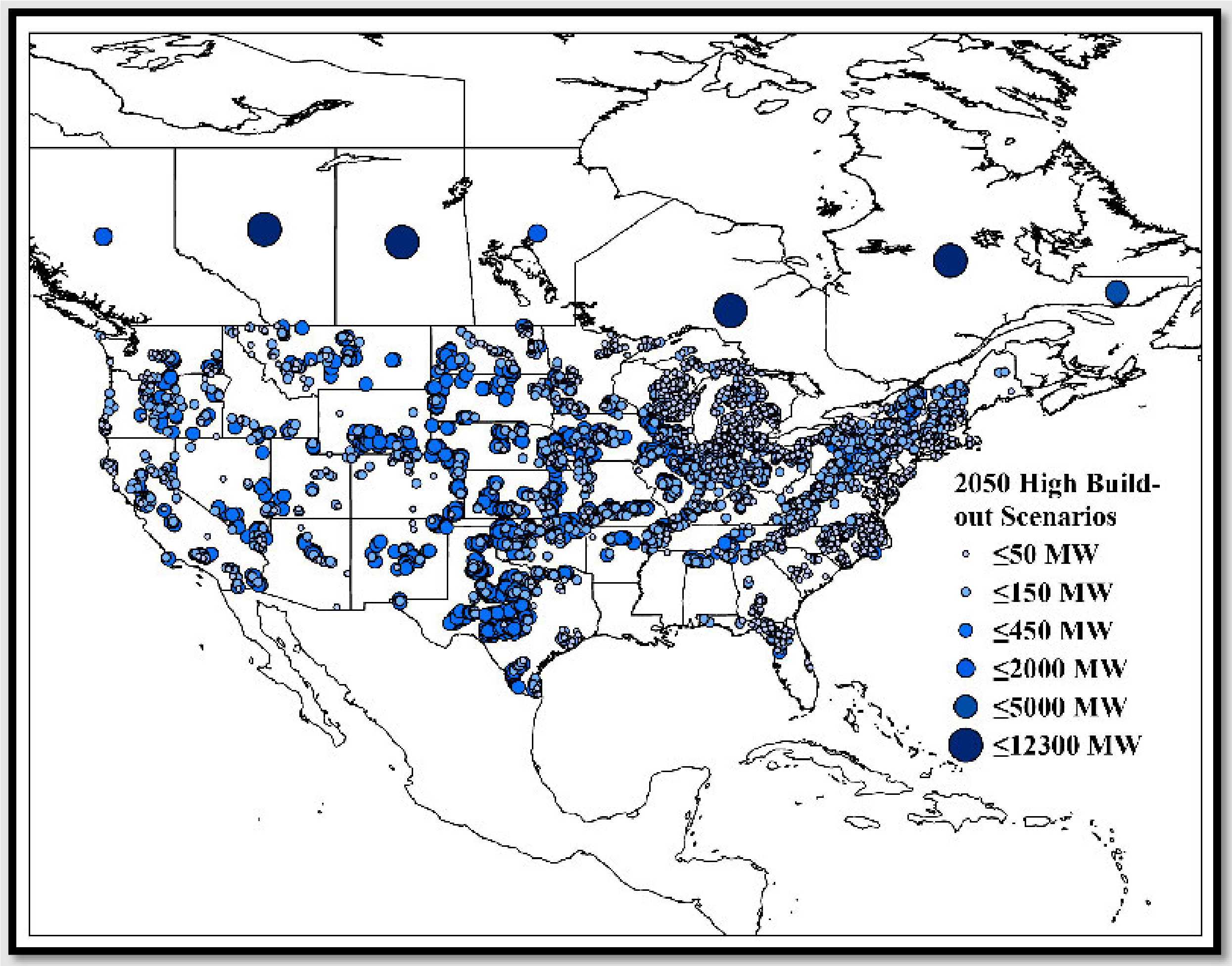

Figure 4.11. Projected wind build-out for the year 2050 per high build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km power grid and Canadian future capacity by province.

Table 4.4. Predicted annual NLEB mortality[5] (25th–75th percentile) by USFWS Region and Canada, based on projected 2050 installed wind capacity under low and high build-out scenarios (Udell et al. 2022, entire).

Location	**Low build-out**	**High build-out**
Region 2	0 (0–0)	33 (11–33)
Region 3	57 (18–70)	1,395 (447–1,703)
Region 4	3 (1–4)	307 (93–380)
Region 5	138 (42–172)	1,157 (349–1,440)
Quebec	4 (1–5)	35 (11–43)
Total	**202 (62–250)**	**2,926 (911–3,600)**

Climate Change

[5] It is likely that percent composition will decline as the species declines over time. To capture insights on the sensitivity of the results to wind energy mortality, we ran scenarios with zero and 50% reduction in wind energy mortality (see Appendix 1-B).

Over the next few decades, annual average temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C) relative to 1985 to 2015**,** regardless of any currently used representative concentration pathway (RCP 2.6 to RCP 8.5) (Hayhoe et al. 2018, p. 86). Larger increases are projected by late century of 2.3 to 6.7 degrees F (1.3 to 3.7 degrees C) under RCP4.5 and 5.4 to 11.0 degrees F (3.0 to 6.1 degrees C) under RCP8.5, relative to 1986 to 2015 (Hayhoe et al. 2018, p. 86).

For the period of 2070 to 2099 relative to 1986 to 2015, precipitation increases of up to 20% are projected in winter and spring for northcentral U.S., with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al. 2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

NLEB's responses to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as little brown bat (see above and Appendix 4-C). This includes reduced reproduction due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442), reduced adult survival during periods of drought (Frick et al. 2010, pp. 131–133), or reduced reproduction during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will vary depending throughout the ranges of the species' and on how much the annual temperature actually rises in the future.

Habitat Loss

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources (we have nothing comparable for Canada). This assessment was influenced by a set of future scenarios with varying assumptions regarding global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010 to 2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million hectares (16–34 million acres or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million hectares (9–21 million acres) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8–7). However, regenerating young forests temporarily lack large roosts that provide space and thermal needs for NLEB colonies. While past and projected forest loss and forest regeneration rates can provide a coarse assessment of long-term trends, they are not particularly meaningful for determining the magnitude of impact unless overlaid where the species actually occurs. Loss of essential population needs of roosts and foraging and commuting habitat within NLEB home range where they remain is the issue. Furthermore, loss of roosting and foraging habitat compounds the impacts from WNS (see Appendix 4-D).

Synopsis of Future Threat Conditions

Using the available data and information summarized above and in Chapters 5 and 6, we assigned the scope, severity, and impact given the projected future state conditions for each of the primary influences (Table 4.5). WNS continues to be the greatest threat to NLEB, due to the expected future declines in population abundances (98–100% in known hibernacula) over most to all of its range (Wiens et al. 2022, pp. 226–229). Confidence in impact to NLEB from WNS and wind were "high" due to availability of quantitative data. Wind mortality impact is expected to be pervasive in scope and increase in severity, with population impacts reaching 83% by 2060 (Table A-3D2). Although the increasing severity of wind energy related mortality suggests that a *High* to *Very high* ranking is appropriate, we believe that the fatality rates are likely to decline as the abundance declines. The data were too limited (therefore, our confidence level was "moderate") to discern whether fatality rates have declined as the species' abundance precipitously decreases, so our scenarios did not account for this likelihood. For this reason, we assumed the severity of wind energy mortality will stay constant ("moderate") over time along with the overall impact level (*Medium*).

Our confidence in analysis on the impact of habitat loss and climate change remain "low" to "moderate" due to minimal quantitative data. Both habitat loss and climate change are forecasted to remain pervasive across the species' range, while the severity of population level declines are predicted to range from slight to moderate due a reduction in the spatial distribution of the species across the range. Given NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies), the severity of habitat loss at occupied sites will vary between slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between *Low Impact* and *Very High Impact*. Lastly, increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase in the future leading to increased negative effects to NLEB (e.g., increased mortality, reduced reproductive success); therefore, our impact analysis predicts *Medium Impact* from climate change under future state conditions.

Table 4.5 Assessment of future impact from primary threats (adapted from Master et al. 2012 and Cheng et al. 2021, p. 5). See criteria definitions in Chapter 1 (Figure 1.2).

<table>
<tr><th>Criteria</th><th>WNS</th><th>Wind Mortality</th><th colspan="2">Habitat Loss</th><th>Climate Change</th></tr>
<tr><td>Scope</td><td>Pervasive</td><td>Pervasive</td><td colspan="2">Pervasive</td><td>Pervasive</td></tr>
<tr><td>Severity</td><td>Extreme</td><td>Moderate</td><td colspan="2">Slight-Extreme</td><td>Moderate</td></tr>
<tr><td>Impact</td><td>Very High</td><td>Medium</td><td>Low</td><td>Very High</td><td>Medium</td></tr>
<tr><td>Confidence Level</td><td>High</td><td>Moderate</td><td colspan="2">Moderate</td><td>Low</td></tr>
</table>

CHAPTER 5 – CURRENT CONDITION

In this chapter, we describe the current demographic conditions and the projected number, health, and distribution of NLEB populations given these current conditions (Figure 5.1). Current state conditions encompass the current abundance, growth rate, WNS occurrence, and installed wind energy capacity. We projected abundance under current state conditions to garner insight into viability, which we describe in Chapter 7.

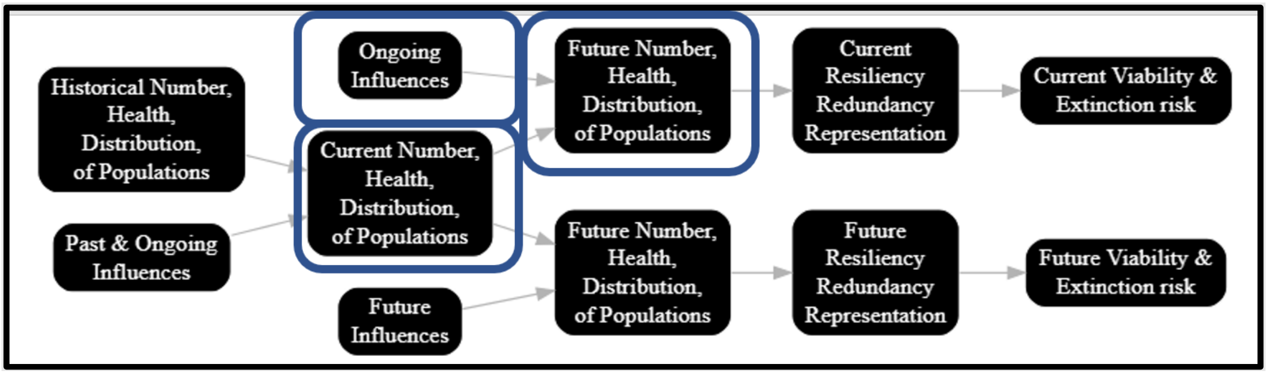

Figure 5.1. Highlighting (blue rectangles) the current step in our analytical framework.

Current demographic conditions– Available evidence indicates NLEB abundance has and will continue to decline substantially over the next 10 years under current conditions (Figure 5.2). Evidence of the past decline is demonstrated in available data in both winter and summer. For example, rangewide winter abundance has declined by 49% and the number of extant winter colonies (populations) by 81% (Figure 5.2, Table A–3A1). There has also been a noticeable shift towards smaller colony sizes, with a 96–100% decline in the number of large hibernacula (≥100 individuals) (Figure 5.3). Although the declines are widespread, the magnitudes of the winter declines vary spatially (Figure 5.4). In the Eastern Hardwoods, the core of NLEB range, abundance declined by 56% and the number of sites by 88%. Abundance and the number of sites declined in the remaining 4 RPUs (87% and 82% - East Coast RPU, 90% and 44% - Midwest RPU, 24% and 70% - Southeast RPU, and 0% and 40% - Subarctic RPU, respectively; Table A–3B3). Across all RPUs, the potential of population growth is low; the probability of RPU growth rates (λ) ≥1 ranges from 0 to 11% (Table A-3B2).

Declining trends in abundance and occurrence are also evident across much of NLEB summer range. Based on derived rangewide summaries from Stratton and Irvine (2022, p. 102), rangewide occupancy has declined by 80% from 2010–2019 (Table A-3B4, Figure 5.7). Although these declines attenuate westward, the probability of occupancy declined in all RPUs (Table A-3B4). Similarly, Whitby et al. (2022, p. 160), using data collected from mobile acoustic transects, found a 79% decline in rangewide relative abundance from 2009–2019. Measurable declines were also found in the Midwest RU (91%) followed by the Eastern Hardwoods (85%), East Coast (71%), and Southeast (57%) RPUs (Table A–3B4). Data were not analyzed in the Subarctic RPU due to a lack of observations. Finally, Deeley and Ford (2022, p. 18, 21–23) observed a significant decrease in mean capture rate post-WNS arrival. Estimates derived from their results indicted a 43–77% decline in summer mist net captures compared pre and post arrival of WNS (Table A–3B4).

Future projections based on current conditions - Collectively, these data indicate NLEB has declined and given the declining trajectories, will continue to decline. Future projections from the BatTool, assuming no further WNS spread nor increases in wind capacity (current stressor conditions), show sharp declines in rangewide abundance, number of hibernacula, and spatial extent into the future.

- By 2030 (~ 1 generation), rangewide abundance declines by 95% (CI 75–99%; Figure 5.2).
- The number of extant hibernacula declines by 99%, with 11 of the 737 historically occupied hibernacula extant by 2030 (Figure 5.5) and 1 extant hibernaculum by 2040.
- The winter colony sizes also become reduced, with the number of large hibernacula (≥100 bats) declining from 53 in 2000, 20 in 2020, to 1 hibernaculum (98% decline from 2020) by 2030 (Figure 5.3).
- Subsequent to declines in the number of hibernacula, NLEB's known winter range declines by 75% (Table A-3B1), with the vast majority (90%) of individuals becoming concentrated in a smaller number of hibernacula, going from 66 hibernacula in 2000, 29 in 2020 to 6 by 2030.

The projected declines are widespread across the RPUs.

- Median hibernacula abundances in the Southeast, East Coast, Midwest, and Subarctic RPUs decline to 2–16 (CI 0–4,118) or 99–100% decline, with corresponding low probabilities of persistence by 2030 (Tables A-3B3, Figure 5.6).
- In the Eastern Hardwoods RPU, median abundance declines 99%, with bats persisting in 10 hibernacula by 2030. Of the projected extant hibernacula, 1 is projected to be large (≥100 individuals; Figure 5.3).

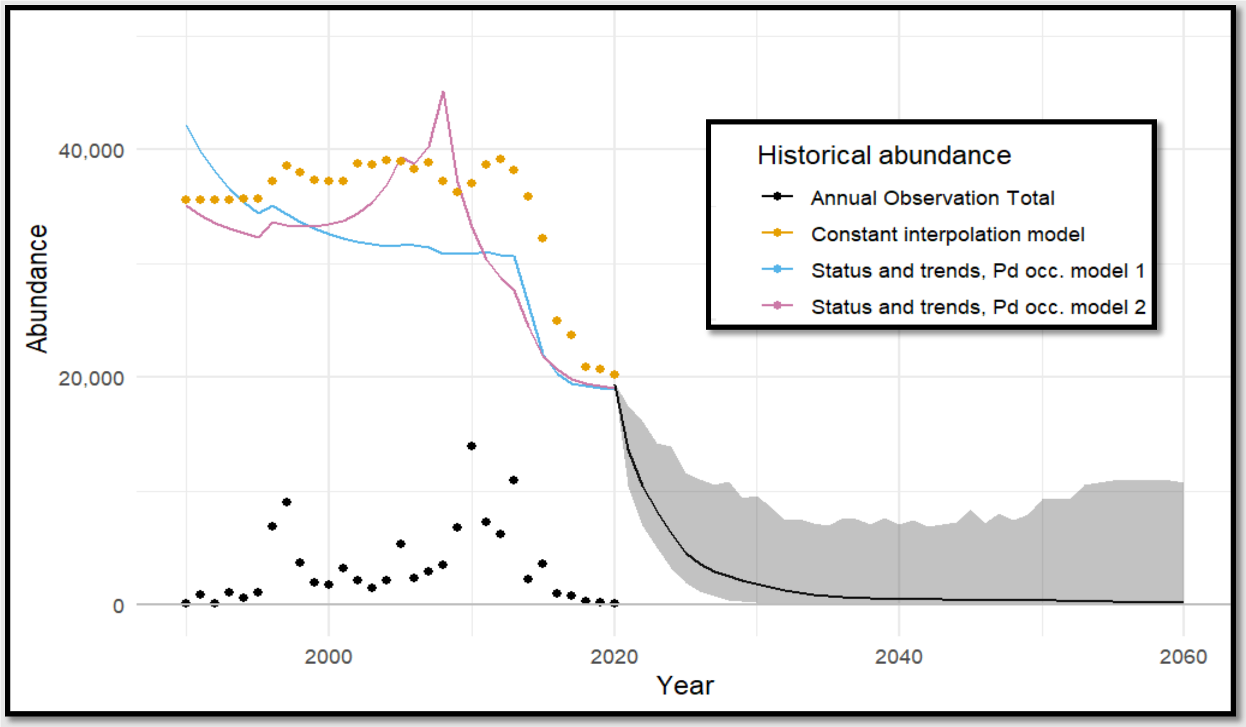

Figure 5.2. Median projected rangewide abundance (black line) and 90% CI (gray) given CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity). Abundance from 1990 – 2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status & trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

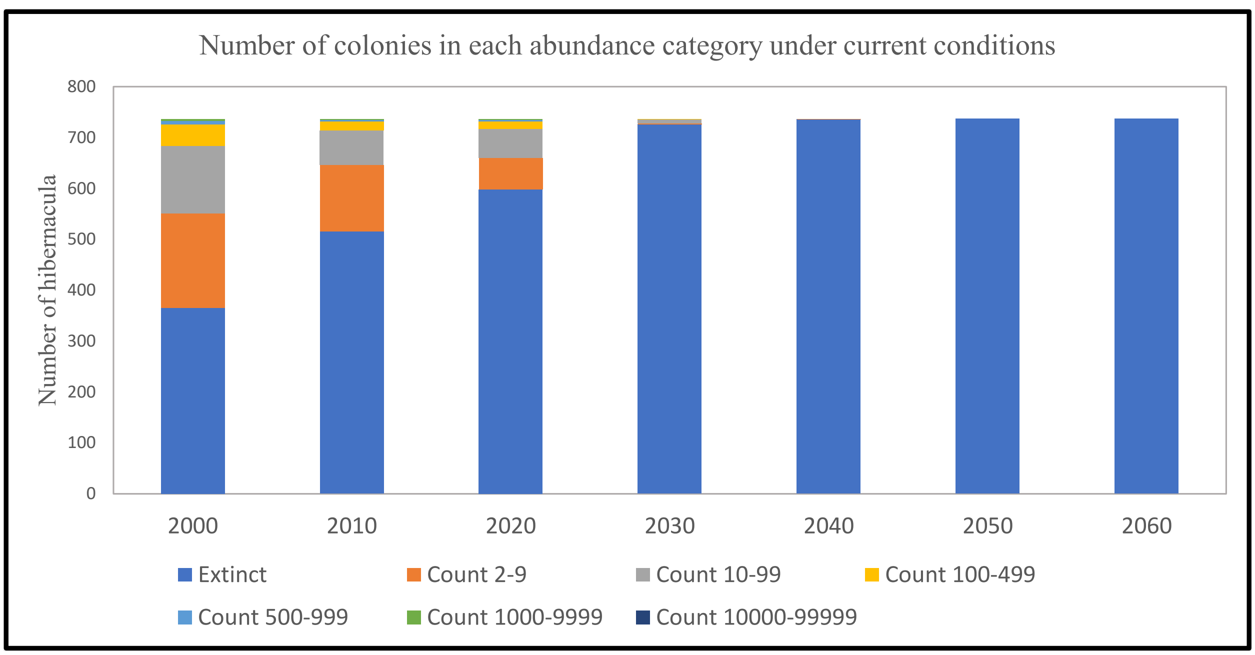

Figure 5.3. The number of hibernacula in each colony abundance category under current state conditions.

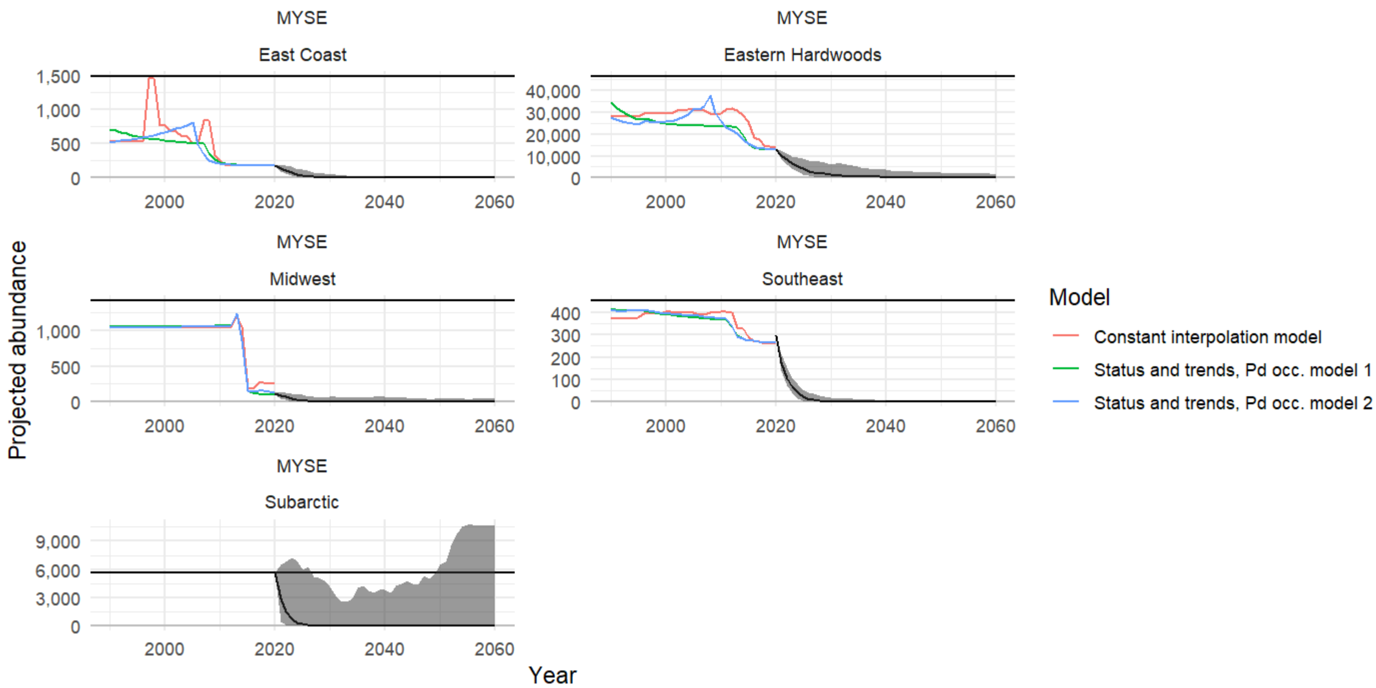

Figure 5.4. Median projected RPU abundance (black line) and 90% CI (gray) under CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity for the 3 RPUs. Abundance from 1990–2020 derived from winter colony count data using a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).

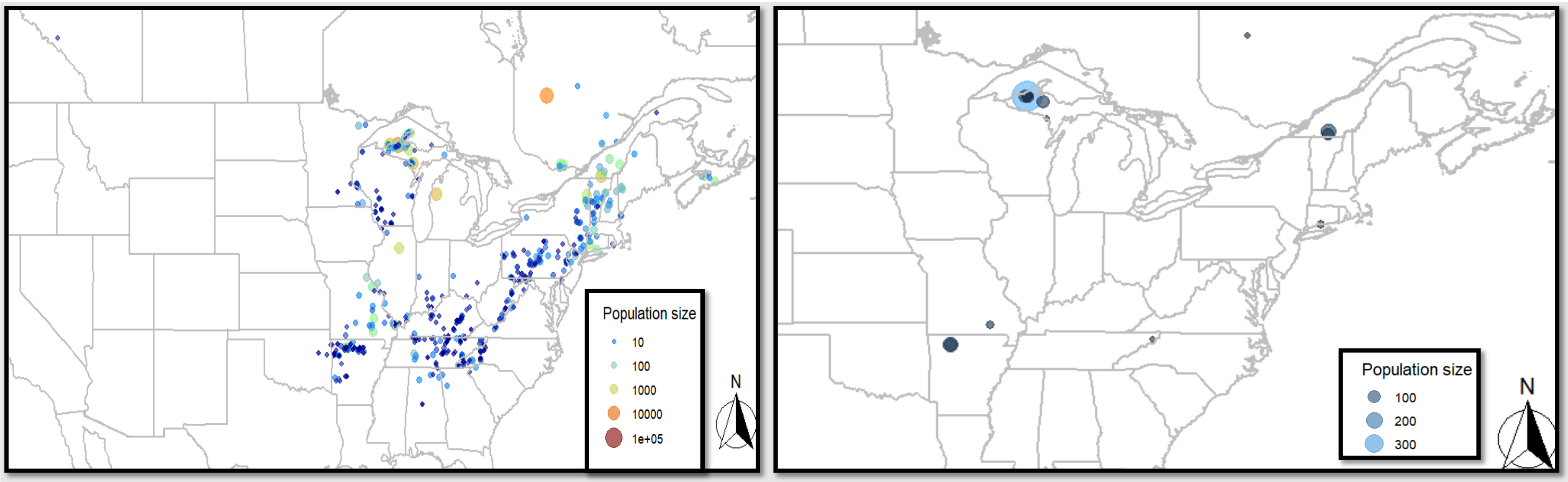

Figure 5.5. NLEB extant hibernacula at year 2000 (left) and projected at 2030 (right) given CURRENT state conditions. Color and size reflect median hibernacula abundance.

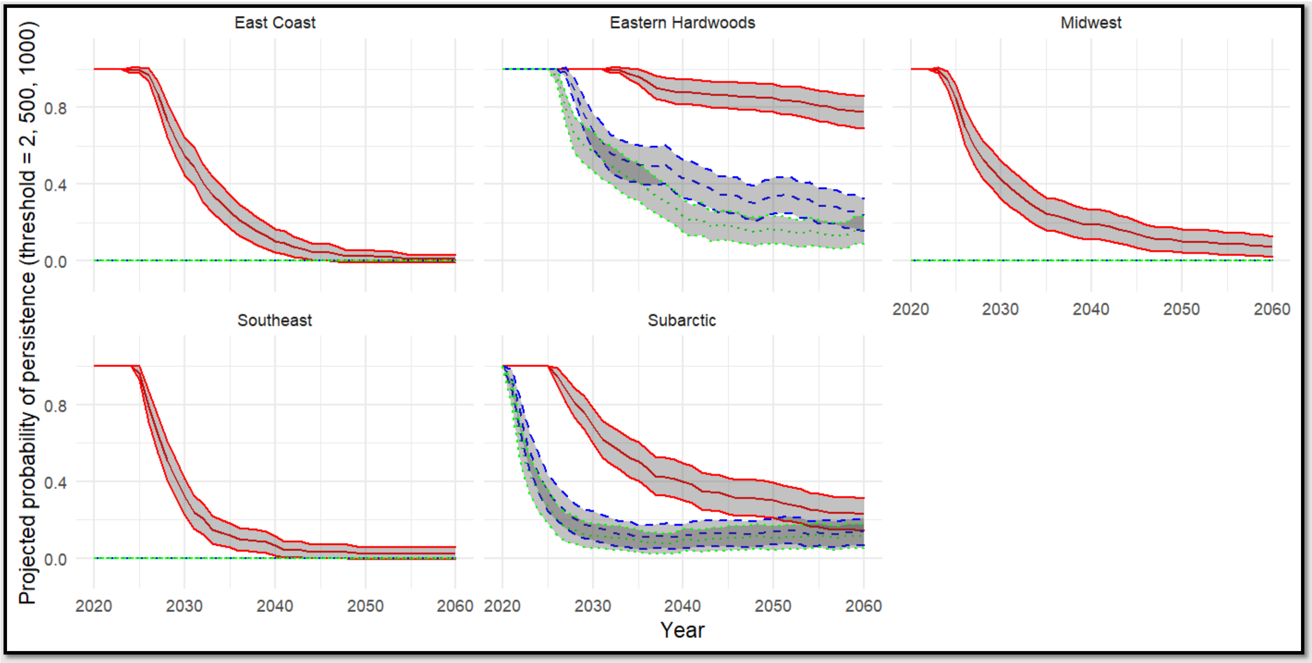

Figure 5.6. Probability of RU-abundance remaining above X individuals given CURRENT state conditions, x= 2 bats (red), x=500 bats (blue), and b) x=1000 bats (green).

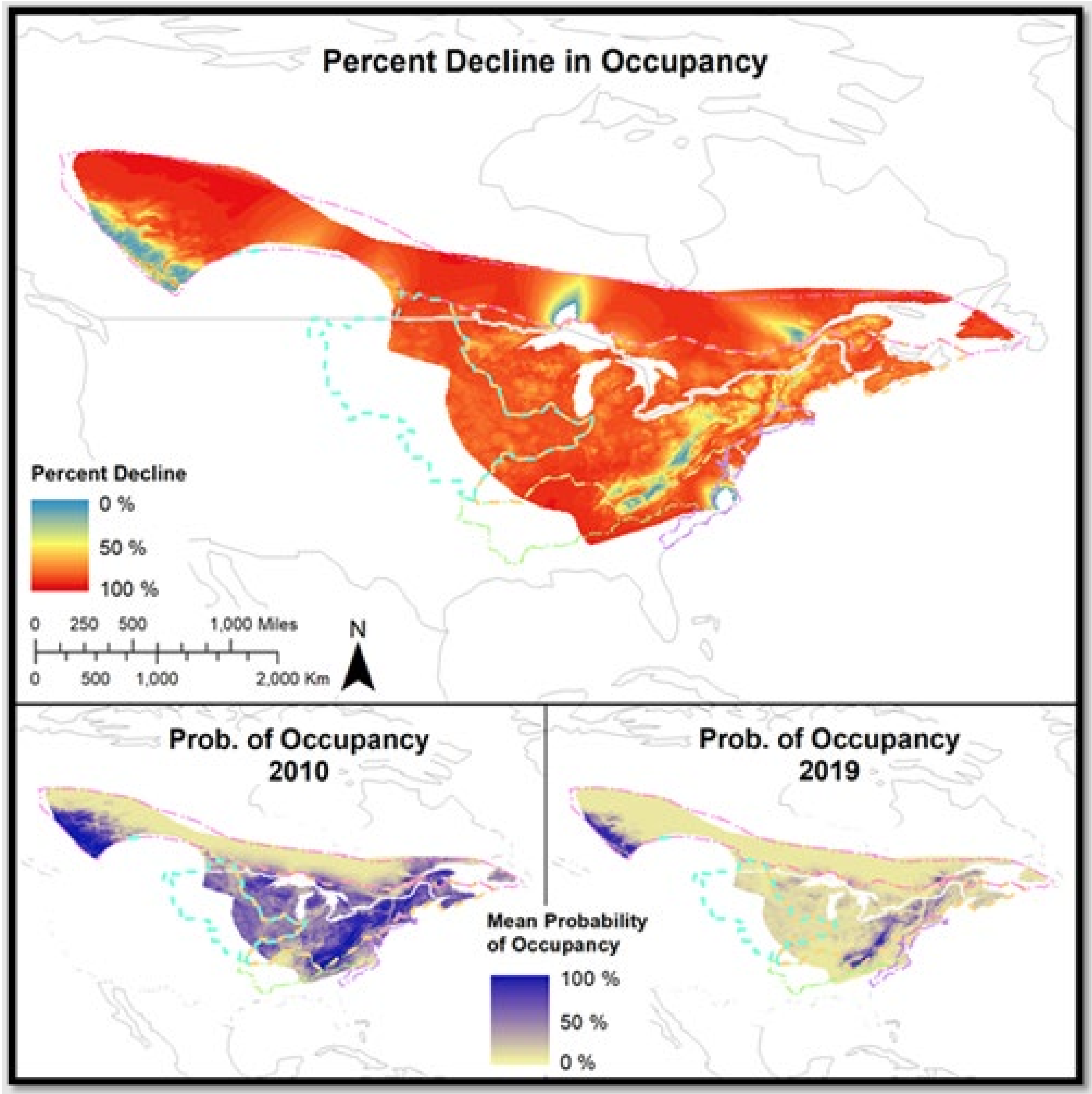

Figure 5.7. Predicted percent decline in probability of occupancy (top) and probability of NLEB summer occupancy in 2010 (bottom left) and 2019 (bottom right) based on data collected from stationary and mobile transect acoustic monitoring and capture records summarized at the 10km x 10km NABat grid cell (Stratton and Irvine 2022, entire). Dotted boundaries correspond to RPUs. Cooler colors represent lower percent declines (top panel) or higher probability of occupancy (bottom panels).

CHAPTER 6—FUTURE CONDITION

Future viability is the ability of NLEB to sustain healthy populations into the future given its current demographic condition and future condition of the influences (Figure 6.1). To assess NLEB future viability, we again used the BatTool to project hibernaculum abundance over time given projected *Pd* spread and wind energy build-out (see Chapter 4, *Future Scenarios* subsection, for further description). Projection of future number, distribution, and health of populations is needed to understand NLEB's future ability to withstand normal stochasticity, stressors, catastrophic events, and novel environmental changes (i.e., its viability under future influences). In this chapter, we describe the projected number, health, and distribution of NLEB given future state conditions (i.e., future *Pd* occurrence and future installed wind energy capacity) and describe the viability implications under future influences in Chapter 7.

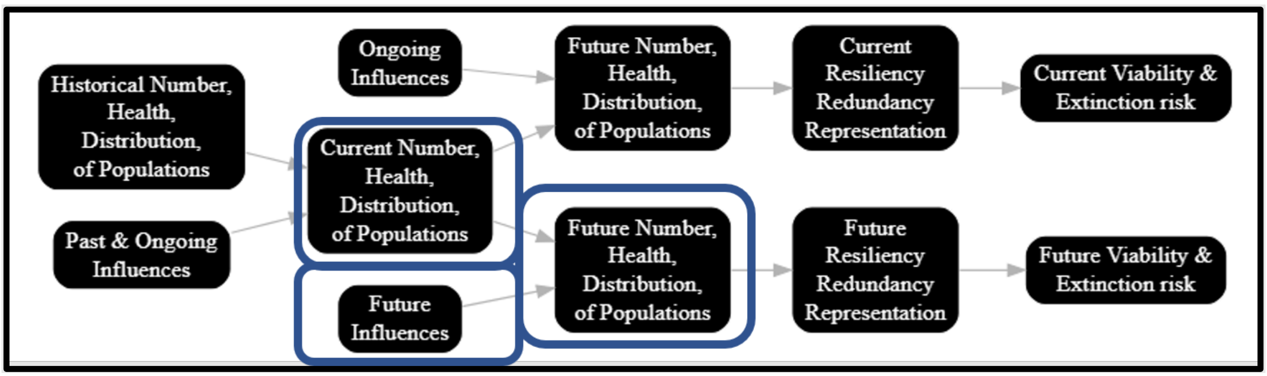

Figure 6.1. Highlighting (blue rectangles) current step in our analytical framework

Under both future scenarios, the declines worsen precipitously.

- Median rangewide abundance declines 95% by 2030 (74–100% CI) and reaches 99% by 2060 (67–100% CI). Under the future scenarios, the decline trajectory continues (despite no impacts due to WNS being applied 15 years after *Pd* arrival; Figure 6.2, Table A–3C1).
- The number of extant hibernacula decline to 9 by 2030 and 0 hibernacula by 2050 (Figure 6.3, Table A–3C1).
- Colony sizes continue to shift towards smaller sizes, with 89% of the projected extant colonies in 2030 having fewer than 100 bats (Figure 6.4).
- Spatially, NLEB's winter range declines by 75% by 2030 (100% by 2040) (Table A-3C1).

As projected under the future conditions, declines are widespread and there is limited chance for persistence.

- Median abundances in the Southeast, East Coast, Midwest, and Subarctic RPUs decline to 2–22 (CI 2–6,199) or 99–100% decline, with corresponding low probabilities of persistence by 2030 (Figure 6.6, Table A-3C3).

- In the Eastern Hardwoods RPU, median abundance declines 95%, with bats persisting in 8 hibernacula by 2030. By 2060, all populations at all hibernacula are projected to be extinct (Figure 6.5, Table A-3C2).

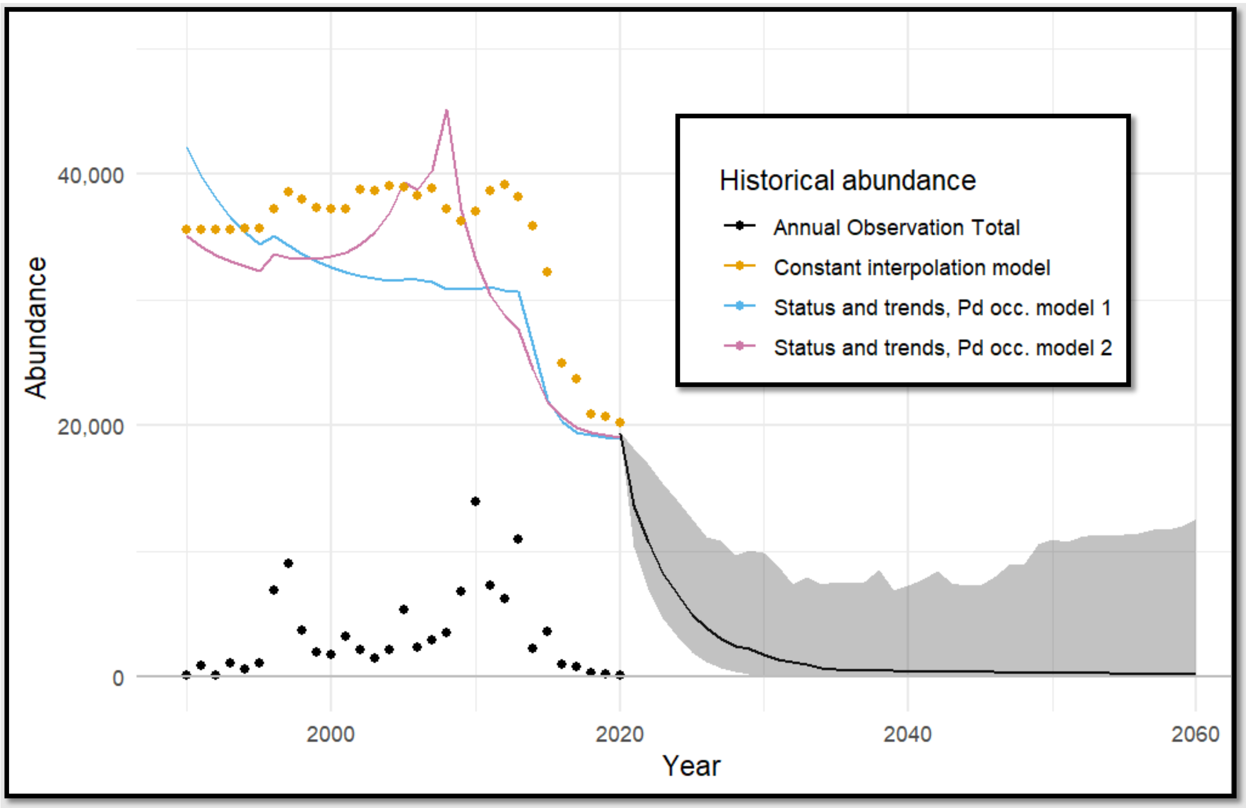

Figure 6.2. Projected median rangewide abundance (black line) and 90% CI (gray shading) under FUTURE state conditions. Abundance from 1990–2020 derived from raw data (black dots) using a) constant interpolation (yellow dots), b) status & trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

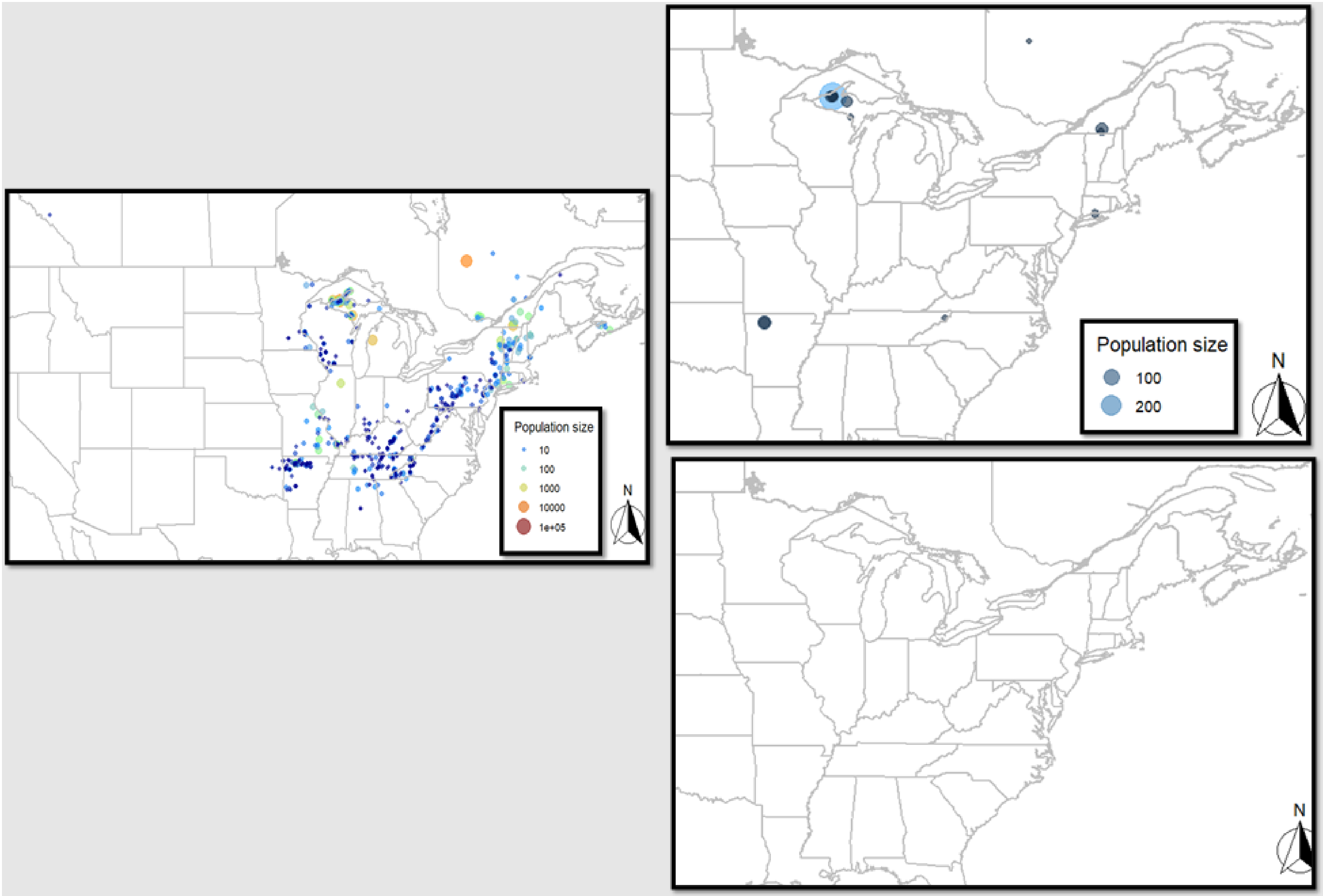

Figure 6.3. NLEB extant hibernacula in 2000 (left) and projected 2030 (upper right) and 2060 (bottom right) given FUTURE state conditions. Color and size reflect medium hibernacula abundance.

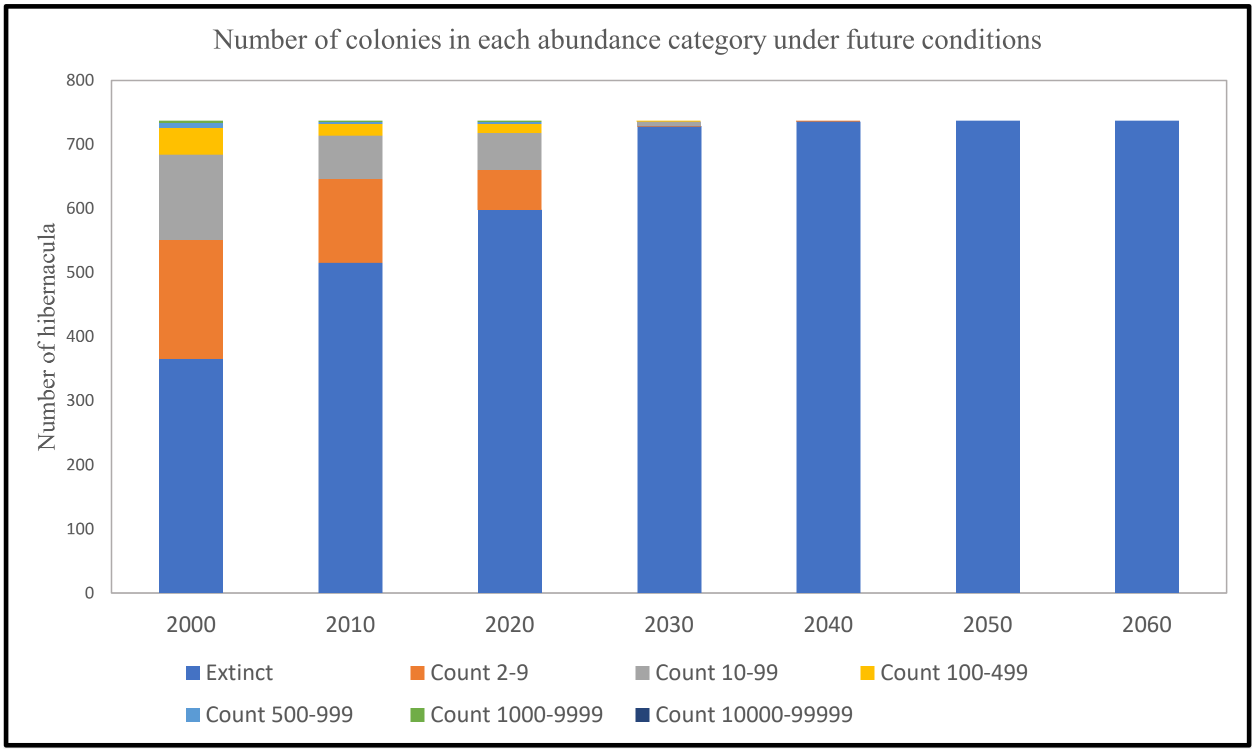

Figure 6.4. The projected number of hibernacula in each colony abundance category under FUTURE state conditions.

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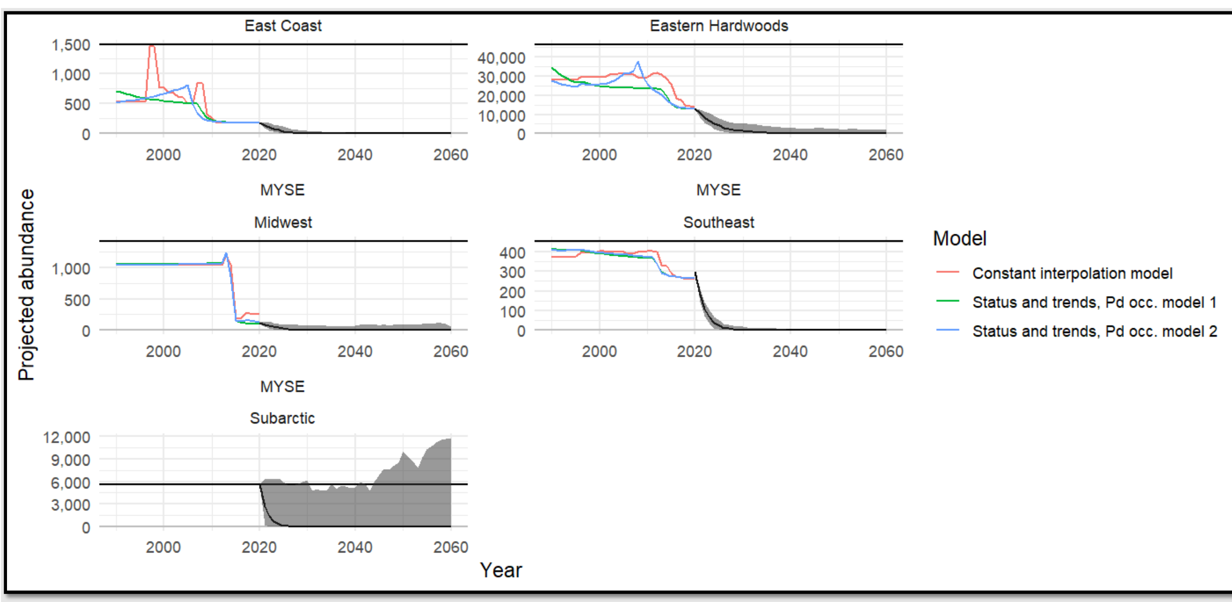

Figure 6.5. Projected median (black line) and 90% CI (gray shading) for RPU abundance under FUTURE state conditions for the 5 RPUs. Abundance from 1990 –2020 derived from raw data using a) constant interpolation (red line), b) status & trend model

informed by Pd occurrence model 1 (green line) and c) status & trend model informed by Pd occurrence model 2 (blue line).

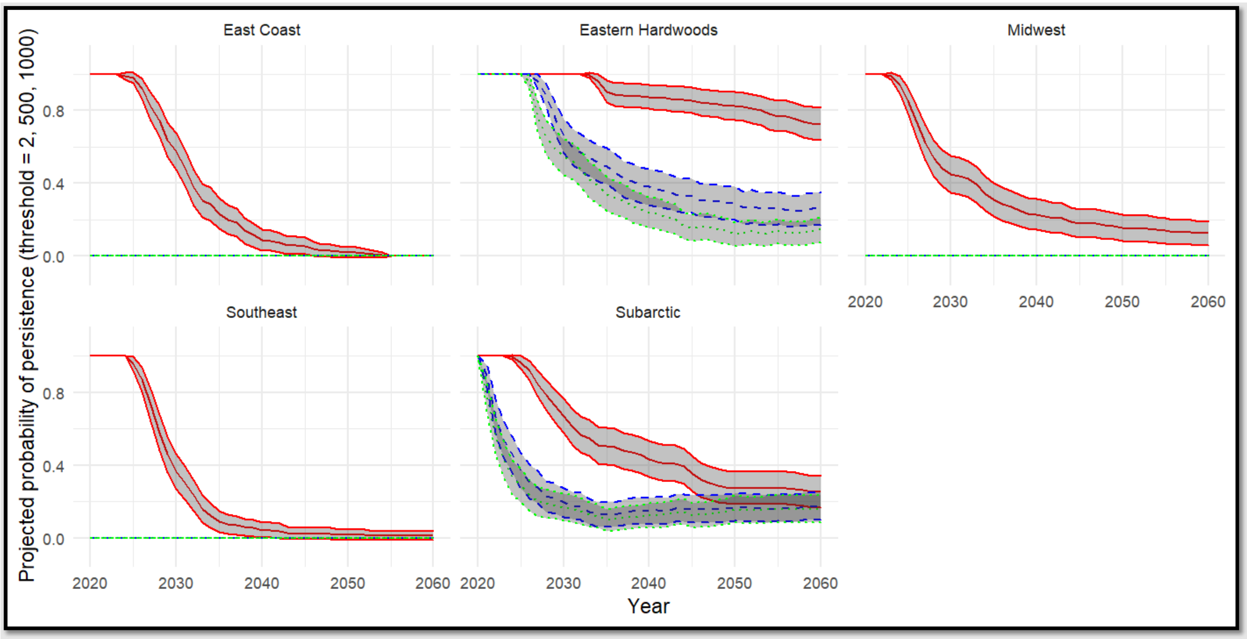

Figure 6.6. Probability of RPU-abundance remaining above X individuals given FUTURE state conditions, x= 2 bats (red), x=500 bats (blue), and x=1000 bats (green).

Habitat Loss and Climate Change

As discussed previously, we did not incorporate habitat loss and the effects of climate change into our quantitative modeling efforts (i.e., not included in the projections depicted in Figures 6.2– 6.6). Ongoing effects from habitat loss and climate change likely continue into the future and may even be exacerbated based on reduced abundance and distribution anticipated under our current and future scenarios. See Table 4.5 for a description of the scope, severity, and impact of future habitat loss and climate change impacts. Additionally, future impacts from habitat loss and climate change are discussed more thoroughly in Appendix 4.

CHAPTER 7—SPECIES VIABILITY

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences for NLEB viability (Figure 7.1). NLEB viability is influenced by the number, health, and distribution of populations. Across the range and within all RPUs, NLEB abundance and distribution has decreased. Multiple data types and analyses indicate downward trends in NLEB population abundance and distribution over the last 14 years (2006–2020; Table 7.1), and we found no evidence to suggest that this downward trend will change in the future (Figure 7.2). As is the case for all species status assessments we do not have perfect information on NLEB's occurrence, but the best available data suggest that bats at unknown hibernacula will undergo similar declines observed at known winter colonies. We outline the key uncertainties in our analyses and our resolution of them in Appendix 1.

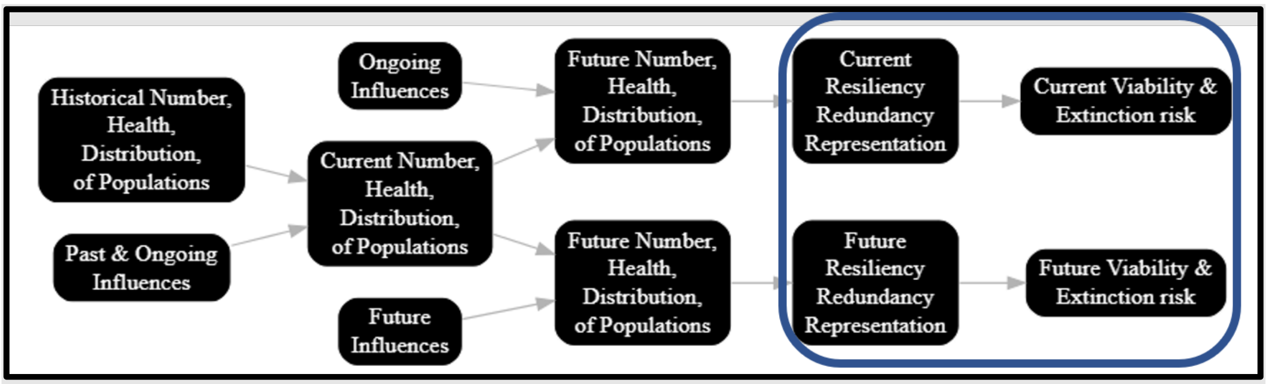

Figure 7.1. Highlighting (blue rectangle) the current step in our analytical framework.

Table 7.1. Summary of recent NLEB population trends from multiple data types and analyses. Winter Colony analysis – derived from Wiens et al. (2022, entire) data; Summer Occupancy analysis –Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire).[1] No data available.

Representation Unit	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Southeast	-24%	-85%	-47%	-50%
Eastern Hardwoods	-56%	-78	-43%	-87%
Subarctic	-0%	-63%	-[1]	-[1]
Midwest	-90%	-87%	-77%	-99.9%
East Coast	-87%	-79%	-43%	-69%
Rangewide	-49%	-80%	-43–77%	-79%

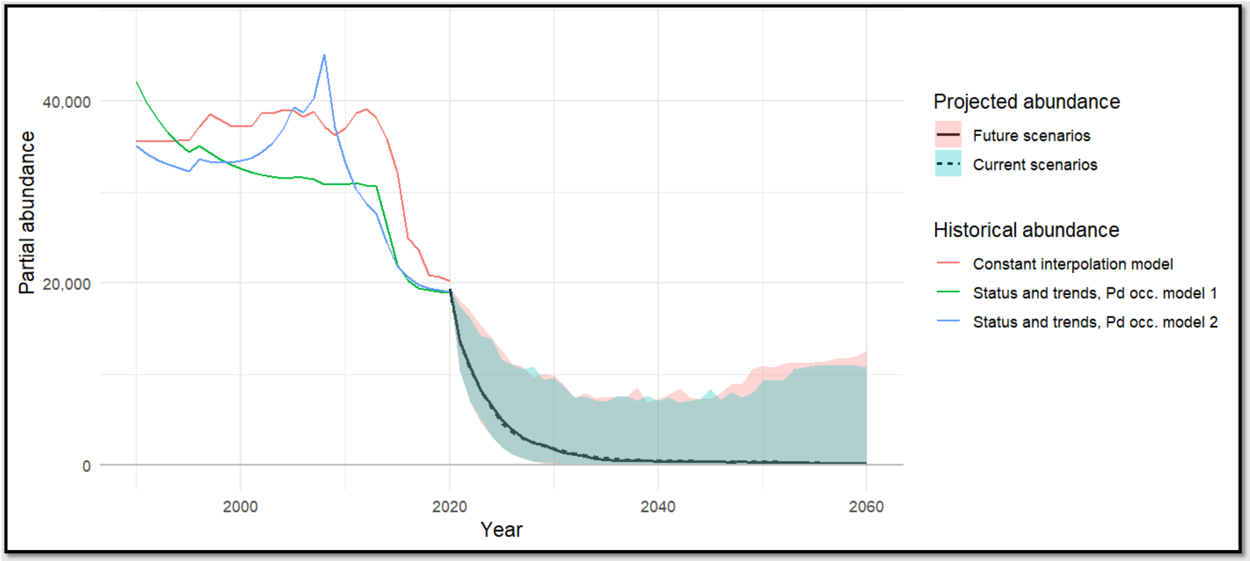

Figure 7.2. The projected NLEB abundance over time given current (blue) WNS spread and installed wind capacity and plausible future scenarios (pink) for WNS spread and increased installed wind energy capacity. The dotted and solid lines represent the median abundance under current and future scenarios, respectively. Historical abundance from 1990 –2020 derived from a) constant interpolation (red), b) status & trend model informed by Pd occurrence model 1 (green line) and c) status & trend model informed by Pd occurrence model 2 (blue line).

The viability of a species depends upon its ability to sustain populations in the face of normal environmental and demographic stochasticity, catastrophes, and novel changes in its environment. For example, demographically and physically healthy populations better withstand and recover from environmental variability and disturbances. Additionally, populations spread across heterogeneous conditions are unlikely to be exposed at the same time to poor environmental conditions, thereby guarding against synchronous population losses. Similarly, species with genetically healthy populations (large N_e, which begets genetic diversity) spread across the breadth of genetic and phenotypic diversity preserve a species' adaptive capacity, which is essential for adapting to their continuously changing environment (Nicotra et al. 2015, p. 1269). Without such variation, species are less responsive to change and more prone to extinction (Spielman et al. 2004, p. 15263). Lastly, having multiple healthy populations widely distributed guards against losses of adaptive diversity and RPU-level extirpation in the face of catastrophic events.

We quantitatively assessed NLEB's current viability by projecting the species' abundance and distribution given current WNS occurrence (no further spread) and current installed wind energy capacity, and future viability given future plausible scenarios of further WNS spread and increased wind energy capacity. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. All existing data and our qualitative and quantitative analyses suggest that NLEB's viability has and will continue to steeply decline over time under the current and plausible future conditions.

Unquestionably, WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future (Table 7.2). Currently, WNS occurs across 59% of NLEB's range (Cheng et al. 2021, p. 7) and is impacting 99–100% hibernacula (Wiens et al. 2022, pp. 226–229, 231–247). In addition, WNS is predicted to reach 100% of the species' range in the U.S. by 2025 (Wiens et al. 2022, pp. 226–229). Prior to WNS, NLEB was abundant and widespread, and abundance and occupancy were generally stable (Cheng et al. 2022, p. 204). WNS impacts have resulted in most winter colonies experiencing a 97–100% decline in abundance compared to historical conditions (Cheng et al. 2021, entire).

Wind energy related mortality, although not currently acting as a driver in NLEB's viability, is projected to be more impactful in the future as it will increase in pervasiveness and severity (Table 7.2). Based on 2020 wind build-out, an estimated 38 to 150 (mean = 122) NLEBs are killed annually at wind facilities and annual mortality is projected to increase to 202 to 2,926 individuals by 2050 under the future low and high build-out scenarios, respectively (Figures 4.10 and 4.11, Tables 4.1 and 4.4). Wind related mortality is discernible, particularly in future scenarios, even with ongoing declines from WNS (Figure A-1B2; see also Whitby et al. 2022, pp. 151–153). NLEB abundance is projected to decline 18 and 77% from 2030 to 2060 from wind related mortality alone under current conditions and from 28 to 80% under the future scenarios. Consequently, mortality from wind turbines likely has and will continue to cause detectable declines in NLEB abundance.

Although we consider habitat loss pervasive across NLEB range, impacts to NLEB and its habitat are often realized at the individual or colony level. Loss of hibernation sites (or modifications such that the site is no longer suitable) can result in impacts to winter colonies. Impacts from forest loss (e.g., roosting or foraging habitat) vary depending on the timing, location, and extent of the removal. Given how common and wide-ranging NLEB was throughout much of its range prior to the arrival of WNS, we assume the range-wide magnitude of impact from habitat loss was low. However, as NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and summer colonies and remaining populations are anticipated to be less resilient), habitat loss at occupied sites will vary from slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between low to very high (Table 7.2).

Climate change impacts are challenging to describe for wide-ranging species, such as NLEB. The changing climate has and will likely continue to have a multitude of impacts on species throughout North America (Foden et al. 2018, p. 9). Despite being pervasive; however, we believe the rangewide magnitude of impact is currently low (Table 7.2). In addition, there are questions about whether some negative effects are currently offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). Although there may be some offsetting of effects under current climate conditions, increasing negative impacts are anticipated in the future (Table 7.2). Increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase, leading to increased NLEB mortality and reduced reproductive success. As mentioned above, as NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies) and populations anticipated to be less resilient, effects from climate change may be more impactful than if the populations were well distributed and robust.

Table 7.2. Threat (impact) level for the primary influences currently and projected future low and high impact scenarios.

	WNS	Wind Mortality	Habitat Loss		Climate Change
Current	Very High	Medium	Low		Low
Low Impact	Very High	Medium	Low	Very High	Medium
High Impact	Very High	Medium	Low	Very High	Medium

While we focused our analyses on ongoing and anticipated effects from WNS, wind, climate change and habitat loss, we also recognize that novel threats (e.g., new disease or invasive species) may emerge for NLEB. NLEB's mobility and roost-shifting behaviors provide mechanisms for individual bats to respond to changes in temperature, prey availability and roost suitability. However, as discussed in Chapter 2 and Appendix 2-B, temperate zone insectivorous bats including NLEB have several inherent traits that limit their ability to respond to changes in the environment, especially to rapid changes. These include their high site fidelity (winter and summer), specialized winter habitat requirements and summer roost microclimate needs, and low reproductive output. We have already observed the extremely limited ability for NLEB to respond to the novel threat WNS.

Viability under Current Conditions

Under current conditions, NLEB abundance, number of occupied hibernacula, spatial extent, probability of persistence, summer habitat occupancy (measured by bat captures and acoustic recordings) across the range and within all RPUs are decreasing (Chapter 5 and Table 7.1). Since the arrival of WNS, NLEB abundance steeply declined, with most (91%) winter colonies having fewer than 100 individuals. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, NLEB's ability to recover from these low abundances is limited given their low reproduction output (1 pup per year). Therefore, NLEB's resiliency is greatly compromised in its current condition. Additionally, NLEB's spatial extent is projected to decline, with 75% reduction by 2030. As NLEB's abundance and spatial extent decline, NLEB will also become more vulnerable to catastrophic events

In addition to reduced redundancy and resiliency, NLEB's representation has also been reduced. As explained above, NLEB's capacity to adapt is constrained by its life history and the level of its intraspecific diversity (e.g., genetic, phenotypic, behavioral, ecological variability). The steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reduced NLEB adaptive capacity. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit NLEB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen NLEB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further *Pd* spread and additional wind energy development, NLEB's viability is likely to rapidly decline over the next 10 years (Figures 7.2 and 7.3).

Viability under Future Scenarios

Under the projected range of plausible future scenarios, WNS spread reaches close to 100% of NLEB's entire range (Wiens et al. 2022, pp. 226–229) and wind energy related mortality increases by 66% to 2,298% (Udell et al. 2022, entire; see Table 4.4). By 2060, NLEB abundance declines by 99% (Figure 7.2) and the number of extant hibernacula declines by 100% (Figure 7.3). Under the future scenario, by 2040, only one hibernaculum is projected to remain in the Eastern Hardwoods RPU. By 2050, no hibernacula remain in any of the RPUs (Figure 7.3). Given the projected low abundance and the few number and restricted distribution of winter colonies, NLEB's currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

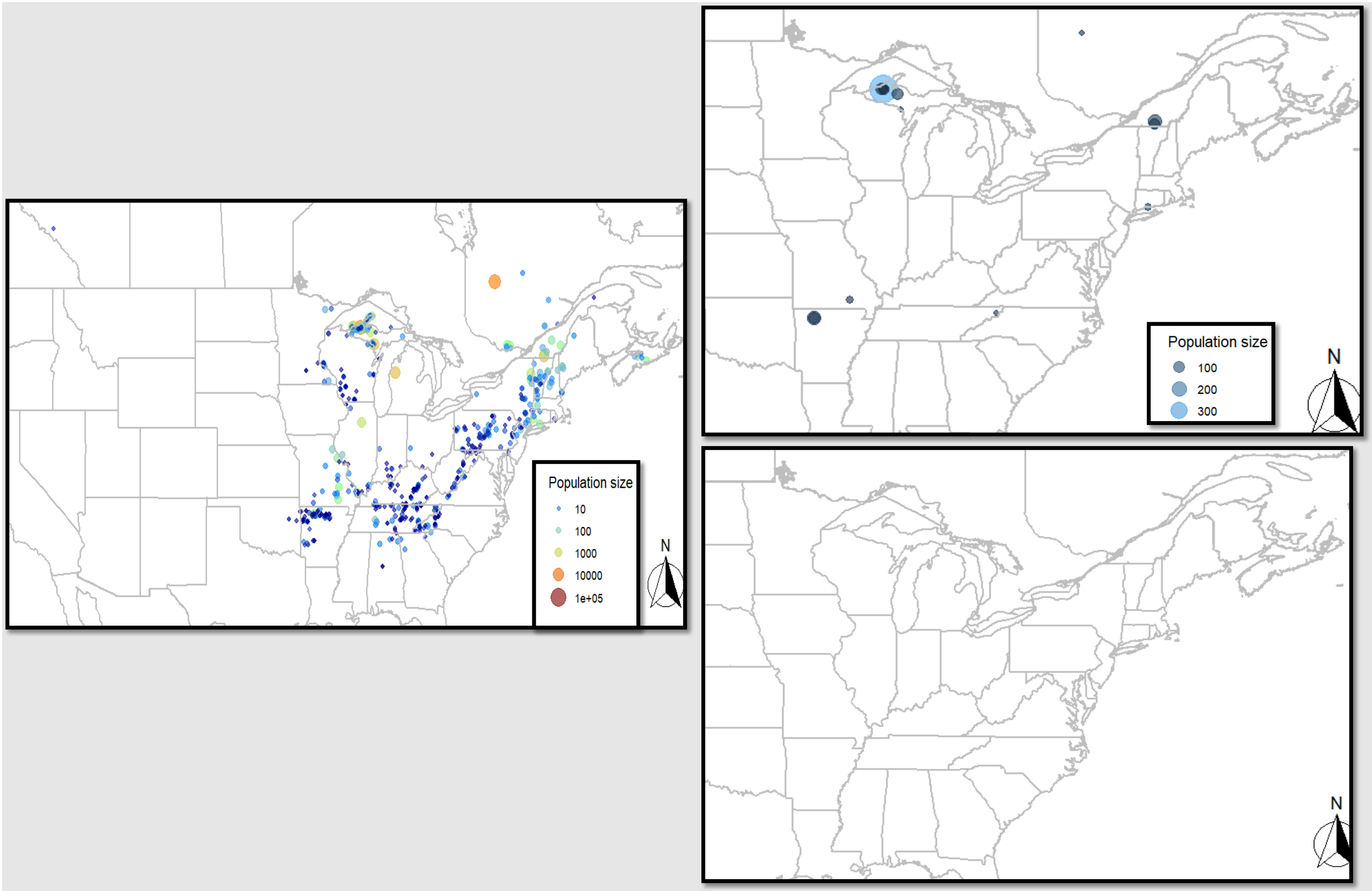

Figure 7.3. Projected change in NLEB winter distribution over time: 2000 (far left); 2030 under current conditions (top right), and 2060 under future conditions (bottom right).

REFERENCES CITED[6]

Adams, R.A. 2010. Bat reproduction declines when conditions mimic climate change projections for western North America. Ecology 91(8):2437–2445.

Adams, R.A., and M.A. Hayes. 2008. Water Availability and Successful Lactation by Bats as Related to Climate Change in Arid Regions of Western North America. Journal of Animal Ecology 77(6):1115–1121.

Alves, D.M.C.C, L.C. Terribile, and D. Brito. 2014. The Potential Impact of White-Nose Syndrome on the Conservation Status of North American Bats. PLoS ONE 9(9): e107395. doi:10.1371/journal.pone.0107395.

Amelon, S., and D. Burhans. 2006. Conservation assessment: *Myotis septentrionalis* (northern long-eared bat) in the eastern United States. Pages 69–82 *in* Conservation assessments for five forest bat species in the eastern United States, Thompson, F. R., III, editor. U.S. Department of Agriculture, Forest Service, North Central Research Station, General Technical Report NC-260. St. Paul, Minnesota. 82pp.

American Wind Energy Association (AWEA). 2020. Wind Facts at a Glance. Available at: https://www.awea.org/wind-101/basics-of-wind-energy/wind-facts-at-a-glance.

American Wind Wildlife Institute (AWWI). 2020. AWWI Technical Report: 2nd Edition: Summary of Bat Fatality Monitoring Data Contained in AWWIC. Washington, DC. Available at www.awwi.org. © 2020 American Wind Wildlife Institute.

Arnett, B., J. P. Hayes, and M. M. P. Huso. 2006. Patterns of preconstruction bat activity at a proposed wind facility in south-central Pennsylvania. An annual report submitted to the Bats and Wind Energy Cooperativer. Bat Conservation Internationa, Austin, TX. 46 pp. Available at: https://batsandwind.org/wp-content/uploads/precon_pa.pdf

Arnett, E B, W.K Brown, W.P. K Fiedler, B.L Hamilton, T.H. Henry, A. Jain, G.D. Johnson, J. Kerns, R. R. Koford, C. P. Nicholson, T. J. O'Connell, M. D. Piorkowski, and R. D. Tankersley, Jr. 2008. Patterns of bat fatalities at wind energy facilities in North America. Journal of Wildlife Management 72:61–78.

Arnett E.B., C.D. Hein, M.R. Schirmacher, M.M.P. Huso, and J.M. Szewczak. 2013. Evaluating the effectiveness of an ultrasonic acoustic deterrent for reducing bat fatalities at wind turbines. PLoS ONE 8(6):e65794.

Anthony, E.L.P., M.H. Stack, and T.H. Kunz. 1981. Night roosting and the nocturnal time budget of the little brown bat, *Myotis lucifugus*: Effects of reproductive status, prey density, and environmental conditions. Oecologia 51:151–156.

[6] Includes references from body of document and appendices.

Arnold, B.D. 2007. Population Structure and Sex-biased Dispersal in the Forest Dwelling Vespertilionid Bat, *Myotis septentrionalis*. The American Midland Naturalist, 157(2):374–384.

Badin, H.A. 2014. Habitat selection and roosting ranges of northern long-eared bats (*Myotis septentrionalis*) in an experimental hardwood forest system. Muncie, IN: Ball State University. M.S. thesis.

Baerwald, E.F., G. H. D'Amours, B.J. Klug, and R.M.R.Barclay. 2008. Barotrauma is a significant cause of bat fatalities at wind turbines. Current Biology, 18(16):695–696.

Baijnath-Rodino, J.A., C. Duguay, and E. LeDrew. 2018. Climatological trends of snowfall over the Laurentian Great Lakes Basin. International Journal of Climatology 38:3942–3962.

Bailey, R.G. 2016. Ecoregions of the United States. [online] https://www.fs.usda.gov/rmrs/ecoregions-north-america [Accessed 1 March, 2021].

Barbour, R.W., and W.H. Davis. 1969. Bats of America. The University of Kentucky Press, Lexington, Kentucky. 311pp.

Barclay, R.M.R., J. Ulmer, C.J.A. MacKenzie, M.S. Thompson, L. Olson, J. McCool, E. Cropley, and G. Poll. 2004. Variation in the reproductive rate of bats. Canadian Journal of Zoology 82:688–693.

Barclay, R.M.R., and A. Kurta. 2007. Ecology and Behavior of Bats Roosting in Tree Cavities and Under Bark. Chapter 2: pp. 17–60 *in* Bats in Forests: Conservation and Management, M.J. Lacki, J.P. Hayes, and A. Kurta, editors. The Johns Hopkins University Press, Baltimore, Maryland, 352pp.

Beever, E.A., J. O'Leary, C. Mengelt, J.M West, S. Julius, N. Green, D. Magness , L. Petes, B. Stein , A.B. Nicotra, and J.J. Hellmann. 2016. Improving conservation outcomes with a new paradigm for understanding species' fundamental and realized adaptive capacity. Conservation Letters, 9:131–137.

Belwood, J.J. 2002. Endangered bats in suburbia: observations and concerns for the future. In The Indiana bat: biology and management of an endangered species (A. Kurta and J. Kennedy, editors). Bat Conservation International, Austin, Texas, 193–198.

Bennett, V.J., and A.M. Hale. 2018. Texturizing wind turbine towers to reduce bat mortality DE-EE007033. Texas Christian University. Final Technical Report. Available at: https://www.osti.gov/servlets/purl/1502997.

Berkeley Lab. 2020. Electricity Markets and Policy. Wind Technologies Market Report. https://emp.lbl.gov/wind-technologies-market-report. Accessed 21 May 2021.

Bernard, R.F. and G.F. McCracken. 2017. Winter behavior of bats and the progression of white-nose syndrome in the southeastern United States. Ecology and Evolution 7:1487–1496. https://doi.org/10.1002/ece3.2772

Bernard, R.F., J.D. Reichard, J.T.H. Coleman, J.C. Blackwood, M.L. Verant, J.L. Segers, J.M. Lorch, J. Paul White, M.S. Moore, A.L. Russell, R.A. Katz, D.L. Lindner, R.S. Toomey, G.G. Turner, W.F. Frick, M.J. Vonhof, C.K.R. Willis, and E.H.C. Grant. 2020. Identifying research needs to inform white-nose syndrome management decisions. Conservation Science and Practice 2(8):e220.

Berthinussen, A., O.C. Richardson, and J.D. Altringham. 2021. Bat Conservation: Global Evidence for the Effects of Interventions. Conservation Evidence Series Synopses. University of Cambridge, Cambridge, UK. Available at: www.conservationevidence.com.

Bird Studies Canada, Canadian Wind Energy Association, Environment and Climate Change Canada and Ontario Ministry of Natural Resources and Forestry. 2018. Wind Energy Bird and Bat Monitoring Database Summary of the Findings from Post-construction Monitoring Reports. 56 pp. Available at: https://www.bsceoc.org/resources/wind/2018_Database_Summary_Report.pdf.

Blehert, D.S., A.C. Hicks, M.J. Behr, C.U. Meteyer, B.M. Berlowski-Zier, E.L. Buckles, J. Coleman T.H., S.R. Darling, A. Gargas, R. Niver, J.C. Okoniewski, R.J. Rudd, and W.B. Stone. 2009. Bat White-Nose Syndrome: an emerging fungal pathogen? Science, 323:227.

Bohrman, J., and D. Fecske. 2013. White-Nose Syndrome Surveillance and Summer Monitoring of Bats at Great Swamp National Wildlife Refuge, Morris County, New Jersey. A final report prepared for United States Fish and Wildlife Service, 115pp.

Boyles, J.G. 2016. Microclimate of Indiana bat hibernacula in southern Indiana. Unpublished technical report. Prepared by Eko Consulting LLC, Cobden, Illinois for U.S. Fish and Wildlife Service, Indiana Ecological Services Field Office, Bloomington, Indiana. 41 pp.

Boyles, J.G., and A.E. McKechnie. 2010. Energy Conservation in Hibernating Endotherms: Why "Suboptimal" Temperatures Are Optimal. Ecological Modelling, 221:1644–1647.

Brack V. Jr. 2007. Temperatures and Locations Used by Hibernating Bats, Including *Myotis sodalis* (Indiana Bat), in a Limestone Mine: Implications for Conservation and Management. Journal of Environmental Management, 40:739–746.

Brack V. Jr., and J.O. Whitaker. 2001. Foods of the northern myotis, *Myotis septentrionalis*, from Missouri and Indiana, with notes on foraging. Acta Chiropterologica, 3(2):203–210.

Brack, V. Jr., C.W. Stihler, R.J. Reynolds, C.M. Butchkoski, and C.S. Hobson. 2002. Effect of climate and elevation on distribution and abundance in the Midwestern United States. Pp. 21–28 in The Indiana bat: biology and management of an endangered species (A. Kurta and J. Kennedy, eds.). Bat Conservation International, Austin, Texas.

Breed, G., S. Stichter, and E. Crone. 2013. Climate-driven changes in northeastern US butterfly communities. Nature Climate Change, 3:142–145.

Briggler, J., and J. Prather. 2003. Seasonal Use and Selection of Caves by the Eastern Pipistrelle Bat (*Pipistrellus subflavus*). The American Midland Naturalist, 149(2):406–412.

Broders, H.G., G.J. Forbes, S. Woodley, and I.D. Thompson. 2006. Range Extent and Stand Selection for Roosting and Foraging in Forest-Dwelling Northern Long-Eared Bats and Little Brown Bats in the Greater Fundy Ecosystem, New Brunswick. Journal of Wildlife Management 70(5):1174–1184. 2006.

Broders, H.G., L.E. Burns, and S.C. McCarthy. 2013. First Records of the Northern Myotis (*Myotis septentrionalis*) from Labrador and Summer Distribution Records and Biology of Little Brown Bats (*Myotis lucifugus*) in Southern Labrador. The Canadian Field Naturalist, 127:266–269.

Brown, L., and C. Duguay. 2010. The response and role of ice cover in lake-climate interactions. Progress in Physical Geography, 34:671–704.

Burke, H.S. Jr. 1999. Maternity colony formation in *Myotis septentrionalis* using artificial roosts: the rocket box, a habitat enhancement for woodland bats? Bat Research News, 40:77–78.

Burles, D.W., R.M Brigham, R.A., Ring, R.A., and T.E. Reimchen. 2009. Influence of weather on two insectivorous bats in a temperate Pacific Northwest rainforest. Canadian Journal of Zoology 87:132–138.

Caceres, M.C., and M.J. Pybus. 1997. Status of the northern long-eared bat (*Myotis septentrionalis*) in Alberta. Alberta Environmental Protection, Wildlife Management Division, Wildlife Status Report No. 3, Edmonton, AB, 19pp.

Caceres, M.C., and R.M.R. Barclay. 2000. *Myotis septentrionalis*. Mammalian Species, 634:1–4.

Caire, W., R.K. LaVal, M.L. LaVal, and R. Clawson. 1979. Notes on the ecology of *Myotis keenii* (Chiroptera, Vespertilionidae) in Eastern Missouri. American Midland Naturalist, 102(2): 404–407.

Canada Energy Regulator (CER). 2019. Canada's Energy Future 2019: Energy Supply and Demand Projections to 2040. 52 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2019/index.html

Canada Energy Regulator (CER). 2020. Canada's Energy Future 2020: Energy Supply and Demand Projections to 2050. 104 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2020/canada-energy-futures-2020.pdf

Carter, T.C., and G. Feldhamer. 2005. Roost tree use by maternity colonies of Indiana bats and northern long-eared bats in southern Illinois. Forest Ecology and Management, 219:259– 268.

Cheng, T.L., A. Gerson, M.S. Moore, J.D. Reichard., J. DeSimone, C.K.R. Willis, and A.M. Kilpatrick. 2019. Higher fat stores contribute to persistence of little brown bat populations with white-nose syndrome. Journal of Animal Ecology, 88:591–600.

Cheng, T.L., J.D. Reichard, J.T.H. Coleman, T.J. Weller, W.E. Thogmartin, B.E. Reichert, A.B. Bennett, H.G. Broders, J. Campbell, K. Etchison, D.J. Feller, R. Geboy, T. Hemberger, C. Herzog, A.C. Hicks, S. Houghton, J. Humber, J.A. Kath, R.A. King, S.C. Loeb, A. Massé, K.M. Morris, H. Niederriter, G. Nordquist, R.W. Perry, R.J. Reynolds, D.B. Sasse, M.R. Scafini, R.C. Stark, C.W. Stihler, S.C. Thomas, S.C. Thomas, G.G. Turner, S. Webb, B. Westrich, and W.F. Frick. 2021. The scope and severity of white-nose syndrome on hibernating bats in North America. Conservation Biology 2021:1–12.

Cheng, T., B. E. Reichert, W. E. Thogmartin, B. J. Udell, A. M. Wiens, M. Whitby, W. Frick, J.D Reichard, and J. Szymanski. 2022. Winter Colony Count Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter D in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Chevin L.M., R. Lande, and G.M. Mace. 2010. Adaptation, plasticity, and extinction in a changing environment: Towards a predictive theory. PLOS Biology 8(4):e1000357.

Cole, W. National Renewable Energy Laboratory (personal communication, August 3, 2020).

Cole, W.S.C., N. Gates, T. Mai, and P. Das. 2020. 2020 Standard Scenarios Report: A U.S. Electricity Sector Outlook. Golden, CO: National Renewable Energy Laboratory. Available at NREL/TP-6A20-77442.

Coleman, J.T.H., and J.D. Reichard. 2014. Bat White-Nose Syndrome in 2014: A brief assessment seven years after discovery of a virulent fungal pathogen in North America. Outlooks on Pest Management, 25(6):374–377.

Cope, J.B., and S.R. Humphrey. 1972. Reproduction of the bats *Myotis Keenii* and *Pipistrellus subflavus* in Indiana. Bat Research News, 13:9–10.

Crandall K.A., O.R. Bininda-Emonds, G.M. Mace, and R.K. Wayne. 2000. Considering evolutionary processes in conservation biology. Trends in Ecology and Evolution, 15:290–295.

Cryan, P.M., and R.M.R. Barclay. 2009. Causes of bat fatalities at wind turbines: Hypotheses and predictions. Journal of Mammalogy, 90:1330–1340. https://doi.org/10.1644/09-MAMM-S-076R1.1.

Cryan, P.M., C. Meteyer, J.G. Boyles, and D.S. Blehert. 2010. Wing pathology of white-nose syndrome in bats suggests life-threatening disruption of physiology. BioMed Central Biology 8(1):135–143.

Cryan, P.M., P.M. Goressen, C.D. Hein, M.R. Schirmacher, R.H. Diehl, M.M. Huso, D.T.S. Hayman, P.D. Fricker, F.J. Bonaccorso, D.H. Johnston, K. Heist, and D. C. Dalton. 2014. Behavior of bats at wind turbines. Proceedings of the National Academy of Sciences 111(42):15126–15,131. https://doi.org/10.1073/pnas.1406672111.

Cryan, P.M., P. Marcos Gorresen, D. Dalton, S. Wolf , and F. Bonaccorso. 2016. Ultraviolet illumination as a means of reducing bat activity at wind turbines. Presented at the Wind Wildlife Research Meeting XI in Broomfield, Colorado. 2 December 2016. Available at https://www.nationalwind.org/wp-content/uploads/2017/04/83_Cryan.pdf

Davis, W.H., and H.B. Hitchcock. 1965. Biology and Migration of the Bat, *Myotis lucifugus*, in New England. Journal of Mammalogy, 46(2):296–313.

Davy, C.M., G.F. Mastromonaco, J.L. Riley, J.H. Baxter-Gilbert, H. Mayberry, and C.K.R. Willis. 2017. Conservation implications of physiological carry-over effects in bats recovering from white-nose syndrome. Conservation Biology 31(3):615–624. DOI: 10.1111/cobi.12841. Epub 2016 Dec 5. PMID: 27641049.

Deeley, S.M., and W.M. Ford. 2022. Summary Assessment of USFWS Capture Data for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter A in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

De la Cruz, J.L., R.L. Ward, and E.S. Schroder. 2018. Landscape characteristics related to use of artificial roosts by northern long-eared bats in north-central West Virginia. Northeastern Naturalist, 25(3):487–501.

Dobony, C.A., A.C. Hicks, K.E. Langwig, R.I. von Linden, J.C. Okoniewski, and R.E. Rainbolt. 2011. Little brown myotis persist despite exposure to white-nose syndrome. Journal of Fish and Wildlife Management, 2:190–195.

Dodd, L.E., E.G. Chapman, J.D. Harwood, M.J. Lacki, and L.K. Rieske. 2012. Identification of prey of *Myotis septentrionalis* using DNA-based techniques. Journal of Mammalogy, 93(4):1119–1128.

Dowling, Z.R., and D.I. O'Dell. 2018. Bat use of an island off the coast of Massachusetts. Northeastern Naturalist, 25(3):362–382.

Dwyer, P.D. 1971. Temperature regulation and cave-dwelling in bats: An evolutionary perspective. Mammalia, 35(3):424–455.

Easterla, D.A. 1968. Parturition of Keen's Myotis in Southwestern Missouri. Journal of Mammalogy, 49(4):770.

Easterling, D.R., K.E. Kunkel, J.R. Arnold, T. Knutson, A.N. LeGrande, L.R. Leung, R.S. Vose, D.E. Waliser, and M.F. Wehner. 2017. Precipitation change in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W.

Environment and Climate Change Canada. 2018. Recovery Strategy for the Little Brown Myotis (*Myotis lucifugus*), the Northern Myotis (*Myotis septentrionalis*), and the Tri-colored Bat (*Perimyotis subflavus*) in Canada. Species at Risk Act Recovery Strategy Series. Environment and Climate Change Canada, Ottawa. ix + 172 pp.

Environment Yukon. 2011. Yukon Bats. Government of Yukon, Environment Yukon, Whitehorse, Yukon. 22pp.

Erickson, J.L., and S.D. West. 2002. The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. Acta Chiropterologica, 4:17–24

Erickson, W., G. Johnson, D. Young, D. Strickland, R. Good, M. Bourassa, K. Bay, and K. Sernka. 2002. Synthesis and comparison of baseline avian and bat use, raptor nesting and mortality information from proposed and existing wind developments. WEST, Inc. 2003 Central Avenue, Cheyenne, Wyoming 82001.

Erickson, R.A., W.E. Thogmartin, and J.A. Szymanski. 2014. BatTool: an R package with GUI for assessing the effect of white-nose syndrome and other take events on Myotis spp. of bats. Source Code Biol Med 9, 9. https://doi.org/10.1186/1751-0473-9-9.

Erkert, H. G. 1982. Ecological aspects of bat activity rhythms. Pages 201–242 in Ecology of bats. T. H. Kunz (ed). Plenum Press, New York, New York, USA. 425pp.

Farrow L.J., and H.G. Broders. 2011. Loss of forest cover impacts the distribution of the forest-dwelling tri-colored bat (*Perimyotis subflavus*). Mammalian Biology, 72(2):172–179. doi: 10.1016/j.mambio.2010.04.004.

Feldhamer, G.A., T.C. Carter, A.T. Morzillo, and E.H. Nicholson. 2003. Use of bridges as day roosts by bats in southern Illinois. Transactions of the Illinois State Academy of Science, 96(2): 107–112

Feldhamer, G.A., T.C. Carter, and J.O. Whitaker Jr. 2009. Prey Consumed by Eight Species of Insectivorous Bats from Southern Illinois. The American Midland Naturalist, 162(1):43– 51.

Fenton, M.B. 1969. Summer activity of *Myotis lucifugus* (Chiroptera:Vespertilionidae) at hibernacula in Ontario and Quebec. Canadian Journal of Zoology, 47(4)597–602.

Field, K.A., J.S. Johnson, T.M. Lilley, S.M. Reeder, E.J. Rogers, M.J. Behr, and D.M. Reeder. 2015. The White-Nose Syndrome transcriptome: activation of anti-fungal host responses in wing tissue of hibernating little brown myotis. PLoS Pathog 11(10):e1005168.

Field, K.A., B.J. Sewall, J.M. Prokkola, G.G. Turner, M. Gagnon, T.M. Lilley, J P. White, J. S. Johnson, C.L. Hauer, and D.M. Reeder. 2018. Effect of torpor on host transcriptomic responses to a fungal pathogen in hibernating bats. Molecular Ecology, 27: 3727–3743.

Fitch, J.H., and K.A. Shump, Jr. 1979. *Myotis keenii.* Mammalian Species, 121:1–3.

Foden, W.B., B.E. Young, H.R. Akcakaya, R.A. Garcia, A.A. Hoffmann, B.A. Stein, C.D. Thomas, C.J. Wheatley, D. Bickfod, J.A. Carr, D.G. Hole, T. G. Martin, M. Pacifici, J.W. Pearce-Higgins, P.J. Platts, P. Cisconti, J.E.M. Watson, and B. Huntley. 2018. Climate change vulnerability assessment of species. WIREs Climate Change 10(e551):1–36. https://doi.org/10.1002/wcc.551.

Forsman A. 2014. Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion, and infection biology. Proceedings of the National Academy of Sciences of the United States of America 111:302–307.

Foster, R.W., and A. Kurta. 1999. Roosting ecology of the Northern bat (*Myotis septentrionalis*) and comparisons with the endangered Indiana bat (*Myotis sodalis*). Journal of Mammalogy, 80(2):659–672.

Francl, K.E., W.M. Ford, D.W. Sparks, and V. Brack. 2012. Capture and reproductive trends in summer bat communities in West Virginia: assessing the impact of white-nose syndrome. Journal of Fish and Wildlife Management 3(1):33–42. DOI: https://doi.org/10.3996/062011-JFWM-039

Frick, W.F., D.S. Reynolds, and T.H. Kunz. 2010. Influence of climate and reproductive timing on demography of little brown myotis *Myotis lucifugus*. Journal of Animal Ecology, 79:128–136.

Frick, W.F., S.J. Puechmaille, J.R. Hoyt, B.A. Nickel, K.E. Langwig, J.T. Foster, K.E. Barlow, T. Bartonička, D. Feller, A. Haarsma, C. Herzog, I. Horaček, J. Kooij, B. Petrov, R. Reynolds, L. Rodrigues, C.W. Stihler, G.G. Turner, and A.M. Kilpatrick. 2015. Disease alters macroecological patterns of North American bats. Global Ecology and Biogeography, 24:741–749.

Frick, W.F., S.J. Puechmaille, and C.K.R. Willis. 2016. White-nose syndrome in bats. Pages 245–262 *In* C.C.Voigt, and T. Kingston, eds. Bats in the Anthropocene: conservation of bats in a changing world, Springer International Publishing, Cham, Heidelberg, New York, Dordrecht, London.

Frick, W.F., T.L. Cheng, K.E. Langwig, J.R. Hoyt, A.F. Janicki, K.L. Parise, J.T. Foster, and

A.M. Kilpatrick. 2017. Pathogen dynamics during invasion and establishment of white-nose syndrome explain mechanisms of host persistence. Ecology 98: 624–631.

Fuller, N.W., J.D. Reichard, M.L. Nabhan, S.L. Fellows, L.C. Pepin, and T.H. Kunz. 2012. Free-ranging little brown myotis (*Myotis lucifugus*) heal from wing damage associated with white-nose syndrome. EcoHealth, 8:154–162.

Fuller, N.W., L.P. McGuire, E.L. Pannkuk, T. Blute, C.G. Haase, H.W. Mayberry, T. S. Risch, and C K.R. Willis. 2020. Disease recovery in bats affected by white-nose syndrome. Journal of Experimental Biology 223: jeb211912 doi: 10.1242/jeb.211912.

Funk W.C., B.R. Forester, S.J. Converse, C. Darst, and S. Morey. 2019. Improving conservation policy with genomics: A guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics 20:115–134.

Gargas A., M.T. Trest, M. Christensen, T.J. Volk, and D.S. Blehert. 2009. *Geomyces destructans* sp. nov. associated with bat white-nose syndrome. Mycotaxon, 108:147–154.

Garroway, C.J., and H.G. Broders. 2007. Nonrandom association patterns at northern long-eared bat maternity roosts. Canadian Journal of Zoology, 85:956–964.

Geipel, I., M. J. Smeekes, W. Halfwerk, and R.A. Page. 2019. Noise as an informational cue for decision-making: the sound of rain delays bat emergence Journal of Experimental Biology:222, jeb192005. doi:10.1242/jeb.192005

Ghalambor C.K., J.K. McKay, S.P. Carroll, and D.N. Reznick. 2007. Adaptive versus non-adaptive phenotypic plasticity and the potential for contemporary adaptation in environments. Functional Ecology, 21:394–407.

Gilman, P. U.S. Department of Energy (personal communication, July 14, 2020).

Girder, J.F., A.L. Larson, J.A. Homyack, and M.C. Kalcounis-Rueppell. 2016. Winter activity of coastal plain populations of bat species affected by White-Nose Syndrome and wind energy facilities. PLoS ONE 11(11): e0166512. doi:10.1371/journal.pone.0166512.

Glover, A.M., and J.D. Altringham. 2008. Cave selection and use by swarming bat species. Biological Conservation, 141:1493–1504.

Goehring, H.H. 1954. *Pipistrellus subflavus obscurus, Myotis keenii*, and *Eptesicus fuscus fuscus* hibernating in a storm sewer in central Minnesota. Journal of Mammalogy, 35(3):434–436.

Government of Canada. 2020. Canadian Wind Turbine Database. Available at: https://open.canada.ca/data/en/dataset/79fdad93-9025-49ad-ba16-c26d718cc070

Griffin, D.R. 1940a. Reviewed notes on the life histories of New England cave bats. Journal of Mammalogy, 21(2):181–187.

Griffin, D.R. 1940b. Migrations of New England bats. Bulletin of the Museum of Comparative Zoology, 86(6):215–246.

Griffin, D.R. 1945. Travels of banded cave bats. Journal of Mammalogy, 26(1):15–23.

Griffith, L.A., and J.E. Gates. 1985. Food habits of cave-dwelling bats in the central Appalachians. Journal of Mammalogy, 66(3):451–460.

Grindal, S., Collard, T., Brigham, R., & Robert M. R. Barclay. 1992. The Influence of Precipitation on Reproduction by Myotis Bats in British Columbia. The American Midland Naturalist, 128(2):339–344. doi:10.2307/2426468

Grodsky, S.M., M.J. Behr, A. Gendler, D. Drake, B.D. Dieterle, R.J. Rudd, and N.L. Walrath. 2011. Investigating the causes of death for wind turbine-associated bat fatalities. Journal of Mammalogy 92:917–925.

Groves, C.R., D. B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, M.G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. BioScience 526:499–512.

Haavik, L. 2019. Northeastern US Forest Pests. (February, 15, 2019). U.S. Department of Agriculture, Forest Service, Climate Change Resource Center. http://www.fs.fed.us/ccrc/topics/northeastern-us-forest-pests [accessed 3/21/21].

Hall, J.S., and F.J. Brenner. 1968. Summer netting of bats at a cave in Pennsylvania. Journal of Mammalogy, 49(4):779–781.

Hallam, T.G., and P. Federico. 2012. The Panzootic White-nose Syndrome: An Environmentally Constrained Disease? Transboundary & Emerging Diseases 59:269–278.

Hammerson G.A., M. Kling, M. Harkness, M. Ormes, and B.E. Young. 2017. Strong geographic and temporal patterns in conservation status of North American bats. Biological Conservation 212:144–152.

Hanski I., and M. Gilpin, eds. 1997. Metapopulation Biology: Ecology, Genetics, and Evolution. London, UK: Academic Press. 512 pp.

Hayes, M.A., L.A. Hooton, K.L. Gilland, C. Grandgent, R.L. Smith, S.R. Lindsay, J.D. Collins, Susan M. Schumacher, P.A. Rabie, J. C. Gruver, and J. Goodrich-Mahoney. 2019. A smart curtailment approach for reducing bat fatalities and curtailment time at wind energy facilities. Ecological Applications: e01881.

Hayhoe, K., D.J. Wuebbles, D.R. Easterling, D.W. Fahey, S. Doherty, J. Kossin, W. Sweet, R. Vose, and M. Wehner, 2018: Our changing climate. *In* Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 72–144. doi: 10.7930/NCA4.2018.CH2.

Hayman, D. T. S., J. R. C. Pulliam, J. C. Marshall, P. M. Cryan, C. T. Webb. 2016. Environment, host, and fungal traits predict continental-scale white-nose syndrome in bats. Sci. Adv. 2, e1500831.

Hefley, T.J., R.E. Russel, A.E. Ballmann, and H. Zhang. 2020. When and where: estimating the date and location of introduction for exotic pests and pathogens. Available at: https://arxiv.org/abs/2006.16982

Hein, C.D., S.B. Castleberry, and K.V. Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257(4):1200–1207.

Hein, C., and B. Straw. 2021. Proceedings from the State of the Science and Technology for Minimizing Impacts to Bats from Wind Energy. 80 pp. Available at: https://tethys.pnnl.gov/publications/proceedings-state-science-technology-minimizing-impacts-bats-wind-energy

Hendry A.P., Kinnison M.T., Heino M., Day T, Smith T.B., Fitt G., Bergstrom CT, Oakeshott J., Jørgensen P.S., Zalucki M.P., Gilchrist G. 2011. Evolutionary principles and their practical application. Evolutioxnary Applications 4:159–183.

Henderson, L.E., and H.G. Broders. 2008. Movements and resource selection of the northern long-eared myotis (*Myotis septentrionalis*) in a forest-agriculture landscape. Journal of Mammalogy, 89(4):952–963.

Henderson, L.E., L.J. Farrow, and H.G. Broders. 2008. Intra-specific effects of forest loss on the distribution of the forest-dependent northern long-eared bat (*Myotis septentrionalis*). Biological Conservation, 141:1819–1828.

Hoffmann A.A., Sgro C.M. 2011. Climate change and evolutionary adaptation. Nature 470:479–485.

Hoen, B.D., J.E. Diffendorfer, J.T. Rand, L.A. Kramer, C.P. Garrity, and H.E. Hunt. 2018. United States Wind Turbine Database (V3.2, October 14, 2020): U.S. Geological Survey, American Clean Power Association, and Lawrence Berkeley National Laboratory data release, https://doi.org/10.5066/F7TX3DN0. Data downloaded 3 November 2020.

Holderegger R., U. Kamm, and F. Gugerli. 2006. Adaptive vs neutral genetic diversity: Implications for landscape genetics. Landscape Ecology 21:797–807.

Horn, J. W., E. B. Arnett, M. Jensen, and T. H. Kunz. 2008a. Testing the effectiveness of an experimental acoustic bat deterrent at the Maple Ridge wind farm. The Bats and Wind Energy Cooperative and Bat Conservation International, Austin, TX. 30 pp.

Horn, J.W., E.B. Arnett, and T.H. Kunz. 2008b. Behavioral responses of bats to operating wind turbines. Journal of Wildlife Management 72:123–132.

Hoyt, J.R., A.M. Kilpatrick, and K.E. Langwig. 2021. Ecology and impacts of white-nose syndrome on bats. Nature Reviews Microbiology 19:196–210.

Hoyt, J.R., K.E. Langwig, K. Sun, K.L. Parise, A. Li, Y. Wang, X. Huang, L. Worledge, H. Miller, J.P. White, H.M. Kaarakka, J.A. Redell, T. Görföl, S.A. Boldogh, D. Fukui, M. Sakuyama, S. Yachimori, A. Sato, M. Dalannast, A. Jargalsaikhan, N. Batbayar, Y. Yovel, E. Amichai, I. Natradze, W.F. Frick, J.T. Foster, J. Feng, and A.M. Kilpatrick. 2020. Environmental reservoir dynamics predict global infection patterns and population impacts for the fungal disease white-nose syndrome. Proceedings of the National Academy of Sciences 117:7255–7262.

Humphrey, S.R. 1975. Nursery roosts and community diversity of nearctic bats. Journal of Mammalogy 56(2):321–346.

Humphrey, S., A. Richter, and J. Cope. 1977. Summer Habitat and Ecology of the Endangered Indiana Bat, *Myotis sodalis*. Journal of Mammalogy, 58(3):334–346. doi:10.2307/137933.

Humphries, M., Thomas, D. & Speakman, J. 2002. Climate-mediated energetic constraints on the distribution of hibernating mammals. Nature 418, 313–316 (2002). https://doi.org/10.1038/nature00828.

Humphries, M.M., J. Umbanhowar, and K.S. McCann. 2004. Bioenergetic Prediction of Climate Change Impacts on Northern Mammals, Integrative and Comparative Biology, 44(2):152–162, https://doi.org/10.1093/icb/44.2.152

Huso, M., T. Conkling, D. Dalthorp, M. Davis, H. Smith, A. Fesnock, and T. Katzner. Relative energy production determines effect of repowering on wildlife mortality at wind energy facilities. Journal of Applied Ecology DOI: 10.1111/1365-2664.13853

Hyzy, B.A., R.E. Russel, A. Silvis, W.M. Ford, J. Riddle, and K. Russel. 2020. Investigating maternity roost selection by northern long-eared bats at three sites in Wisconsin. Endangered Species Research, 41:55–65, 2020.

Ineson, M.I., T.J. O'Shea, C.W. Kilpatrick, K.L. Parise, and J.T. Foster. 2020. Ambiguities in using telomere length for age determination in two North American bat species. Journal of Mammalogy. 101(4):958–969, 2020 DOI:10.1093/jmammal/gyaa064

Johnson, J.B., W.M. Ford, and J.W. Edwards. 2012. Roost networks of northern myotis (*Myotis septentrionalis*) in a managed landscape. Forest Ecology and Management, 266:223-231.

Johnson, J.B., J.H. Roberts, T.L. King, J.W. Edwards, W.M. Ford, and D.A. Ray. 2014. Genetic structuring of northern myotis (*Myotis septentrionalis*) at multiple spatial scales. Acta Theriologica, 59(2):223–231.

Johnson, J.S., M.R. Scafini, B.J. Sewall, and G.G. Turner. 2016. Hibernating bat species in Pennsylvania use colder winter habitats following the arrival of white-nose syndrome. Pp. 181–199 in (C.M. Butchkoski, D.M. Reeder, G.G. Turner, and H.P. Whidden, eds.) Conservation and Ecology of Pennsylvania's Bats. The Pennsylvania Academy of Sciences.

Johnson, L.N.L., B.A. McLeod, L.E. Burns, K. Arseneault, T.R. Frasier, and H.G Broders. 2015. Population genetic Structure within and among seasonal site types in the little brown bat (*Myotis lucifugus*) and the northern long-eared bat (*M. septentrionalis*). PLoS ONE 10(5): e0126309. doi:10.1371/journal.pone.0126309.

Jones, G., D.S. Jacobs, T.H. Kunz, M.R. Willig, and P.A. Racey. 2009. Carpe noctem: the importance of bats as bioindicators. Endangered Species Research 8:93–115.

Jones, G. and H. Rebelo. 2013. Responses of bats to climate change: learning from the past and predicting the future. In: Adams, R., S. Pedersen. (eds) Bat Evolution, Ecology, and Conservation. Springer, New York, NY. https://doi.org/10.1007/978-1-4614-7397-8_22.

Jordan, G. W. 2020. Status of an Anomalous Population of Northern Long-Eared Bats in Coastal North Carolina. Journal of Fish and Wildlife Management 11(2):665–678.

Keller, S., J.M. Lorch, B. Berlowski-Zier, A. Ballman, and D.S. Blehert. 2021. Analysis of Archival Specimens Confirms White-nose Syndrome in Little Brown Bats (*Myotis lucifugus*) from New York, USA, in Spring 2007. Journal of Wildlife Diseases 57:1–4.

Kerns, J., and P. Kerlinger. 2004. A study of bird and bat collision fatalities at the Mountaineer Wind Energy Center, Tucker County, West Virginia Annual Report for 2003. Unpublished report. 39 pp.

Kilpatrick, A.M., J.R. Hoyt, R.A. King, H.M. Kaarakka, J.A. Redell, J.P. White, K.E. Langwig. 2020. Impact of censusing and research on wildlife populations. Conservation Science and Practice, 2: e264. https://doi.org/10.1111/csp2.264

Kindvall O. 1996. Habitat heterogeneity and survival in a bush cricket metapopulation. Ecology 77(1):207–214.

Krochmal, A.R., and D.W. Sparks. 2007. Timing of Birth and Estimation of Age of Juvenile *Myotis septentrionalis* and *Myotis lucifugus* in West–Central Indiana. Journal of Mammalogy, 88(3):649–656.

Kunz, T.H. 1971. Reproduction of Some Vespertilionid Bats in Central Iowa. American Midland Naturalist, 86(2):477–486.

Kunz, T.H. 1973. Temporal and Spatial Components of Bat Activity in Central Iowa. Journal of Mammalogy, 54(1):14–32.

Kunz, T.H., J.A. Wrazen, and C.D. Burnett. 1998. Changes in body mass and fat reserves in pre-hibernating little brown bats (*Myotis lucifugus*), Écoscience, 5(1):8–17.

Kunz, T.H., E.B. Arnett, W.P. Erickson, A.R. Hoar, G.D. Johnson, R.P. Larkin, M.D. Strickland, R.W. Thresher, and M. D Tuttle. 2007. Ecological impacts of wind energy development on bats: questions, research needs, and hypotheses. Frontiers in Ecology and Environment.5(6):315–324.

Kurta, A., J. Caryl, and T. Tipps. 1997. Bats and Tippy Dam: species composition, seasonal use, and environmental parameters. Michigan Academician, 29(4):473–490.

Kurta, A. 2005. Roosting ecology and behavior of Indiana bats (*Myotis sodalis*) in summer. Pp. 29–42 in K.C. Vories and A. Harrington (eds.), Proceedings of the Indiana bat and coal mining: a technical interactive forum. Office of Surface Mining, U.S. Department of the Interior, Alton, IL.

Kurta, A. 2013. Peer review of the 12-month Find on the Eastern Small-footed Bat and Northern Long-eared Bat. (dated 11/12/2013).

Kurta, A., and S.M. Smith. 2020. Changes in Population Size and Clustering Behavior of Hibernating Bats in the Upper Peninsula of Michigan After Arrival of White-Nose Syndrome. Northeastern Naturalist 27:763–772.

Kurta, A., R.W. Foster, B.A. Daly, A.K. Wilson, R.M. Slider, C.D. Rockey, J.M. Rockey, B.L. Long, G.G. Auteri, J.D. Collins, J.P. White, H.M. Kaarakka, J.A. Redell, and D.M. Reeder. 2020. Exceptional Longevity in Little Brown Bats Still Occurs, despite Presence of White-Nose Syndrome. Journal of Fish and Wildlife Management 11:583–587.

Lacki, M.J. 2013. Peer review of the Proposed Listing of the Northern Long-Eared Bat (*Myotis septentrionalis*) as Endangered. (dated 10/29/2013).

Lacki, M.J. and J.H. Schwierjohann. 2001. Day-roost characteristics of northern bats in mixed mesophytic forest. The Journal of Wildlife Management, 65(3):482–488.

Lacki, M.J., S.K. Amelon, and M.D. Baker. 2007. Foraging ecology of bats in forests. Pp. 83–127 In Lacki, M.J., J.P. Hayes, and A. Kurta eds.. 2007. Bats in forests: conservation and management. John Hopkins University Press, Baltimore, Maryland, USA.

Langwig, K.E., W.F. Frick, J.T. Bried, A.C. Hicks, T.H. Kunz, and A. Marm Kilpatrick. 2012. Sociality, density-dependence and microclimates determine the persistence of populations suffering from a novel fungal disease, white-nose syndrome. Ecology Letters 15(9):1050–1057.

Langwig, K.E., W.E. Frick, R. Reynolds, K.L. Parise, K.P. Drees, J.R. Hoyt, T.L. Cheng, T.H. Kunz, J.T. Foster, and A.M. Kilpatrick. 2015a. Host and pathogen ecology drive the seasonal dynamics of a fungal disease, white-nose syndrome. Proceedings of the Royal Society B 282(1799):1–7.

Langwig, K.E., J. Voyles, M.Q. Wilber, W.F. Frick, K.A. Murray, B.M. Bolker, J.P. Collins, T.L. Cheng, M.C. Fisher, J.R. Hoyt, D.L. Lindner, H.I. McCallum, R. Puschendorf, E.B. Rosenblum, M. Toothman, C.K.R. Willis, C.J. Briggs, and A.M. Kilpatrick. 2015b. Context-dependent conservation responses to emerging wildlife diseases. Frontiers in Ecology and the Environment 13(4):195–202.

Lankau R., P.S. Jørgensen, D.J. Harris, and A. Sih. 2011. Incorporating evolutionary principles into environmental management and policy. Evolutionary Applications 4:315–325.

LaVal, R.K., R.L. Clawson, M.L. LaVal and W. Caire. 1977. Foraging Behavior and Nocturnal Activity Patterns of Missouri Bats, with Emphasis on the endangered species *Myotis grisescens* and *Myotis sodalis.* Journal of Mammalogy, 58(4):592–599.

Lawson M., D. Jenne, R. Thresher, D. Houck, J. Wimsatt, and B. Straw. 2020. An investigation into the potential for wind turbines to cause barotrauma in bats. PLoS ONE 15(12): e0242485. https://doi.org/10.1371/journal.pone.0242485

Lee, Y.F. and G.F. McCraken. 2004. Flight activity and food habits of three species of Myotis bats (Chiroptera: Vespertilionidae) in sympatry. Zool. Studies. 43:589–597.

Lindner, D.L., A. Gargas, J.M. Lorch, M.T. Banik, J. Glaeser, T.H. Kunz, and D.S. Blehert. 2011. DNA-based detection of the fungal pathogen *Geomyces destructans* in soils from bat hibernacula. Mycologia 103(2):241–246.

Linton, D.M., and D.W. MacDonald. 2018. Spring weather conditions influence breeding phenology and reproductive success in sympatric bat populations. Journal of Animal Ecology 87:1080–1090.

Loeb, S. C., and E. A. Winters. 2012. Indiana bat summer maternity distribution: effects of current and future climates. Ecology and Evolution, 3:103–114.

Lorch, J.M., C.U. Meteyer, M.J. Behr, J.G. Boyles, P.M. Cryan, A.C. Hicks, A.E. Ballmann, J.T.H. Coleman, D.N. Redell, D.M. Reeder, and D.S. Blehert. 2011. Experimental infection of bats with *Geomyces destructans* causes white-nose syndrome. Nature 480:376–379.

Lorch, J.M., L.K. Muller, R.E. Russell, M. O'Connor, D.L. Lindner, and D.S. Blehert. 2013. Distribution and Environmental Persistence of the Causative Agent of White-Nose Syndrome, *Geomyces destructans*, in Bat Hibernacula of the Eastern United States. Applied and Environmental Microbiology 79(4): 1293–1301; DOI: 10.1128/AEM.02939-12

Lorch, J.M., J.M. Palmer, D.L. Linder, A.E. Ballman, K.G. George, K. Griffin, S. Knowles, J.R. Huckabee, K.H. Haman, C.D. Anderson, P.A. Becker, J.B. Buchanan, J.T. Foster, and D.S. Blehert. 2016. First detection of white-nose syndrome in western North America. mSphere DOI**:** 10.1128/mSphere.00148-16

Lowe, A.J. 2012. Swarming Behaviour and Fall Roost-Use of Little Brown (*Myotis lucifugus*), and Northern Long-Eared Bats (*Myotis septentrionalis*) in Nova Scotia, Canada. Master's Thesis. St. Mary's University, Halifax, Nova Scotia, Canada. 88pp.

Lucan, R.K., M. Weiser, and V. Hanak. 2013. Contrasting effects of climate change on the timing of reproduction and reproductive success of a temperate insectivorous bat. Journal of Zoology 290:151–159.

Lundy, M., I. Montgomery, and J. Russ. 2010. Climate change-linked range expansion of Nathusius' pipistrelle bat, *Pipistrellus nathusii* (Keyserling & Blasius, 1839). Journal of Biogeography 37:2232–2242.

Maher, S.P., A.M. Kramer, J.T. Pulliam, M.A. Zokan, S.E. Bowden, H.D. Barton, K. Magori, and J.M. Drake. 2012. Spread of white-nose syndrome on a network regulated by geography and climate. Nature Communications 3:1306.

Mangal, M., and Tier C. 1993. A simple direct method for finding persistence times of populations and application to conservation problems. Proceedings of the National Academy of Sciences of the United States of America 90:1083–1086.

Martínková, N., J. Pikula, J. Zukal, V. Kovacova, H. Bandouchova, T. Bartonička, A.D. Botvinkin, J. Brichta, H. Dundarova, T. Kokurewicz, N.R. Irwin, P. Linhart, O.L. Orlov, V. Piacek, P. Škrabánek, M.P. Tiunov and A. Zahradníková Jr. 2018. Hibernation temperature-dependent *Pseudogymnoascus destructans* infection intensity in Palearctic bats. Virulence 9:1734–1750. DOI: 10.1080/21505594.2018.1548685.

Master, L.L., D. Faber-Langendoen, R. Bittman, G.A. Hammerson, B. Heidel, L. Ramsay, K. Snow, A. Teucher, and A. Tomaino. 2012. NatureServe Conservation Status Assessments: Factors for Evaluating Species and Ecosystem Risk. NatureServe, Arlington, VA. 64 pp.

McClure, M. L., D. Crowley, C. G. Haase, L. P. McGuire, N. W. Fuller, D. T. S. Hayman, C. L. Lausen, R. K. Plowright, B. G. Dickson, and S. H. Olson. 2020. Linking surface and subterranean climate: implications for the study of hibernating bats and other cave dwellers. Ecosphere 11:e03274. 10.1002/ecs2.3274.

McGuire, L.P., H.W. Mayberry, and C.K.R. Willis. 2017. White-nose syndromeincreases torpid metabolic rate and evaporative water loss in hibernating bats. American Journal of Physiology; Regulatory, Integrative and Comparative Physiology 313: R680–R686.

McNab, B. 1982. Evolutionary Alternatives in Physiological Ecology of Bats. Pp. 151–200 In (Kunz, T.H., ed.) Ecology of Bats. Plenum Press, New York.

Meierhofer, M. B., J. S. Johnson, K. A. Field, S. S. Lumadue, A. Kurta, J. A. Kath, and D. M. Reeder. 2018. Bats recovering from white-nose syndrome elevate metabolic rate during wing-healing in spring. Journal of Wildlife Diseases 54(3): 480-490.

Menzel, M.A., S.F. Owen, W.M. Ford, J.W. Edwards, P.B. Wood, B.R. Chapman, and K.V.

Miller. 2002. Roost tree selection by northern long-eared bat (*Myotis septentrionalis*) maternity colonies in an industrial forest of the central Appalachian mountains. Forest Ecology and Management, 155:107–114.

Meteyer, C.U., E.L. Buckles, D.S. Blehert, A.C. Hicks, D.E. Green, V. Shearn-Bochsler, N.J. Thomas, A. Gargas, and M.J. Behr. 2009. Histopathologic criteria to confirm white-nose syndrome in bats (cited as: Pathology criteria for confirming white-nose syndrome in bats). Journal of Veterinary Diagnostic Investigation 21:411–414.

Meteyer, C.U., D. Barber, and J.N. Mandl. 2012. Pathology in euthermic bats with white-nose syndrome suggests a natural manifestation of immune reconstitution inflammatory syndrome. Virulence 3:583–588.

Miller-Butterworth, C.M., M.J. Vonhof, J. Rosenstern, G.G. Turner, and A.L. Russell. 2014. Genetic Structure of Little Brown Bats (*Myotis lucifugus*) Corresponds With Spread of White-Nose Syndrome Among Hibernacula. Journal of Heredity 105:354–364.

Moore, M.S., J.D. Reichard, T.D. Murtha, B. Zahedi, R.M. Fallier, and T.H. Kunz. 2011. Specific Alterations in Complement Protein Activity of Little Brown Myotis (*Myotis lucifugus*) Hibernating in White-Nose Syndrome Affected Sites. PLoS ONE (11):e27430.

Muller, L.K., J.M. Lorch, D.L. Lindner, M. O'Connor, A. Gargas, and D.S. Blehert. 2013. Bat white-nose syndrome: a real-time TaqMan polymerase chain reaction test targeting the intergenic spacer region of Geomyces destructans, Mycologia 105:(2):253–259. DOI: 10.3852/12-242

Mumford R.E., and J.B. Cope. 1964. Distribution and status of the chiroptera of Indiana. American Midland Naturalist, 72(2):473–489.

Murray, S. W., and A. Kurta. 2004. Nocturnal activity of the endangered Indiana bat (*Myotis sodalis*). Journal of Zoology 262:197–206.

Muthersbaugh, M.S., W.M. Ford, A.Silvis, and K.E.Powers. 2019. Activity Patterns of Cave-Dwelling Bat Species during Pre-Hibernation Swarming and Post-Hibernation Emergence in the Central Appalachians. Diversity, 11(9):159.

Nagorsen, D.W. and R.M. Brigham. 1993. Bats of British Columbia. Royal British Columbia Museum, Victoria, and the University of British Columbia Press, Vancouver. 164 pp.

National Conference of State Legislatures (NCSL). 2021. State Renewable Portfolio Standards and Goals. <https://www.ncsl.org/research/energy/renewable-portfolio-standards.aspx >. Accessed 12 May 2021.

National Research Council 2007. Environmental Impacts of Wind-Energy Projects. Washington, DC: The National Academies Press. https://doi.org/10.17226/11935.

Neubaum, D. J., and Siemers, J. L. 2021. Bat swarming behavior among sites and its potential for spreading white-nose syndrome. Ecology 00(00):e03325. 10.1002/ecy.3325

Nicholson, C.P., R.D. Tankersley Jr., J.K. Fiedler, N.S. Nicholas. 2005. Assessment and Prediction of bird and bat mortality at wind energy facilities in the southeastern United States. Tennessee Valley Authority, Knoxville, TN. 65 pp.

Nicotra, A.B., E.A. Beever, A.L. Robertson, G.E. Hofmann, J. O'Leary. 2015. Assessing the components of adaptive capacity to improve conservation and management efforts under global change. Conservation Biology 29:1268–1278.

North American Bat Monitoring Program (NABat 2020) Database v5.3.11 (Provisional Release): U.S. Geological Survey. Accessed 2020-11-18. NABat Request Number 10. https://doi.org/10.5066/P9UXA6CF.

North American Bat Monitoring Program (NABat 2021) Database v6.0.12 (Provisional Release): U.S. Geological Survey. Accessed 2021-02-10. NABat Request Number 12. https://doi.org/10.5066/P9UXA6CF.

Ofori, B., Stow, A., Baumgartner, J., & Beaumont, L. 2017. Influence of adaptive capacity on the outcome of climate change vulnerability assessment. Scientific Reports 7:1–12. [12979]. DOI: 10.1038/s41598-017-13245-y.

Oliver T., D.B. Roy, J.K. Hill, T. Brereton, C.D. Thomas. 2010. Heterogeneous landscapes promote population stability. Ecology Letters 13:473–484.

Olivera-Hyde, M., A. Silvis, E.M. Hallerman, W.M. Ford, and E.R. Brittzke. 2019 Relatedness within and among northern long-eared bat (*Myotis septentrionalis*) colonies at a local scale. Canada Journal of Zoology. 97: 724–735.

Osborn, S. California Department of Fish and Game (personal communication April 30, 2021).

O'Shea, T.J., P.M. Cryan, D.T.S. Hayman, R.K. Plowright, and D.G. Streicker. 2016. Mammal Review. 46(3): 175–190.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2019. Forest Resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 223 p. https://doi.org/10.2737/WO-GTR-97

Owen, S.F., M.A. Menzel, W.M. Ford, J.W. Edwards, B.R. Chapman, K.V. Miller, and P.B. Wood. 2002. Roost tree selection by maternal colonies of Northern long-eared Myotis in an intensively managed forest. USDA Forest Service. Newtown Square, Pennsylvania. 10 pp.

Owen, S. F., M.A. Menzel, W. M. Ford, B. R. Chapman, K. V. Miller, J. W. Edwards, P. Bohall Wood. 2003. Home-Range Size and Habitat Used by the Northern Myotis (*Myotis septentrionalis*). American Midland Naturalist 150(2): 352–359.

Parmesan, C., N. Ryrholm, C. Stefanescu, J.K. Hillk, C.D. Thomas, H. Descimon, B. Huntley, L. Kaila, J. Kullberg, T. Tammaru, W.J. Tennent, J.A. Thomas, and M. Warren. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature 399:579–583.

Parsons, S., K.L. Lewis, and J.M. Psyllakis. 2003. Relationships between roosting habitat of bats and decay of aspen in the sub-boreal forests of British Columbia. Forest Ecology and Management 177:559–570.

Patriquin, K.J., M.L. Leonard, H.G. Broders, and C.J. Garroway. 2010. Do social networks of female northern long-eared bats vary with reproductive period and age? Behavioral Ecology and Sociobiology 84:899–913.

Patriquin, K.J., M.L. Leonard, H.G. Broders, W.M. Ford, E.R. Britzke, and A. Silvis. 2016. Weather as a proximate explanation for fission–fusion dynamics in female northern long-eared bats. Animal Behaviour 122:47–57.

Pauli, B. P., P. A. Zollner, G. S. Haulton, G. Shao, and G. Shao. 2015. The simulated effects of timber harvest on suitable habitat for Indiana and northern long-eared bats. Ecosphere 6(4):58.

Pearson, E.W. 1962. Bats hibernating in silica mines in southern Illinois. Journal of Mammalogy, 43(1):27–33.

Perry, R.W., and R.E. Thill. 2007. Roost selection by male and female northern long-eared bats in a pine-dominated landscape. Forest Ecology and Management 247:220–226.

Perry. R.W. 2013. A review of factors affecting cave climates for hibernating bats in temperate North America. Environmental Reviews 21:28–39.

Peste, F., A. Paula, L. P. da Silva, J. Bernardino, P. Pereira, M. Mascarenhas, H. Costa, J. Vieira, C. Bastos, C. Fonseca, and M. J. R. Pereira. 2015. How to mitigate impacts of wind farms on bats? A review of potential conservation measures in the European context. Environmental Impact Assessment Review 51:10–22.

Platt, R.N., B.C. Faircloth, K. A. M. Sullivan, T. J. Kiernan, T. C. Glenn, M. W. Vandewege, T. E. Lee, R. J. Baker, R. D. Stevens, and D. A. Ray. 2018. ConflictingEvolutionary Histories of the Mitochondrial and Nuclear Genomes in New World Myotis Bats. Systematic Biology 67(2):236–249.

Plumb, G. and P. Budde 2011. Unpublished data from U.S. Fish and Wildlife Service Region 3 data request regarding status of 7 cave bats species (sent 03/30/2011).

Puechmaille, S.J., G. Wibbelt, V. Korn, H. Fuller, F. Forget, K. Mühldorfer, A. Kurth, W. Bogdanowicz, C. Borel, T. Bosch, T. Cherezy, M. Drebet, T. Görföl, A. Haarsma, F. Herhaus, G. Hallart, M. Hammer, C. Jungmann, Y. Le Bris, L. Lutsar, M. Masing, B. Mulkens, K. Passior, M. Starrach, A. Wojtaszewski, U. Zöphel, and E.C. Teeling. 2011. Pan-European Distribution of White-Nose Syndrome Fungus (Geomyces destructans) Not Associated with Mass Mortality. PLoS ONE 6(4):e19167.

Rabie, P.A., D. Riser-Espinoza, J. Studyvin, D. Dalthorp, and M. Huso. 2021. Performance of the GenEst mortality estimator compared to the Huso and Shoenfeld Estimators. American Wind Wildlife Institute Technical Report. 10 March 2021. 28 pp. Available at https://awwi.org/wp-content/uploads/2021/04/Performance-of-the-GenEst-Statistical-Mortality-Estimator.pdf

Racey, P. 1982. Ecology of Bat Reproduction. pp 57–104 in Ecology of Bats (ed Kunz, T.). Boston University, Boston, MA.

Racey, P.A., and S.M. Swift. 1981. Variations in gestation length in a colony of pipistrelle bats (Pipistrellus pipstrellus) from year to year. Journal of Reproductive Fertility 61:123–129.

Raesly, R.L., and J.E. Gates. 1987. Winter habitat selection by north temperate cave bats. American Midland Naturalist, 118(1):15–31.

Randall, J. and H.G. Broders. 2014. Identification and characterization of swarming sites used by bats in Nova Scotia, Canada. Acta Chiropterologica, 16:109–116.

Ratcliffe, J.M. and J.W. Dawson. 2003. Behavioral flexibility: the little brown bat, *Myotis lucifugus*, and the northern long-eared bat, *M. septentrionalis*, both glean and hawk prey. Animal Behaviour, 66:847–856.

Rebelo, H., P. Tarroso, and G. Jones. 2010.Predicted impact of climate change on European bats in relation to their biogeographic patterns.Global Change Biology 16:561–576.

Redford, K.H., G. Amato, J. Baillie, P. Beldomenico, E.L. Bennett, N. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G.M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to conserve a (vertebrate) species? BioScience, 61:39–48.

Reeder, D. M., C. L. Frank, G. G. Turner, C. U. Meteyer, A. Kurta, E. R. Britzke, M. E. Vodzak, S. R. Darling, C. W. Stihler, A. C. Hicks, R. Jacob, L. E. Grieneisen, S. A. Brownlee, L. K. Muller, and D. S. Blehert. 2012. Frequent arousal from hibernation linked to severity of infection and mortality in bats with white-nose syndrome. Plos One 7:e38920. <http://dx.doi.org/10.1371%2Fjournal.pone.0038920>.

Reichard, J. D., and T. H. Kunz. 2009. White-nose syndrome inflicts lasting injuries to the wings of little brown myotis (*Myotis lucifugus*). Acta Chiropterologica 11:457–464. <https://doi.org/10.3161/150811009X485684>.

Reynolds, H.T., T. Ingersoll, and H.A. Barton. 2015. Modeling the environmental growth of

Pseudogymnoascus destructans and its impact on the white-nose syndrome epidemic. Journal of Wildlife Diseases 51: 318–331.

Rodenhouse, N.L., L.M. Christenson, D. Parry, and L.E. Green. 2009.Climate change effects on native fauna of Northeastern forests. Canadian Journal of Forest Research 39:249–263.

Rollins, K.E., D.K. Meyerholz, G.D. Johnson, A.P. Capparella and S.S. Loew. 2012. A Forensic Investigation into the Etiology of Bat Mortality at a Wind Farm: Barotrauma or Traumatic Injury? Veterinary Pathology 49: 362–371.

Romano, W.B., J.R. Skalski, R.L. Townsend, K.W. Kinzie, K.D. Coppinger, and M.F. Miller. 2019. Evaluation of an acoustic deterrent to reduce bat mortalities at an Illinois wind farm. Wildlife Society Bulletin 43(4):608–618.

Ruckelhaus M., P. McElhany, and. M.J. Ford. 2002. Recovering species of conservation concern: Are populations expendable? Pp. 305–329 in Kareiva P, Levin S., eds. The Importance of Species: Perspectives on Expendability and Triage. Princeton, NJ: Princeton University Press.

Sasse, D.B., and P.J. Pekins. 1996. Summer roosting ecology of northern long-eared bats *(Myotis septentrionalis)* in the white mountain national forest. Bats and Forests Symposium October 1995, Victoria, British Columbia, Canada, pp. 91–101.

Sasse, D.B., M. L. Caviness, M. J. Harvey, J. L. Jackson, and P. N. Jordan. 2014. Journal of the Arkansas Academy of Science 68(32):170–173.

Scheel, D, T.L.S. Vincent, and G.N. Cameron. 1996. Global Warming and the Species Richness of Bats in Texas. Conservation Biology 10(2)452–464.

Schirmacher, M.S., A. Prichard, T. Mabee, and C.D. Hein. 2018. Evaluating a Novel Approach to Optimize Operational Minimization to Reduce Bat Fatalities at the Pinnacle Wind Farm, Mineral County, West Virginia, 2015. An annual report submitted to NRG Energy and the Bats and Wind Energy Cooperative. Bat Conservation International. Austin, Texas, USA.

Schirmacher, M. R., 2020. Evaluating the Effectiveness of Ultrasonic Acoustic Deterrents in Reducing Bat Fatalities at Wind Energy Facilities." U.S. Department of Energy, DE-EE0007036.

Sgro C.M., Lowe A.J., Hoffmann A.A. 2011. Building evolutionary resilience for conserving biodiversity under climate change. Evolutionary Applications 4:326–337.

Shaffer ML, Stein BA. 2000. Safeguarding our precious heritage. Pp. 301–321 in Stein BA, Kutner LS, Adams JS, eds. Precious Heritage: The Status of Biodiversity in the United States. New York: Oxford University Press.

Sheffield, S.R., J.H. Shaw, G.A. Heidt, and L.R. McClenaghan, 1992. Guidelines for the protection of bat roosts. Journal of Mammalogy 73(3):707–710.

Sherwin, H.A., W.I. Montgomery, and M.G. Lundy. 2013. The impact and implications of climate change for bats. Mammal Review. 18 pp.

Silvis, A., W.M. Ford, E.R. Britzke, N.R. Beane, and J.B. Johnson. 2012. Forest succession and maternity day-roost selection by *Myotis septentrionalis* in a mesophytic hardwood forest. International Journal of Forestry Research 2012:1–8.

Silvis, A., W.M. Ford,,E.R. Britzke, and J.B. Johnson. 2014. Association, roost use and simulated disruption of *Myotis septentrionalis* maternity colonies. Behavioural Processes, 103:283–290.

Silvis, A., W.M. Ford, and E.R. Britzke. 2015. Effects of Hierarchical Roost Removal on Northern Long-Eared Bat (*Myotis septentrionalis*) Maternity Colonies. PLoS ONE, 10(1):1–17.

Silvis, A., R.W. Perry, and W.M. Ford. 2016. Relationships of three species of bats impacted by White-Nose Syndrome to forest condition and management. Gen. Tech. Rep. SRS–214. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 48 p.

Simmons, N.B. 2005. Order Chiroptera: Subfamily Myotinae. p. 516 *in* Mammal species of the world: a taxonomic and geographic reference, D.E. Wilson and D.M. Reeder, editors. The John Hopkins University Press, Baltimore, Maryland. 2000 pp.

Solick, D., D. Pham, K. Nasman, and K. Bay. 2020. Bat activity rates do not predict bat fatality rates at wind energy facilities. Acta Chiroptera 22(1):135–146.

Spanjer, G.R., and M.B. Fenton. 2005. Behavioral responses of bats to gates at caves and mines. Wildlife Society Bulletin, 33:1101–1112.

Sparks, J.K., B.J. Foster, and D.W. Sparks. 2004. Utility pole used as a roost by a northern myotis, *Myotis septentrionalis*. Bat Research News, 45:94.

Spielman, D., B.W. Brook, and R. Frankham. 2004. Most species are not driven to extinction before genetic factors impact them. PNAS 101(42): 15261–15264.

Stratton, C., and K.M. Irvine. 2022. Summertime Analysis Statistical Report for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter B in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Straw, B. U.S. Geological Survey (personal communication, April 6, 2021).

Suriano, Z.J., D.A. Robinson, and D.J. Leathers. 2019. Changing snow depth in the Great Lakes basin (USA): Implications and trends. Anthropocene 26:1–11.

Taucher, J., T. Librandi Mumma, and W. Capouillez. 2012. Pennsylvania Game Commission Wind energy Voluntary Cooperation Agreement Third Summary Report. 79 pp.

Teplitsky C., Robinson M.R., Merilä J. 2014. Evolutionary potential and constraints in wild populations. Pp. 190–208 in Charmantier A, Garant D, and Kruuk LEB, eds. Quantitative Genetics in the Wild. Oxford, UK: Oxford University Press.

Thapa V, G.G. Turner, and M.J. Roossinck. 2021. Phylogeographic analysis of Pseudogymnoascus destructans partitivirus-pa explains the spread dynamics of white-nose syndrome in North America. PLoS Pathog 17(3): e1009236.

Thomas, D.W., and F. Geiser. 1997. Periodic arousals in hibernating mammals: is evaporative water loss involved? Functional Ecology 11:585–591.

Thomas, C., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M. Ferreira de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A. Townsend Peterson13, O.L. Phillips, and S.E. Williams. 2004. Extinction risk from climate change. Nature 427:145–148.

Thurman, L, B. Stein, E. Beever, W. Foden, S. Geange, N. Green, J. Gross, D. Lawrence, O. LeDee, J. O'Leary, J. Olden, L. Thompson, and B. Young. Persist in place or shift in space? Evaluating the adaptive capacity of species to climate change. Frontiers in Ecology and the Environment 18(9):520–528.

Timpone, J.C., J.G. Boyles, K.L. Murray, D.P. Aubrey, and L.W. Robbins. 2010. Overlap in roosting habits of Indiana Bats (*Myotis sodalis*) and northern bats (*Myotis septentrionalis*). American Midland Naturalist 163:115–123.

Turner, G.G., and D.M. Reeder. 2009. Update of white-nose syndrome in bats, September 2009. Bat Research News 50(3):47–53.

Turner, G.G., D.M. Reeder, and J. Coleman T.H. 2011. A Five-year Assessment of Mortality and Geographic Spread of White-nose Syndrome in North American Bats and a Look to the Future. Bat Research News 52(2):13–27.

Tuttle, M.D. and D.E. Stevenson. 1978. Variation in the cave environment and its biological implications. Pp. 108–121, in National Cave Management Symposium Proceedings. Albuquerque: Adobe Press, 1977.

Udell, B.J., B.R. Straw, J. Szymanski, M. Seymour, J. Wong, A.M. Wiens, B.E. Reichert, J.E. Diffendorfer, L. Kramer, and Z. Ancona. 2022. Spatially explicit method of predicting bat wind take at wind facilities and allocating it among winter hibernacula populations. Appendix E-1 in Straw, B.R., J.A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator

Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

U.S. Department of Agriculture, Forest Service (USFS). 2012. Future of America's forest and rangelands: Forest Service 2010 resources planning act assessment. Gen. Tech. Rep. WO-87. Washington, DC. 198 pp.

U.S. Department of Agriculture, Forest Service (USFS). 2013. Adaptive Management Strategy for White-nose Syndrome, Available at: http://www.fs.usda.gov/detail/r2/home/?cid=stelprdb5319926 Accessed 10/21/2020

U.S. Department of Agriculture, Forest Service (USFS). 2014. U.S. Forest Resource Facts and Historical Trends. Forest Service, FS-1035, 64 pp.

U.S. Department of Agriculture, Forest Service (USFS). 2016. Future of America's Forests and Rangelands: Update to the 2010 Resources Planning Act Assessment. Gen. Tech. Report WO-GTR-94. Washington, DC. 250 p.

U.S. Department of Energy (USDOE). 2015. Wind Vision: A New Era for Wind Power in the United States. 289 pp. Available at: https://www.energy.gov/sites/prod/files/WindVision_Report_final.pdf

U.S. Energy Information Administration (USEIA). 2020. Annual Energy Outlook 2020 with projections to 2050. 161 pp. Available at: https://www.eia.gov/outlooks/aeo/pdf/AEO2020%20Full%20Report.pdf

U.S. Fish and Wildlife Service (USFWS). 2007. Indiana Bat (*Myotis sodalis*) Draft Recovery Plan: First Revision. U.S. Fish and Wildlife Service, Fort Snelling, Minnesota, 258 pp.

U.S. Fish and Wildlife Service (USFWS). 2011. A national plan for assisting states, federal agencies, and tribes in managing white-nose syndrome in bats. U.S. Fish and Wildlife Service. Hadley, MA. 21 pp.

U.S. Fish and Wildlife Service (USFWS). 2016. Draft Midwest Wind Energy Multi-species Habitat Conservation Plan in collaboration with the States of Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, and Wisconsin and the American Wind Energy Association. Public Review Draft. April 2016. Available at: https://www.fws.gov/midwest/endangered/permits/hcp/r3wind/DraftHCPandEIS.html

U.S. Fish and Wildlife Service (USFWS). 2015. Endangered and Threatened Wildlife and Plants; Threatened Species Status for the Northern Long-Eared Bat With 4(d) Rule; Final Rule and Interim Rule. Federal Register 80(63): 17974–18033.

van Zyll de Jong, C.G. 1979. Distribution and systematic relationships of long-eared Myotis in western Canada. Canadian Journal of Zoology, 57:987–994.

van Zyll de Jong, C.G. 1985. Handbook of Canadian mammals. National Museums of Canada, Ottawa, Canada. 212pp.

Verant, M.L., J.G. Boyles, W. Waldrep Jr., G. Wibbelt, and D.S. Blehert. 2012. Temperature-Dependent Growth of *Geomyces destructans*, the Fungus That Causes Bat White-Nose Syndrome. PLoS ONE 7(9):e46280.

Verant, M., C. Meteyer, J. Speakman, P. Cryan, J. Lorch, and D.S. Blehert. 2014. White-nose syndrome initiates a cascade of physiologic disturbances in the hibernating bat host. BMC Physiology 14(1):10.

Voigt, C.C., K. Schneeberger, S. L. Voigt-Heucke, and D. Lewanzik. 2011. Rain increases the energy cost of bat flight.Biological Letters 7:793–795

Vose, R.S., D.R. Easterling, K.E. Kunkel, A.N. LeGrande, and M.F. Wehner. 2017. Temperature changes in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 185–206.

Warnecke, L., J.M. Turner, T.K. Bollinger, J.M. Lorch, V. Misra, P.M. Cryan, G. Wibbelt, D.S. Blehert, and C.K.R. Willis. 2012. Inoculation of bats with European *Geomyces destructans* supports the novel pathogen hypothesis for the origin of white-nose syndrome. Proceedings of the National Academy of Sciences 109:6999–7003.

Warnecke, L., J.M. Turner, T.K. Bollinger, V. Misra, P.M. Cryan, D.S. Blehert, G. Wibbelt, and C.K.R. Willis. 2013. Pathophysiology of white-nose syndrome in bats: a mechanistic model linking wing damage to mortality. Biology Letters 9:20130177.

Weaver, S.P., C.D. Hein, T.R. Simpson, J.W. Evans, and I. Castro-Arellano. 2020. Ultrasonic deterrents significantly reduce bat fatality at wind turbines. Global Ecology and Conservation 24:e01099.

Webb, D.R., and J.R. King. 1984. Effects of wetting on insulation of bird and mammal coats. Journal of Thermal Biology 9(3)189–191.

Webb, P. I., J. R. Speakman, P. A. Racey. 1996. A review of the temperatures at which bats hibernate. Candadian Journal of Zoology https://doi.org/10.1139/z96-087

Whitaker, J.O., and L.J. Rissler. 1992. Seasonal activity of bats at copperhead cave. Proceedings of the Indiana Academy of Science, 101:127–134.

Whitaker, J.O., and W.J. Hamilton. 1998. Order Chiroptera: Bats. Chapter 3: pp.89–102 *in* Mammals of the eastern United States, Third Edition, Comstock Publishing Associates, a Division of Cornell University Press, Ithaca, New York, 608pp.

Whitaker, J.O., and R.E. Mumford. 2009. Northern Myotis. pp. 207–214. In Mammals of Indiana. Indiana University Press, Bloomington, Indiana. 688pp.

Whitby, M., B.J. Udell, A.M. Wiens, T. Cheng, W. Frick, B.E. Reichert, and J.D. Reichard. 2022. Summer Mobile Acoustic Transect Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter C in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

White, J.W., P. Freeman, C. A. Lemen. 2017. Habitat selection by the Northern Long-eared Myotis (*Myotis septentrionalis*) in the Midwestern United States: Life in a shredded farmscape. Transactions of the Nebraska Academy of Sciences and Affiliated Societies 37: 1–10.

White, J.A., P.W. Freeman, H.W. Otto, and C.A. Lemen. 2020. Winter use of a rock crevice by northern long-eared myotis (*Myotis septentrionalis*) in Nebraska. Western North American Naturalist: Vol. 80: No. 1, Article 14.

Wiens, A.M., J. Szymanski, B.J. Udell, and W. E. Thogmartin. 2022. Winter Colony Count Data Assessment and Future Scenarios for the Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter E in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Wilcox, A., L. Warnecke, J.M. Turner, L.P. McGuire, J.W. Jameson, V. Misra, T.C. Bollinger, and C.K.R. Willis. 2014. Behaviour of hibernating little brown bats experimentally inoculated with the pathogen that causes white-nose syndrome. Animal Behaviour 88:157–164.

Wilder, A.P., W.F. Frick, K.L. Langwig, and T.H. Kunz. 2011. Risk factors associated with mortality from white-nose syndrome among hibernating bat colonies. Biology Letters 7: 950–953. http://doi.org/10.1098/rsbl.2011.0355

Wilson, R.J., D. Gutierrez, J. Guitierrez, and V.J. Monserrat. 2007. An elevational shift in butterfly species richness and composition accompanying recent climate change. Global Change Biology 13:1873–1887.

Wiser, R., J. Rand, J. Seel, P. Beiter, E. Baker, E. Lantz, and P. Gilman. 2021. Expert elicitation survey predicts 37% to 49% declines in wind energy costs by 2050. Nature Energy. Available at: https://doi.org/10.1038/s41560-021-00810-z

Zackay A. 2007. Random genetic drift and gene fixation. [online] https://www.metabolic-economics.de/pages/seminar_theoretische_biologie_2007/ausarbeitungen/zackay.pdf [Accessed 21 June 2018].

APPENDICES

1. **Key Uncertainties, Wind Energy Mortality Sensitivity Analyses, and State-of-the-Knowledge**
    A. Key Uncertainties
    B. Wind Energy Mortality Sensitivity Analysis
    C. State-of-the-Knowledge
2. **Supplemental Methodology**
    A. Analytical Framework
    B. Adaptive Capacity Analysis
3. **Supplemental Results**
    A. Historical Condition
    B. Current Condition
    C. Future Condition
    D. Qualitative/Comparative Threat Analysis
4. **Supplemental Threat and Future Scenario Information**
    A. WNS
    B. Wind
    C. Climate Change
    D. Habitat Loss
5. **Supplemental Future Scenarios Descriptions**

Appendix 1: Key Uncertainties, Wind Mortality Sensitivity Analyses, and State-of-the-Knowledge

Note: Appendices were created for a three bat (*Myotis lucifugus* [little brown bat, LBB], *Myotis septentrionalis* [northern long-eared bat, NLEB] and *Perimyotis subflavus* [tricolored bat, TCB] SSA). When reference is made to "these bats" or "these species," we are referring to LBB, NLEB or TCB, or all three species.

B. Wind Energy Mortality Sensitivity Analysis

To discern the sensitivity of our results to uncertainty regarding wind energy related mortality, we ran various mortality scenarios. We compared four scenarios: 1) no wind energy related mortality, 2) current predicted mortality, 3) 50% of mortality corresponding to the future high impact scenario, and 4) full projected level of mortality corresponding the high impact scenario. Clearly, WNS is the driving force in the future trajectory of the species (see Figure A-1A1, comparing no WNS impacts to WNS impact scenarios), thus it is not unexpected that the general trend in abundance is unaffected by wind energy mortality (Figure A-1B1). The additive effect of wind energy mortality is, however, discernible as seen when comparing no wind energy related mortality to wind energy mortality scenarios (Figure A-1B2, see bar 1 vs 2 under current conditions and bar 3 vs 4 and 5 under future conditions). The results are markedly sensitive to the range of uncertainty in future mortality levels among scenarios (Figure A-1B2, see bar 4 vs 5).

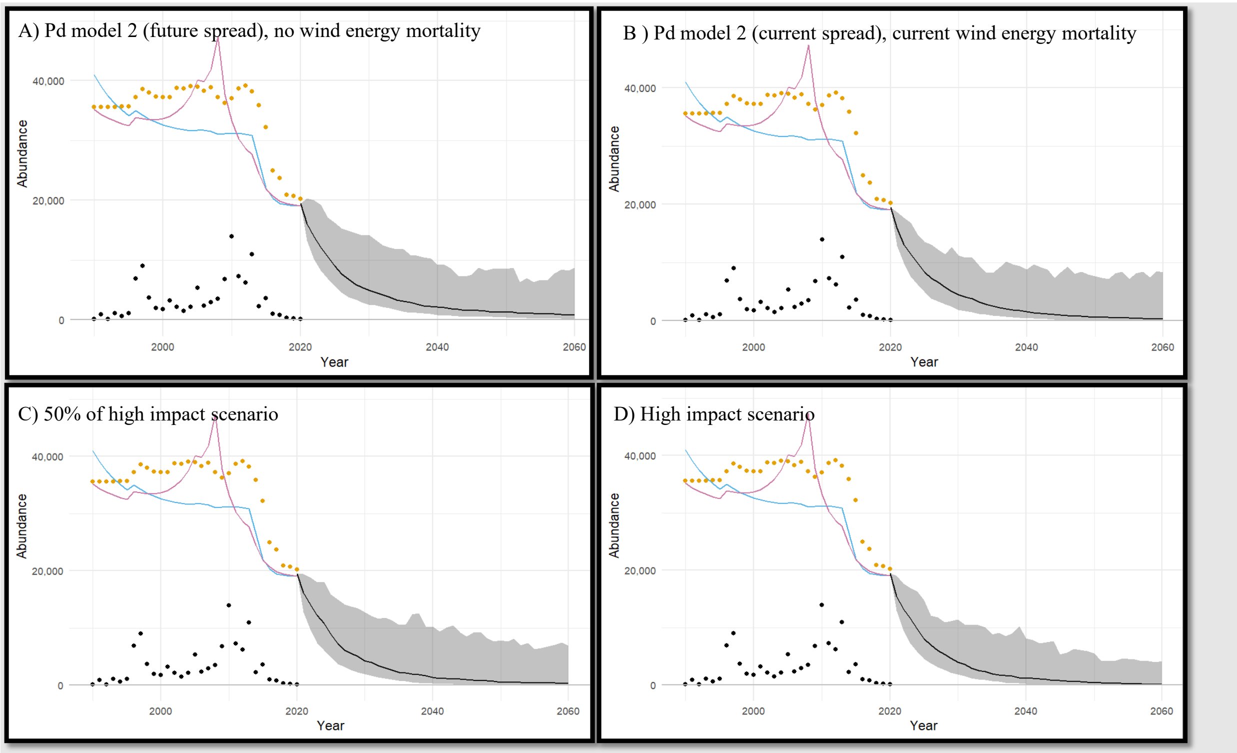

Figure A-1B1. NLEB projected abundance under various wind mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) 50% of the future wind energy mortality under the high impact scenario, and (D) high impact scenario mortality. Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

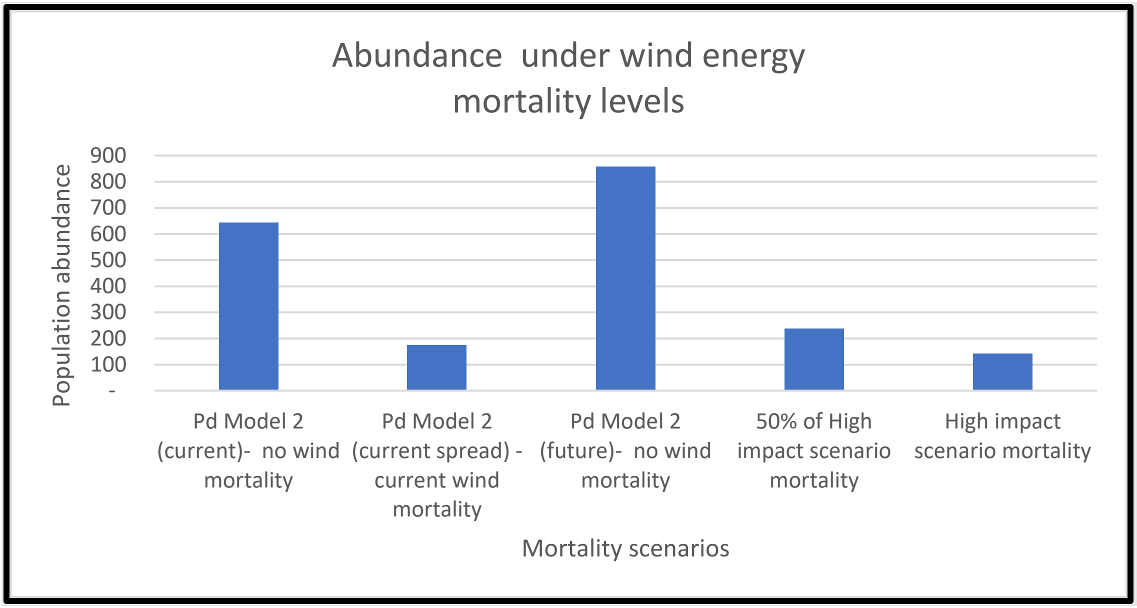

Figure A-1B2. NLEB projected 2060 median abundance under five wind energy related mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) future mortality under low impact scenario, (D) 50% of the future mortality under the high impact scenario, and (E) future mortality under the high impact scenario.

A. Key Uncertainties

Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios, adding environmental stochasticity to our model, and making reasonable assumptions. The key uncertainties are listed in Table A-1.1 and described below.

Table A-1.1. A list of key uncertainties addressed in the analysis.

Current Abundance and Trend	White-nose Syndrome Impacts	Wind Energy Related Mortality	Climate Change and Habitat Loss
Imperfect abundance data over time and space	*Pd* rate of spread*	Future wind energy capacity*	Response to climate change
	WNS impact schedule	Fatality rates	Response to habitat loss
	Duration of WNS impact*	Fatality risk over time and space	
	Bat response where WNS not yet arrived		
	Unknown hibernacula		

*Uncertainties are addressed directly in our high and low impact future scenarios (see Appendix 5).

Abundance and population trend

We do not have **perfect knowledge of current colony abundance and population trend** because hibernacula are not surveyed every year nor concurrently, and there are likely many undocumented hibernacula. Furthermore, bats can be hidden in crevices or inaccessible locations within roosts that are surveyed, and some species are difficult to identify accurately. We address this uncertainty by using predictive models developed by Cheng et al. (2021, entire) and Wiens et al. (2022, pp. 231–247) to predict current abundance and population growth rate (trend) for each known hibernaculum. Cheng et al. (2022, entire) explain that using a statistical model rather than inferring from data summaries is preferred because it can account for site-to-site variation, year-to-year variation, and survey effort, thereby allowing evaluation of the main effects of count over time and the impacts of WNS on counts. Further, statistical methods allow for objectively quantifying the relationships between variables while also quantifying the amount of uncertainty around those results. We summarized the state-of-the-knowledge (raw data summaries) that inform these statistical methods in Appendix 1-C.

The statistical models are constructed from the raw data available (in this case, 3,493 NLEB winter observations). Although these available data are biased towards the eastern portion of the U.S., these data represent the core of the species' known historical and current abundances, and thus are representative of the species' overall condition. Further, while the imminent threats (i.e., WNS, wind, habitat loss, and climate change) may vary temporally, the spatial distribution and overall severity of these threats are not likely to differ markedly (see WNS impacts assumptions below). Coupling this assumption with information concerning the narrow range of optimal conditions for hibernation, we believe these data provide the best available and reliable dataset to assess the current and future viability of the species.

Estimating bat population abundance and trends is challenging due to bats' cryptic nature, wide ranging habits, and variable detectability. A variety of methods have been developed and continue to be improved to fulfil this important information need, including winter and summer colony counts, mist-netting, acoustic monitoring, and mark-recapture studies. However, these efforts are often limited in scope or have been inconsistently applied across species' ranges. For several federally protected hibernating bats (e.g., Indiana bat, Virginia big-eared bat (*Plecotus townsendii virginianus*), and gray bat (*Myotis grisescens*)), successful population monitoring has been achieved through coordinated survey efforts at winter and summer roosts in caves. Fortunately, non-listed species have benefitted from these coordinated survey efforts and monitoring expertise where they overlap with either state or federally listed species. For this reason, estimates of overwintering colony abundance of NLEB are available through a substantial portion of the range over recent decades. Winter survey efforts for these and other hibernating species also increased when concerns about WNS were first raised in North America over 10 years ago. Other sources of data, to date, are more sporadic spatially and temporally but are still useful to inform population status.

We also do not have perfect knowledge of every hibernacula throughout the range of the species (**unknown hibernacula**). NLEBs are commonly found and counted during surveys in cavernicolous (cave-like places) hibernacula in eastern North America. Despite the expectation that many hibernating bats remain unobserved during winter, abundance estimates based on winter counts represent a sound estimate of the site-specific abundances, relative abundances, or at least trends of these species. Importantly, although these surveys do not produce a true census of the populations, they provide an estimate (or index) of abundance during winter when both sexes of these species are roosting together. Summer roost counts are possible, but much less feasible for NLEB due to their roost preferences and frequent roost switching. Mist-netting efforts to estimate capture per unit effort is another method for assessing trends, but these efforts are labor intensive and not commonly available rangewide (as efforts are often concentrated in certain selected areas). Finally, acoustic monitoring can be used to estimate occupancy or indices of abundance that are useful to estimate relative changes in populations but are very difficult to interpret as estimates of abundance. For these reasons, winter colony counts produce the most direct, representative, and feasible method for estimating abundance of NLEB, even if these data only represent minimum estimates of abundance.

Furthermore, WNS is typically detected and causes mortality either during winter or in spring after sick bats emerge from hibernation. Thus, estimating the impacts of this disease is best achieved by evaluating changes in winter colonies, where possible, in response to the arrival of the fungal pathogen. This approach allows for analyses that specify the year of arrival of the fungal pathogen and subsequent changes in population sizes. While winter counts provide the most direct method for estimating the impacts of WNS, additional data streams are used to verify the patterns observed in winter. Analyses of mobile acoustic monitoring and capture efforts provide estimates of changes in relative abundance, while stationary acoustic monitoring produces indices of bat activity. All of these together are also used in occupancy modelling to determine changes in occurrence on the landscape over time. While none of these methods provides a perfect estimate of bat population abundance, together they improve our understanding of the status of the species.

White nose syndrome impacts

To capture the uncertainty in the **rate of spread** of *Pd* by using two different *Pd* occurrence models, a faster spread rate (*Pd* occurrence model 1, Wiens et al. 2022, pp. 226–229) based on spread rates observed and annual changes in the occurrence of *Pd* and a slower spread rate (*Pd* occurrence model 2, Hefley et al. 2020, entire) that incorporates historic occurrence and multiple habitat covariates (Appendices 2A and 5). Both models rely on the same WNS surveillance dataset but each model performs differently in different geographic regions of the country based on the models' parameters. Thus, these two predictions provide a plausible range of the timing of *Pd* spread into the future.

Although we have empirical information on population-level impacts associated with WNS disease progression (on average, 98% decline by the endemic stage, Cheng et al. 2021, entire), there is variability among sites. We identified sites that trended differently (i.e., bats fared better) than most and assumed they do not experience further WNS impacts. For all remaining sites, we assumed they would follow the empirically derived yearly impacts schedule. Wiens et al. (2022, pp. 231–235) used random draws from the impact distribution for each year (Appendix 2-A).

Another source of uncertainty is the **duration of WNS impacts**. We captured the full breadth of uncertainty in our future scenarios. For all scenarios, WNS impacts ameliorate 6 years after the arrival of *Pd*, forming an endemic stage (see Appendix 2-A). Under the low impact scenario, we assumed a 9-year endemic stage and thus yielding a 15-year WNS impacts duration in total. This is the shortest conceivable timeframe based on our analysis of the data available. Under the high impact scenario, we assumed a 34-year endemic stage, thus yielding a 40-year WNS impacts duration in total (Appendix 5). Figure A-1A1 shows results assuming no further WNS impacts beginning in 2020, a 25-year impacts duration, and a 40-year impacts duration.

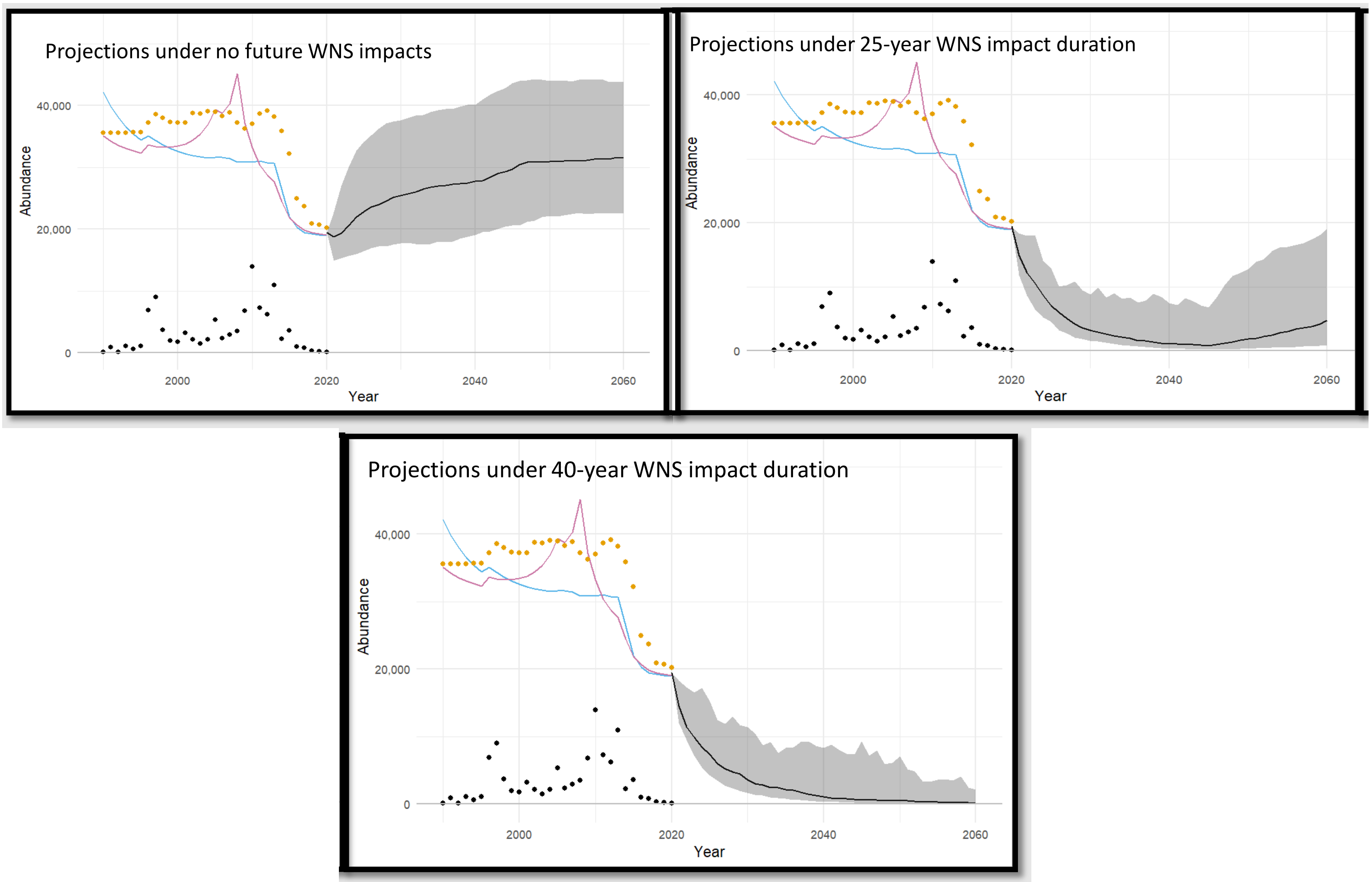

Figure A-1A1. Projected median rangewide abundance (median [black line], 90% CI [gray shading]) over time under no future WNS impacts (top left), a 25-year impacts duration (top right), and a 40-year impacts duration (bottom). Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

Where disease dynamics of WNS have been observed (primarily, but not solely in the eastern half of North America and in cave-like hibernacula), very few colonies of NLEB have avoided severe impacts of the disease. A variety of site characteristics including colony size, temperature, and humidity may explain some of the variability that is observed in the degree of impact caused by WNS. Wilder et al. (2011) predicted that larger colonies will experience impacts of WNS sooner than smaller colonies. Further, Langwig et al. (2012, p. 6) determined that smaller colonies of NLEB may experience less severe impacts than larger colonies during the initial stages of the disease. Frick et al. (2015, p. 6) found that NLEB had a consistently high local extinction risk regardless of pre-WNS colony size. Environmental conditions may also influence impacts of disease. While it has been determined that colder roosts may reduce WNS infections, mortality from WNS has been documented at a wide range of temperatures, including sites with winter temperature approaching 0°C (Langwig et al. 2012, p. 6). Low humidity conditions may also lessen the severity of infection, at least for some species. For example, Indiana bat in drier hibernacula have shown to have less severe impacts from WNS, but this pattern was not observed in NLEB (Langwig et al. 2012, p. 6).

Physiological demands of hibernation limit the ranges of temperature and humidity in which bats can hibernate successfully, although these limits or preferences differ among species. Hibernacula temperatures that are too low present a risk of freezing or raise the energetic cost of torpor. Similarly, hibernacula that are too dry lead to dehydration or frequent arousal from torpor that will consume limited fat reserves. Thus, although these factors may delay or reduce the impacts of WNS, none of them would prevent the arrival of *Pd* or avoid impacts of WNS altogether. Because their winter roosts must be cold and humid to allow for successful hibernation and these conditions are also conducive to growth of *Pd*, it is valid to presume WNS impacts will be similar throughout the portions of the species' ranges where bats hibernate for extended periods, regardless of whether these hibernacula are unknown or human inaccessible.

Wind Mortality

We don't know the **future build-out of wind energy capacity** in the U.S. and Canada. We relied on the National Renewable Energy Laboratory's (NREL) (Cole et al. 2020, entire) and Canadian Energy Regulator's (CER) (CER 2020, entire) projections for the U.S. and Canada, respectively. To capture the uncertainty associated with these projections, we incorporated lower and upper bound capacity projections into our future scenarios. Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario* (Figure 4.10). Our high impact scenario (i.e., higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Chapter 4 and Appendix 5). These build-out scenarios provide reasonable bounds for future expectation of wind capacity in both the U.S. and Canada.

Fatality Rates vary across species, range, and seasons. We used regional specific data garnered from post construction monitoring efforts. We obtained nearly 300 reports spanning 20 states and 4 USFWS Regions. We calculated the mean fatality rate for the species within each USFWS Region using currently accepted methods to account for spatial variability (see Appendix 2).

We also are uncertain about how **fatality risk varies over time and space**. Although it is logical to assume fatality risk declines with decreasing abundance, the functional relationship is unknown. We evaluated fatality rates pre- and post-WNS arrival to discern a relationship between abundance and fatality risk. Where applicable, we applied pre- and post *Pd* fatality rates to account for the uncertainty in fatality risk as abundance changes over time (see Appendix 2). Additionally, we are uncertain of where bats killed at wind facilities originate. To address this uncertainty, we relied on the analysis completed by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) created a distance decay function to allocate total wind mortality per 11x11-km NREL grid cell among hibernacula within the known average maximum migration distance, relative to the size of the hibernating populations as well as the distance from the grid cell centroids (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). However, the analysis did not account for the possibility that some bats may originate from additional unknown hibernacula within the maximum recorded migration distance, or that bats may be migrating farther than previously documented. To look at how this latter uncertainty may affect the results, we ran a scenario in which wind mortality is 50% of what is projected under the high capacity scenario. The additive effect of wind energy mortality is discernible as seen when comparing a no wind to a wind scenario (Figure A-1B2); although from a viability perspective, the results do not appear sensitive to the range of uncertainty in future mortality levels (i.e., no marked changes in the overall trend in abundance).

Climate change

As we detail further in Chapter 4 and Appendix 4, both habitat loss and climate change are pervasive across the species' range and severity of population level declines are currently assumed to be slight (recognizing varying impacts by population). Thus, we believe overall climate change impacts are currently low. While there is uncertainty about the magnitude of future temperature increases and any associated changes in precipitation (e.g., regional changes, rate and intensity of extreme weather events), we have high confidence in the precipitation and temperature changes observed to date and that minimal projected temperature increases (2.2 degrees F (1.2 degrees C), relative to baseline) will occur. Similarly, we have high certainty in observed species responses to changes in temperature and precipitation (which vary geographically). However, we have less certainty about species responses that have not been observed, such as: death of individuals or alteration of hibernacula use due to increased risk of flooding from sea level rise or extreme weather events; reduced reproduction or survival due to increased habitat loss in wildfire prone areas; changes in phenology of bats and their prey; and changes in bat distribution. Lastly, we have uncertainty about possible beneficial impacts from climate change in portions of species' range. While possible, beneficial impacts (e.g., warmer temperatures may lead to shorter hibernation periods, which in turn may decrease the *Pd* exposure duration and thus reduce impacts) are more speculative, at least relative to the observed negative impacts reported in the literature. For this reason, our assessment of effects from climate change likely underestimates risk to these species.

Habitat Loss

We have high confidence that changes in vegetation cover types occur throughout the range of NLEB. We also have high confidence that these changes in landcover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality (during active season tree removal). Despite this knowledge, we have uncertainty about how much forest removal must occur within a home range before impacts associated with winter tree removal are realized. We also have imperfect knowledge of where roosts (summer and winter) for NLEB occur. Therefore, we have uncertainty about which colonies (summer and winter) are at greatest risk of impacts and ultimately the magnitude of risk associated with habitat loss. Also, we have high confidence of prior impacts to winter hibernacula and hibernating bats.

C. State-of-the-Knowledge

For reasons articulated in subsection A above, we relied upon statistical methods rather than raw data alone to assess the species' current status. We summarize the data underlying these methods here.

- We have 3,492 NLEB records from 737 hibernacula (90% of the sites are from the Eastern Hardwoods RPU).
- Based on these raw data:
    - Number of hibernacula with "Last observed = 0": 373 (1990-2020), 5 (2006-2009), 103 (2010–2015), 263 (2016–2020); the ratio (proportion) of extirpated to extant sites increased since WNS discovered in 2006 (Figure A-1C1)
    - Of the 364 potentially extant sites, 84 to 92% have uncertain status (304 and 335 sites do not have ≥ 1 record from 2017–2020 and 2019–2020, respectively)

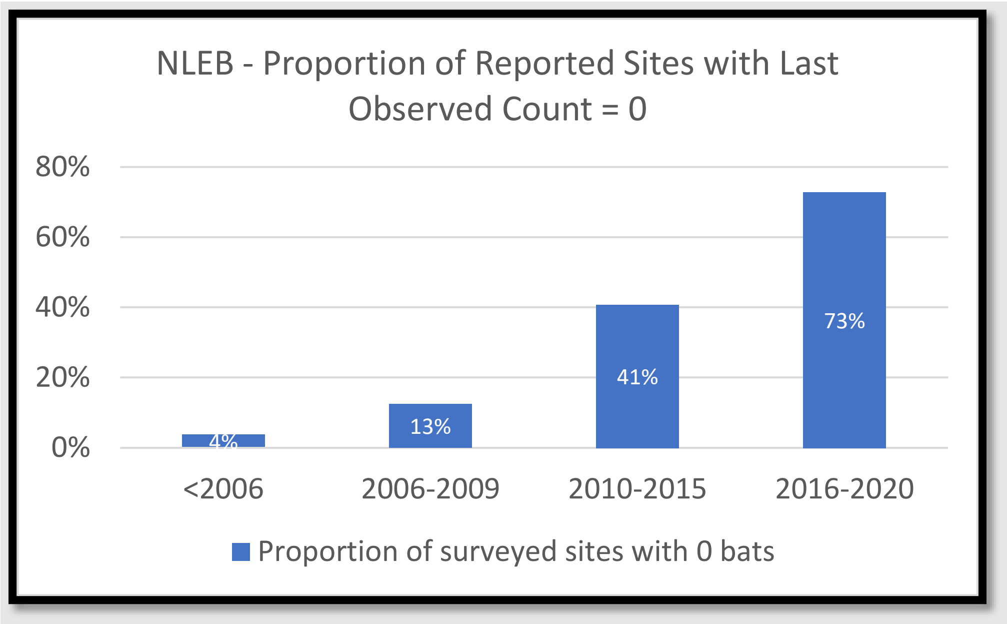

Figure A-1C1. The proportion of sites reported to NABat with 0 as the "last observed count." The proportion is the number of hibernacula with 0 counts divided by the total number of hibernacula surveyed.

- As of 2021, 580 counties across 40 states and 7 provinces have presumed or confirmed *Pd*/WNS (485 are confirmed WNS/*Pd*)(www.whitenosesyndrome.org, accessed May 13, 2021). WNS/*Pd* suspected/confirmed from Nova Scotia southward to South Carolina, westward to Texas, New Mexico, Wyoming, Montana, and Washington (Figure A-1C2).

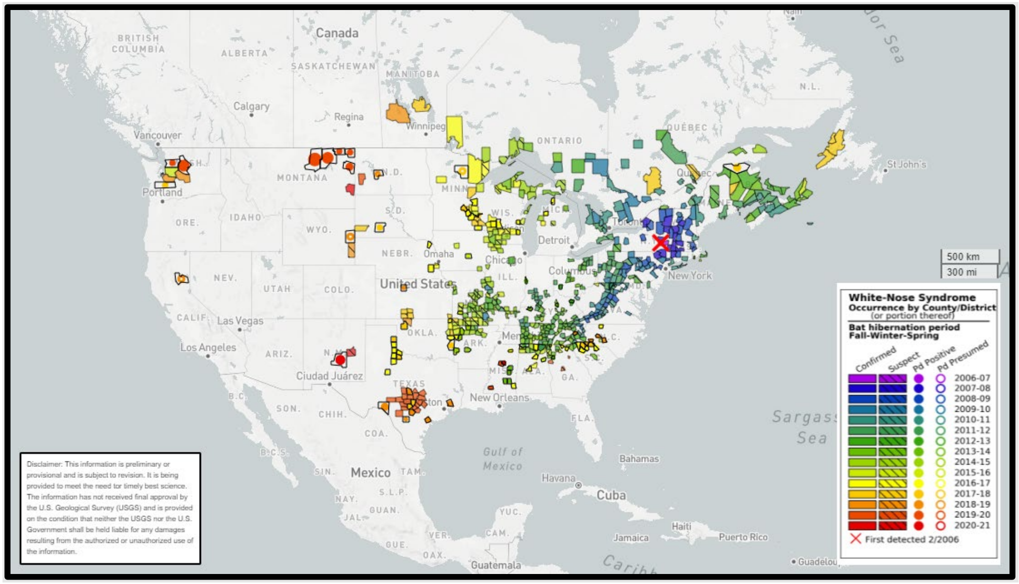

Figure A-1C2. WNS occurrence as of 5/12/2021 (www.whitenosesyndrome.org, accessed May 13, 2021)

- The number of NLEB hibernacula with suspected or confirmed WNS is not available; WNS has been confirmed in every RPU. However, *Pd* has not been detected in the northwestern arm of NLEB's range.
  - As of May 2021, there are 112 NLEB events. Events are winter or summer sites with suspected/confirmed WNS/*Pd* reported on the species of interest (i.e., a species event is recorded only when the species has *Pd*/WNS, even if the WNS/*Pd* confirmed/suspected on other species or the site, *www.whitenosesyndrome.org, accessed May 13, 2021*)
- Where WNS is present, severe declines have occurred, except in a few (3%) hibernacula. On average, NLEB colonies declined by mean 100% (95% CI 97 –100) by the endemic stage of WNS progression (Cheng et al. 2021, p. 7).
- Declines are discernible in summer data as well. Data availability vary among the data type (mobile transect acoustic, stationary acoustic, and mist-net capture data), however we incorporated all available data into the analyses.
  - Using mobile acoustic data from 2009 to 2019, Whitby et al. (2022, entire) found relative abundance declined 50% (Southeast RPU) to 99% (Midwest RPU) from 2009 to 2019. Insufficient data were available for the Subarctic RPU.
  - Using mist-net capture data from 1999 to 2019, Deeley and Ford (2022, entire) found a significant decrease in mean capture rates post-WNS arrival. Estimates derived from their data indicated a 43% (Eastern Hardwoods RPU) to 77%

(Midwest RPU) decline in mean capture rates post-WNS arrival. Insufficient data were available for the Subarctic RPU.

- Using all 3 data types (mobile transect acoustic, stationary acoustic, and mist-net capture data) from 2010 to 2019, Stratton and Irvine (2022, entire) looked at changes in probability of occupancy across the species' range. Although the declines attenuated westward, there was a decline in predicted occupancy across all RPUs (Stratton and Irvine (2022, p. 102). Estimates derived from their results showed declines in the probability of occupancy across all 5 RPUs, ranging from 63% (Subarctic RPU) to 87% (Midwest RPU) from 2010 to 2019.

Appendix 2: Supplementary Methodology

A: Analytical Framework

Below we describe our methods for assessing a species viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and 5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability (Figure A-2A1). Because of the difficulty of delineating individual populations for NLEB, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. The terms populations, winter colonies, and hibernacula are used interchangeably.

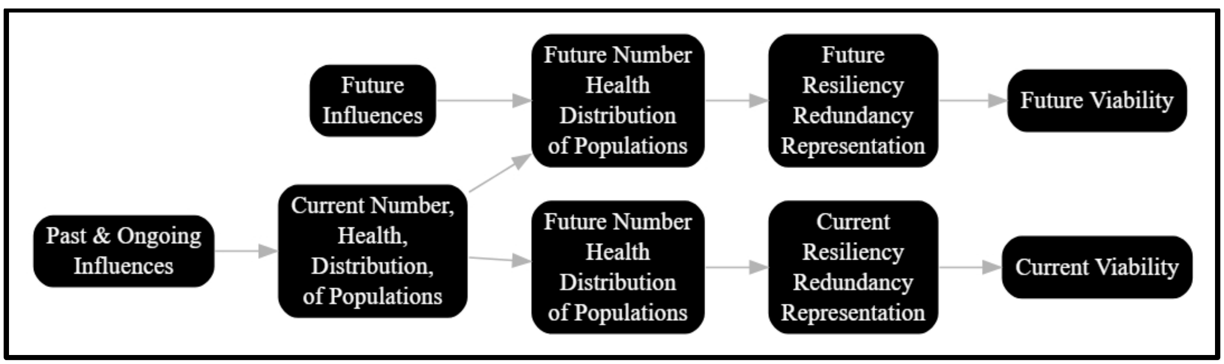

Figure A-2A1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, State and other) across the range to garner summer (capture data and stationary and mobile acoustic) and winter occurrence (hibernacula counts) data. Most of these data are maintained in the North American Bat Monitoring Program (NABat) database, unless otherwise requested by the data contributor or the data was not provided in a format that could be accepted by the database. These efforts yielded thousands of records across the range (Figure A-2A2) and one of the largest bat data repositories we aware of. Hibernacula counts were available for much of the range of NLEB, although occurrence information is limited for the species in parts of the western portion of the U.S. and Canada range. Consistent with the species' biology, we assumed that NLEB employs hibernation in cold, humid roosts even when these roosting locations are not observed by data collectors. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (winter hibernacula surveys; NABat 2021).

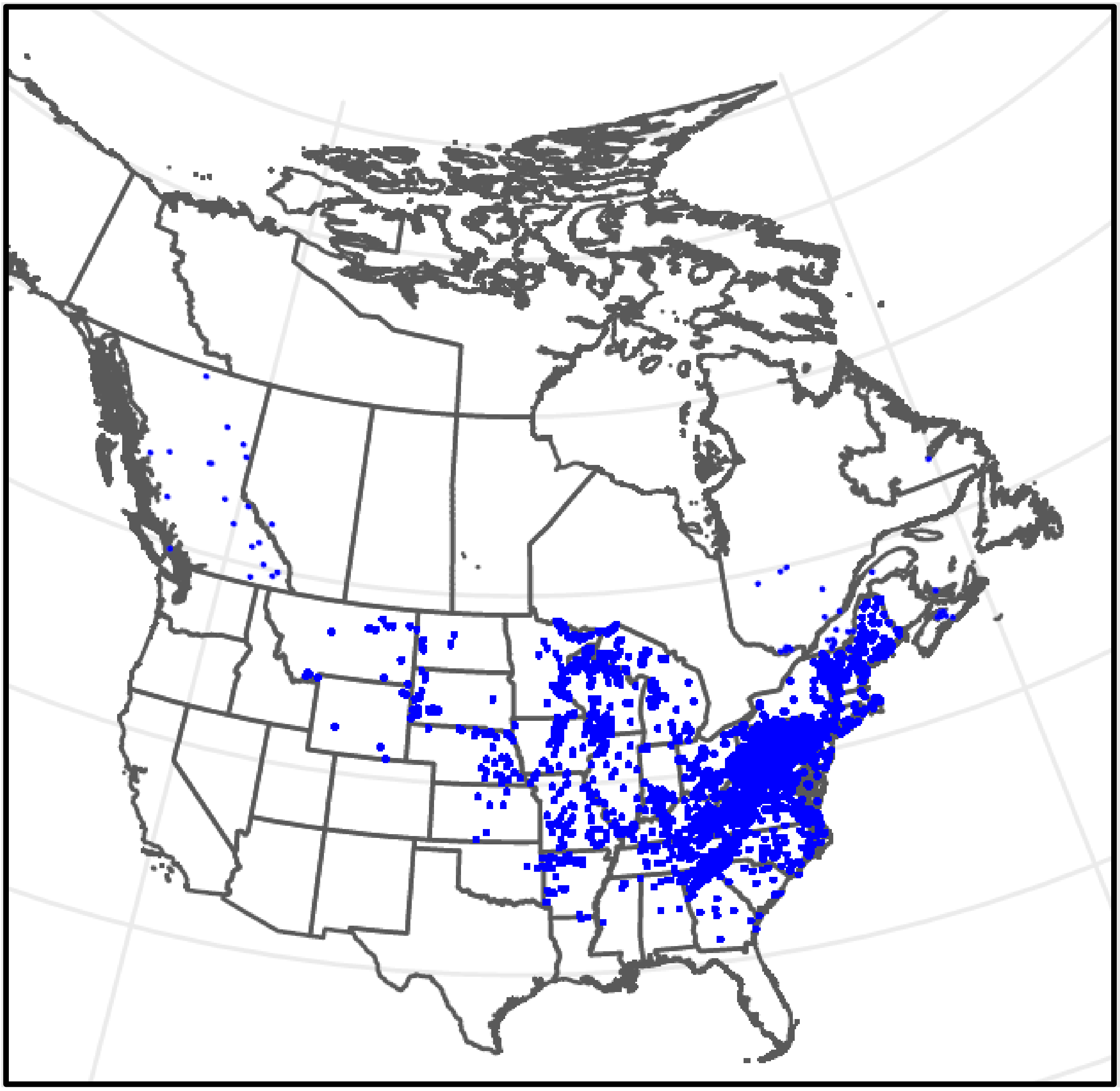

Figure A-2A2. Data available in and contributed to NABat for use in NLEB population analyses. These data show sampling effort and there may be some locations where NLEB was not detected at the survey site. Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center. Disclaimer: Provisional information is subject to revision.

One way we measure population health was hibernacula abundance (N) and population trend (λ). Despite the thousands of winter counts, data are not available for all years and not necessarily both pre and post WNS arrival. Thus, to estimate historical N and λ, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233). For sites with more than 5 data-points (n = 297), they fit the data using a statistical linear mixed effects model (henceforth referred to as Status/Trends model) to estimate the yearly abundance for each hibernaculum from 1990 through 2020. For sites with fewer than 5 colony counts (n = 440, they used last observed count and used the λ from closest hibernaculum or complex of hibernacula. The Status/Trends model relies upon

WNS year of arrival, thus, N and λ estimates vary with the occurrence of *Pd*. Wiens et al. (2022, pp. 231–233) used two projections of *Pd* occurrence (referred to as *Pd* occurrence Model 1 and 2) to identify year of arrival for hibernacula lacking data (see *Current and Future Primary Drivers* subsection below) to capture uncertainty in the presence and spread rate of *Pd* at unknown and uncontaminated sites. Both models use available disease surveillance data documenting past detection of *Pd* but use different parameters to estimate occurrence of *Pd* beyond those detections. Hence, we have two estimates for yearly historical colony N and λ. See Appendix 5 for further details on the Status/Trends model.

Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233) as described above. Additionally, because colony estimates are not available for all hibernacula and because bats occupying a given hibernaculum disperse to many different locations on the summer landscape, we also relied upon the results from USGS-led summer capture records and acoustic records analyses to garner insights on population trends at regional scales (see *Summer Data Analyses* subsection below).

Step 3. Identify the Primary Drivers (Influences)

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) influences of population numbers. We identified WNS, wind related mortality, habitat loss, and climate change as the primary negative influences on the species' abundance. We also identified several other potential influences but based on available information were either too local in scale or lacking data to assess species response.

<u>Qualitative/Comparative Threat Analysis</u> - We assessed the impact of the four influences using an approach adapted from Master et al. (2012, entire) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. Briefly, scope is the proportion of the populations that can be reasonably expected to be affected by the threat within 10 years (current). Severity is the level of damage to the species from the threat. Impact is the degree to which the species is directly or indirectly threatened based on the interaction between the scope and severity values. The criteria used to assign levels are shown in Figure A-2B3.

SCOPE (% of range)	SEVERITY (% of population decline) Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure A-2B3. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012, entire). Impact level (Low to Very High) is based upon the scope and severity assigned.

<u>Quantitative Threat Analysis</u> – We sought to model the impact of the four primary drivers, however, we did not have the time to rigorously determine the species response to changes in climate change and habitat loss. Although we have information on ongoing effects to North American insectivorous bats associated with climate change in specific geographic areas, given the differences in types and magnitude of climate change, the large range of these species, and the fact that we had finite time and resources, we were unable to reliably quantify each species' response in a manner that could be included in the population model (e.g., what specific changes to which specific demographic parameters should we include in response to projected changes in temperature or precipitation). Similarly, habitat loss or alteration can lead to locally consequential effects, especially with the compounding effects of WNS. We considered information on loss or alteration of hibernacula as well as information on changes in landcover types across each species' range; however, given our finite time and resources we were unable to project rangewide future landcover changes or the species associated response in a manner that could be included in the BatTool (e.g., what specific landcover changes would result in what specific changes to which demographic parameters). Instead, we provided a narrative on the spatial extent and magnitude of impact from these two stressors.

To assess the current and plausible future state conditions (magnitude and severity) for WNS and wind related mortality, we used published data, expert knowledge, and professional judgment. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible "low" and "high" impact scenarios. These scenarios were used as inputs to a population-specific demographic model (BatTool, Erickson et al. 2014, entire; explained Step 4 below) to project abundance given specified WNS and wind mortality scenarios.

WNS – To assess the current and future severity of WNS, we calculated disease-induced fatality rates from data gathered from winter colonies following *Pd* arrival (referred to as "WNS impacts schedule", see below). We assumed that the WNS impacts schedule (severity) will not change into the future, and hence, the only difference between the current and future WNS scenarios is the rate of spread (scope) of WNS. To estimate the current and future occurrence of WNS, we relied on two models (several others are available with similar predictions), Wiens et al. (2022, pp. 226–229) and Hefley et al. (2020, entire). We refer to these projections as "*Pd* occurrence model 1 and 2." Both models rely on the same WNS surveillance dataset but allowed us to capture uncertainty in spread rates. Additionally, each model performs differently in different geographic regions of the country, making one model better than the other in a certain area of the country and vice-versa.

Since 2007, collection and management of surveillance data for WNS and *Pd* on bats or in the environment has been coordinated by the National Response to WNS, led by USFWS. State agencies or other appropriate land-management entities conduct most sample collection for disease surveillance and are responsible for reporting county level-determinations of *Pd* status. WNS is confirmed by histopathological observation of lesions characteristic of the disease (Meteyer et al. 2009, entire), molecular detection of the fungus (Muller et al. 2013, entire), or characteristic field signs associated with WNS Case Definitions determined by USGS, National Wildlife Health Center. Year of arrival of WNS or *Pd* at a location is documented at a county-level resolution (available at www.whitenosesyndrome.org).

Wiens et al. (2022, pp. 226–229) used a Gaussian interpolation and projection using linear movement estimates based on observed rates of spread of *Pd* (see Appendix 5 for further information). Hefley et al. (2020, entire) used a diffusion and growth model, which estimates the prevalence (similar to abundance) of *Pd* at a location. In their model, prevalence is influenced by proximity to known occurrences and environmental covariates of percent canopy cover, terrain ruggedness index, waterways, locations of mines, and karst geology. Year of arrival of *Pd* at a location is assigned to the year in which prevalence exceeds 0.25 (this level was chosen by the SSA Core Team based on the prevalence value observed at a subset of sites where *Pd* has already been detected). Separate parameters were calculated to estimate current and future distribution of *Pd* in the Pacific Northwest, where the fungus is expected to have initiated a second epicenter after "jumping" from the nearest known previous occurrence (Lorch et al. 2016, p. 4). Using their estimates of spread rates, future distribution of *Pd* was projected on an annual scale for every 10 km x 10 km grid cell until *Pd* was predicted to be present throughout the entirety of the species' range (Wiens et al. 2022, pp. 226–229) or until statistical confidence interval in the model projection was too great for the value to be reliable (Hefley et al. 2020, entire). The projected *Pd* spread under the two models is shown in Figure A-2B4.

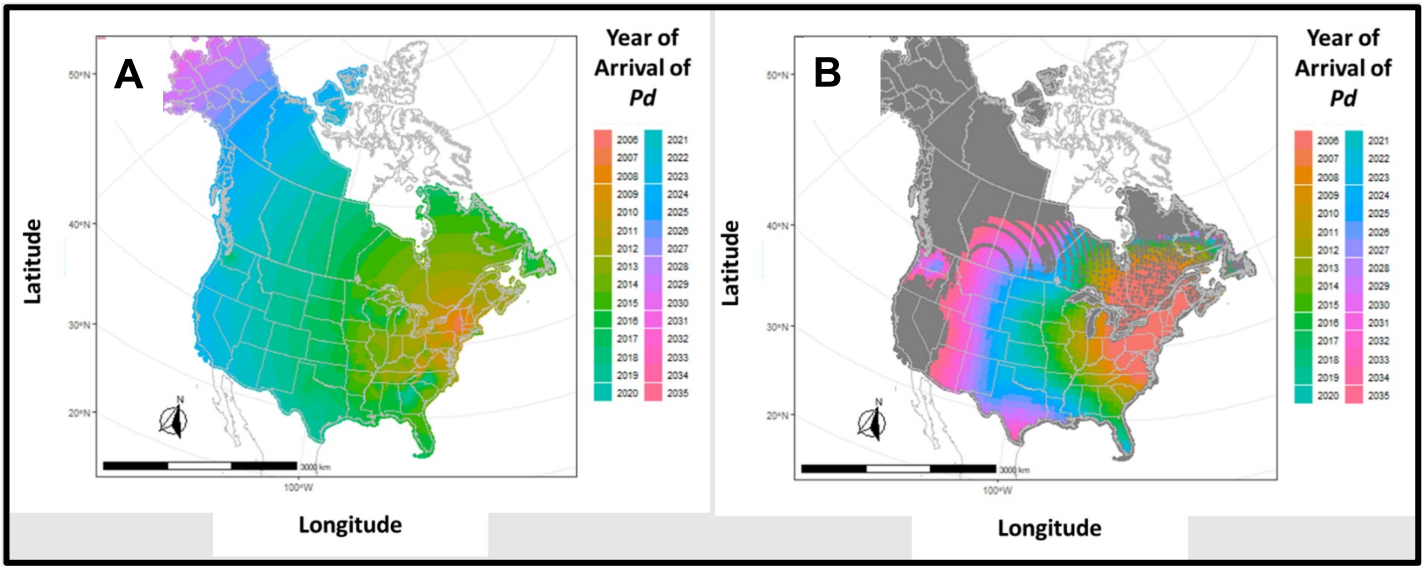

Figure A-2B4. Two models of Pd occurrence in North America since 2007 and into the future. A) A Gaussian interpolation map using spatial relationships and direct observations of Pd occurrence (Wiens et al. 2022, pp. 226–229). B) A diffusion and growth model using observed Pd prevalence in diagnostic samples to predict environmental prevalence of Pd based on spatial and environmental covariates (Hefley et al. 2020, entire).

To estimate current and future WNS impact (fatality rates), we relied on Wiens et al. (2022, pp. 233–235) derived "WNS impacts schedule"; a distribution of annual-specific changes to survival rates. They used data collected during winter hibernacula surveys from 1990 to 2020 and calculated the proportional change in size of the colony between calendar years and between years since arrival of *Pd*. Assuming that change in the estimated colony size was the result of WNS-induced mortality, these estimates of percent change in colony size were translated into changes in adult over-winter survival rate (a parameter in the BatTool). Lastly, they collated these site-specific over-winter survival rates to create annual distributions, i.e., WNS impacts schedule (Figure A-2A5.). This WNS impacts schedule was used in the BatTool to apply WNS impacts to hibernacula over time. For a few sites, the severity of WNS impact has deviated from the norm; for these exceptions, a colony-specific WNS impacts schedule was derived (Wiens et al. 2022, pp. 231–247). See Appendix 5 for additional information and further description of future scenarios.

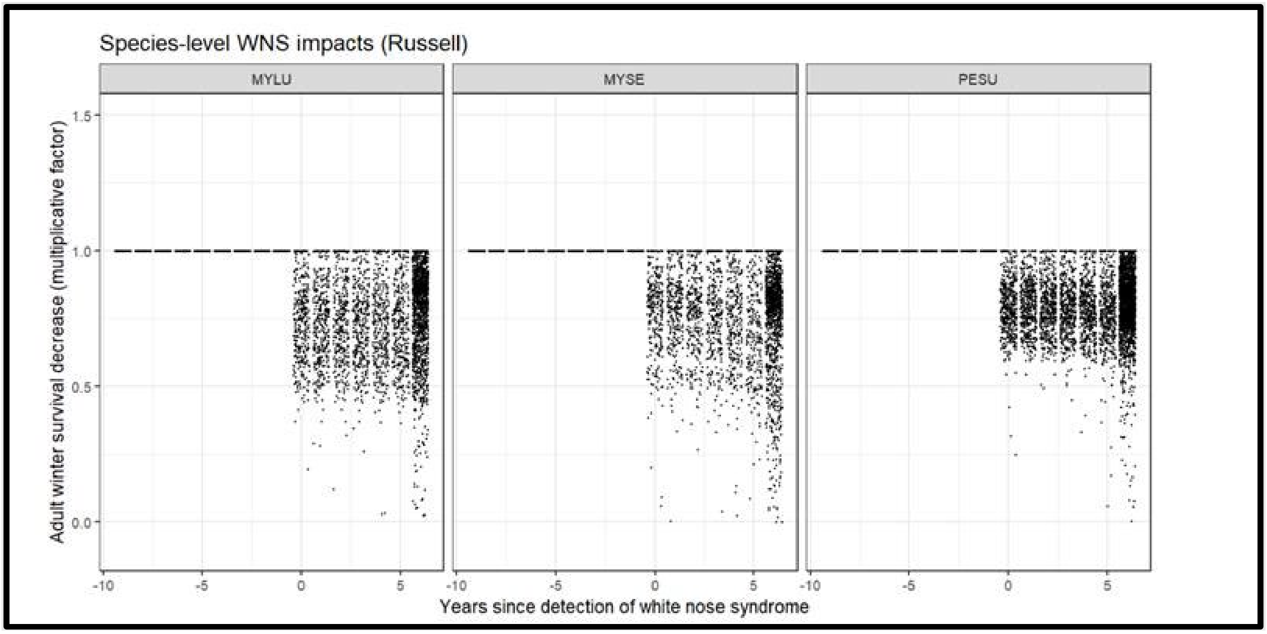

Figure A-2A5. Adult winter survival decreases annually after Pd detection for little brown bat (MYLU), NLEB (MYSE), and tricolored bat (PESU). These data were used to create the WNS impacts schedule. The data depicted for 6 years since detection of Pd include all years since detection ≥ 6.

Wind - To assess the current and future magnitude and severity of current and future wind energy development, we 1) estimated species-specific wind fatality rates (bats per megawatt (MW) per year), 2) applied current and projected future wind capacity within the species' range, and 3) applied species-specific fatality rates to current and future wind capacity to estimate wind related mortality for known hibernating populations. We assumed the only difference between the current and future wind scenarios is the amount of installed wind capacity. NLEB data were too limited to discern differences in percent species composition after WNS arrival, so we assumed no change in fatality rates over time.

To estimate wind fatality rates (severity), we reached out to the public, states, USFWS Ecological Services field offices, and other partners to request data from wind post-construction bat fatality monitoring at wind projects within the ranges of NLEB, little brown bat, and tricolored bat. We obtained 287 reports for wind projects in 20 states within USFWS Legacy Regions (Regions) 3, 4, 5, and 6 (Figure A-2A6).

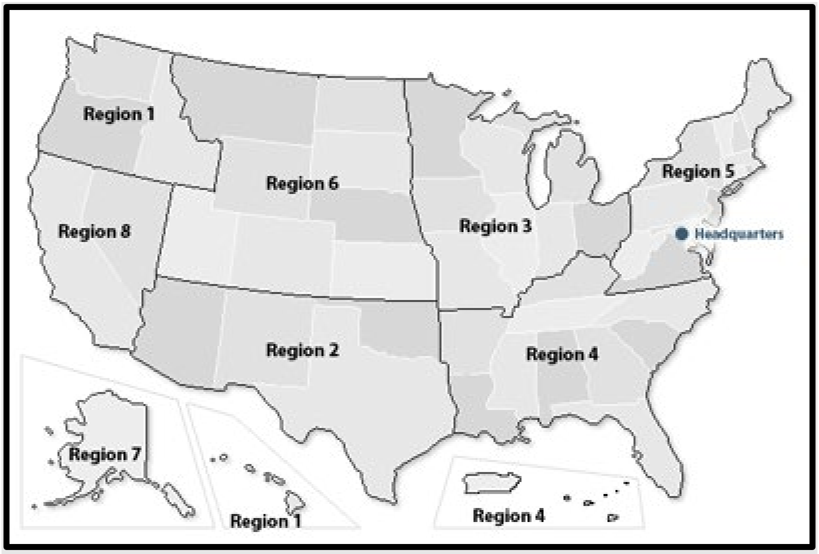

Figure A-2A6. U.S. Fish and Wildlife Service Regions.

For a subset (n = 155) of these reports (those that met our inclusion criteria, described below) we calculated [species]-specific per MW fatality rate using the following equation:

NLEB *per MW fat rate = Bfat * %Sp*

Where *Bfat* is the all-bat fatality rate per MW and *%Sp* is the species-specific percent composition of fatalities reported. *Bfat* was calculated for each Region by deriving annual all-bat per MW fatality rates for each study in our subset, applying corrections for unsearched areas and portions of the year as needed, and then averaging the corrected all-bat fatality rates across the studies in each Region. *%Sp* was calculated by dividing the total number of each species' carcasses reported in our subset of studies by the total number of bat carcasses.

To maximize consistency and comparability across studies in our database, we applied the following inclusion criteria:

1. Study must report a bats/megawatts (MW) or bats/turbine fatality rate, corrected for searcher efficiency (SE) and carcass persistence (CP). If bats/turbines is the only reported fatality rate, the report must also include the number of turbines and MW at the site in order to calculate bats/MW.
2. Turbines were operated without curtailment (i.e., no feathering below manufacturer's or other cut-in speeds) during the study period. In a few instances where studies tested certain cut-in speeds in a subset of turbines and reported separate fatality rates for curtailed versus control (uncurtailed) turbines, the control turbine fatality rate was used.
3. The study search interval was 7 days or less.
4. The study provided the range of dates when carcass searches were performed.
5. The study provided the search area (i.e., plot) dimensions.

Because we only obtained two reports from Region 4, and AWWIC (2020) did not report any NLEB-specific fatality rates, we combined our Region 4 and 5 studies to calculate *Bfat* and *%Sp* in these two Regions. NLEB data from Regions 1, 2, 6, 7, and 8, despite the considerable number of wind projects in these areas, were too limited to generate reliable estimates, and the data for Regions 3, 4 and 5 were too limited to support parsing out by Region. Therefore, we combined all available U.S. studies to derive a single *%Sp* for NLEB. For Canada, we used species composition rates (*%Sp*) reported in Bird Studies Canada et al. (2018). We detected no difference in NLEB *%Sp* by WNS stage; thus, we used rates pre- and post-WNS.

It should be noted that reported fatality rates in our USFWS database were derived using a variety of estimators with differing, imperfect assumptions and biases toward underestimating or overestimating mortality (i.e., see Rabie et al. 2021, entire). Additionally, a recent study by Huso et al. (2021, entire) found that bird and bat fatality rates were relatively constant per unit energy produced by turbines under similar environmental conditions regardless of their size, suggesting that the relative amount of energy produced, rather than simply the size, spacing, or nameplate capacity of turbines, determines the relative all-bat fatality rate. However, bat fatalities per turbine generally increased with turbine size or MW capacity (Huso et al. 2021, p. 4). Lacking information about the capacity factor (total energy produced relative to the theoretical maximum, or nameplate capacity), for all the turbines in our database, we relied on reported bats/MW fatality rates. As such, our averaged fatality rates may overestimate mortality for facilities with high capacity but low energy production (low capacity factor) or vice versa, but are more robust than bats/turbine fatality rates. Moreover, because they are averages across many facilities and states, they should capture the general capacity factor trends across regions, at least for built facilities as of October 2020.

To determine current and future wind capacity (magnitude), we obtained current wind capacity data from the U.S. Wind Turbine Database (USWTDB version 3.2) (Hoen et al. 2018, entire) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we used—at the counsel of experts at USDOE and NREL—the 2020 NREL High and Low Onshore Wind Cost Scenarios data (Cole et al. 2020, p. 26) as reasonable lower and upper bounds of future U.S. wind capacity by state. For Canada, we used Canada Energy Regulator's (CER) Evolving and Reference (baseline) scenarios as our upper and lower bounds, respectively (see Appendix 5 for further description of future scenarios).

Lastly, to calculate hibernacula-specific mortality, we relied upon the analysis by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) summed wind capacity under the lower and upper bound scenarios for each 11x11-km NREL grid cell centroid and calculated a grid cell-specific mortality estimate. They then created a distance decay function to allocate the total mortality per 11x11-km grid cell among hibernacula, relative to the size of the hibernating populations and distance of hibernacula (within the known average maximum migration distance) from the grid cell centroid (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). To account for mortality reductions associated with feathering below the manufacturer's cut-in speed or higher, we applied a 50% mortality reduction to turbines implementing any level of curtailment during the fall or summer and fall seasons, per our 2020 data (USFWS unpublished data). We then multiplied this 50% mortality reduction by t https://www.gray-robinson.com/attorneys-professionals/william-billy-dove he relative proportion of all-bat mortality reported by season in our post-construction mortality database (USFWS unpublished data; Table A–2A1). Based on these proportions, we applied an overall mortality reduction of 50% to turbines curtailing in both summer and fall and a 34% reduction to turbines curtailing in fall only (Table A–2A2).

Table A–2A1. Proportion of all-bat mortality by season (USFWS, unpublished data).

Season	Date Range	Proportion of All-bat Mortality
Spring	March – May 31	0.065
Summer	June 1 – July 30	0.252
Fall	August 1 – November 30	0.68
Summer + Fall	June 1 – November 30	1.0[7]

Table A–2A2. Curtailment categories by season and associated fatality reductions applied to turbine MW.

Category	Curtailment Season	Total Mortality Reduction Applied*
No Curtailment	None	N/A
Fall Only	Fall, Fall + Spring	0.34
Summer + Fall	Summer + Fall, Summer + Fall + Spring	0.50

**Reflects 50% mortality reduction for curtailment multiplied by seasonal proportion of all-bat fatality (Table A-2A1).*

Step 4. Project Future Number, Health, Distribution of Populations Under Current and Future Influences

To project future abundance and trend given current and future state conditions for WNS and wind, we used an existing bat population tool (BatTool, Erickson et al. 2014, entire). The BatTool is a demographic model that projects hibernaculum abundance over time given starting abundance (*N*), trend (λ), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind related mortality as specified by the wind capacity scenarios. Starting abundance (*N*) and trend (λ) were derived from the Status/Trends model described in Step 1 above. For each hibernaculum, the model was run for 100 simulations projecting 40 years into the future.

Using these projected abundance estimates, we calculated various hibernaculum-level and RPU-level (described in Chapter 2) metrics to describe the species' historic, current, and future number, health, and distribution of populations given current and future influences. Figure A-2A7 provides the conceptual framework for the BatTool, which includes the origins of model inputs.

[7]Sum after rounding summer and fall curtailment to nearest tenth.

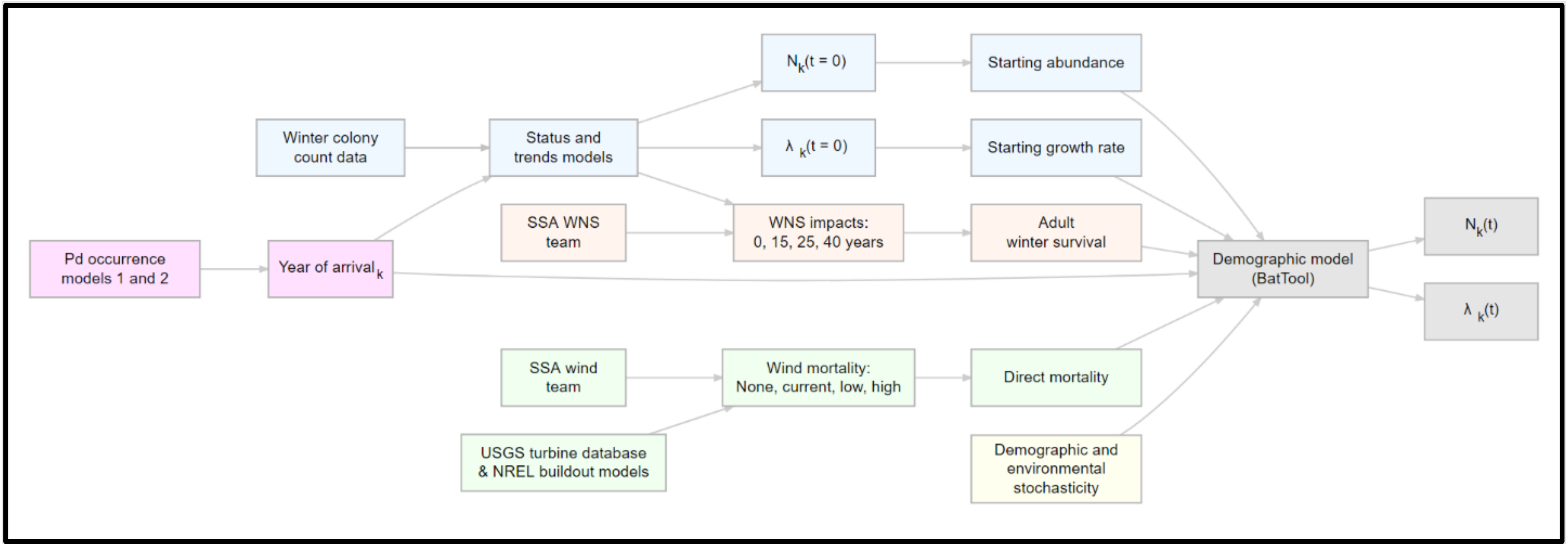

Figure A-2A7. A schematic of the BatTool, including origins of model inputs.

Summer Data Analyses

Because the population of bats monitored at a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led summer capture records and acoustic records analyses. These studies assessed the changes in occupancy (λ) and capture rates over time. We briefly describe their methodologies here; refer to Appendix 5 for further details.

Deeley and Ford (2022, entire) assessed the change in capture rates during summer surveys to garner insights on change in capture rates over time and to assess reproductive conditions of female bats, age structure, and body condition indices of male bats. Between 1999 and 2019, they analyzed NLEB in 9,885 sampling events in which 1,527 (3.6%) records had sufficient information. Rates of capture per unit effort or per sampling event were calculated for each species on an annual timescale by year and by year since arrival of *Pd* based on Wiens et al. (2022, pp. 226–229) *Pd*-occurrence estimates. Stratton and Irvine (2022, entire) assessed recent change in predicted summer occupancy using stationary and mobile acoustic detector records and capture records across NLEB's range. They developed a false-positive occupancy model to estimate probability of occurrence, annual rate of change in summertime occupancy (λ_{avg}), and total change in occupancy (λ_{tot}) from 2010 to 2019. Predicted occupancy was calculated for each 10km by 10km grid cell in NLEB's range and then aggregated to RPU and rangewide scales. The occupancy prediction used covariates of mean elevation, terrain ruggedness index, annual mean precipitation, annual mean temperature, distance to nearest wind farm, percent forest cover, and percent water cover to provide estimates in locations that were not sampled directly. Metrics of change were based on aggregating predicted occupancy between 2010 to 2019 at the RPU and rangewide scale. Whitby et al. (2022, entire) analyzed relative abundance of NLEB annually using acoustical data collected during mobile transect surveys. They analyzed the number of calls detected along driving routes and estimated changes in abundance over the past decade relative to the arrival of WNS and changes in installed wind energy facilities. These analyses were used to estimate rate of change in population at state and RPU scales.

B: Adaptive Capacity Analysis

To garner additional insights into the intrinsic (and historical) ability of NLEB to withstand stressors and adapt to novel changes in the environment, we used the framework put-forth by Thurman et al. (2020, entire). Specifically, Thurman et al. (2020, entire) developed an attribute-based framework for evaluating the adaptive capacity of a given species. Although the basis for the framework is climate change based, the attributes apply to other stressors and changes a species may be exposed to. They identified 12 "core" attributes out of their 36 potential attributes (Figure A-2B1), which collectively provide a comprehensive means of assessing adaptive capacity and are generally available for many species. For each attribute, a species is evaluated on a 5-level "low–moderate–high" scale, with criteria specified for each adaptive capacity level. They do not advise a composite level as many of the attributes interact and some may be "so important that they may overwhelm other considerations (i.e., "deal makers" or "deal breakers"). Using the criteria defined in Thurman et al. (2020, supporting information), we categorized NLEB's level of adaptive capacity for each of the 12 core attributes (Table A-2B1)

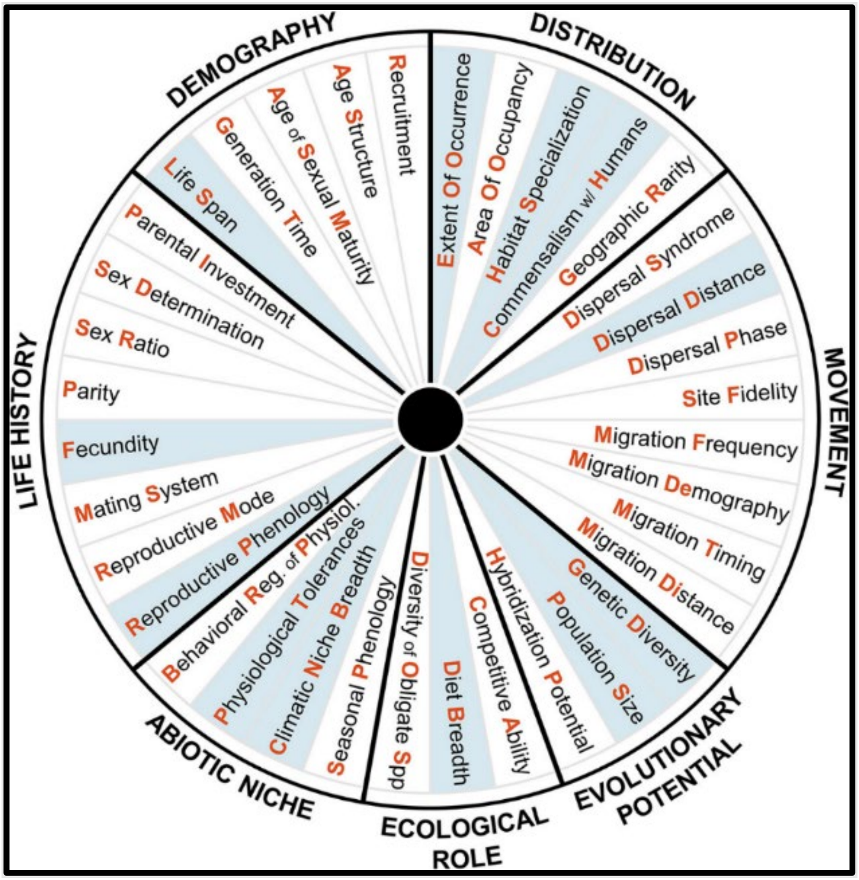

Figure A-2B1. The adaptive capacity "wheel", depicting 36 individual attributes organized by ecological complexes (or themes). Twelve core attributes, representing attributes of particular importance and for which data are widely available, are highlighted in light blue (from Thurman et al. 2020, Figure 1).

Table A-2B1. Assessment of 12 core attributes of NLEB adaptive capacity (from Thurman et al. 2020, Supporting Information).

Core Attribute	Relative Level	Evidence and Relevance
Extent of Occurrence	High	Broadly distributed; typically, a broader distribution is expected to confer higher adaptive capacity.
Habitat Specialization	Low/Moderate	Summer habitat: generalist; suitable roosting habitat includes trees and structures (to lesser degree). However, specific microclimates needed for successful pregnancy and recruitment. Breeding/Winter habitat: specialist; requires suitable hibernacula. High site fidelity in both summer and winter.
Commensalism with Humans	Moderate	Individuals or colonies infrequently utilize man-made structures as summer colony sites (e.g., barns, on utility

Core Attribute	Relative Level	Evidence and Relevance
		poles, behind window shutters, under bridges, and in bat houses) and more frequently use man-made infrastructure as hibernation sites (e.g., mines, tunnels, storm sewer, hydroelectric dam, aqueduct, dry well, crawl space). Use of human-made structures for summer roosts may occur in areas with fewer suitable roost trees (Henderson and Broders 2008, p. 960; Dowling and O'Dell 2018, p. 376).
Genetic Diversity	Low	Although there have been few wide-ranging genetic studies on this species, information collected to date indicates the species to be panmictic (random mating within a population). Johnson et al. (2014, entire) assessed nuclear genetic diversity at one site in New York and several sites in West Virginia, and found little evidence of population structure in NLEB at watershed or regional scales. In addition, studies conducted in Ohio, Nova Scotia and Quebec, Canada, and Kentucky showed variation in NLEB haplotypes at local levels; however, these studies also indicated relatively low levels of overall genetic differentiation between groups and high levels of diversity overall (Arnold 2007, p. 157; Johnson et al. 2015, p. 12; Olivera-Hyde et al. 2020, p.729).
Population Size	Low	Once common, populations have decreased significantly; adaptive capacity may decrease with smaller populations.
Dispersal Distance	Moderate/High	May migrate short distances- up to 89 km (55 mi) between summer and winter habitat (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88)
Climatic Niche Breadth	High	Broad climatic niche breadth across range; may indicate a broader tolerance to climate change because they currently encompass a broader array of climate conditions.
Physiological Tolerances	Moderate	NLEB can employ torpor during food shortages, if conditions allow (even in summer). Clustering and roost selection behavior help to limit the physiological stress from cold or heat waves.
Diet Breadth	High	Use hawking and gleaning foraging behavior. Diverse diet including moths, flies, leafhoppers, caddisflies, and beetles (Griffith and Gates 1985, p. 452; Nagorsen and Brigham

Core Attribute	Relative Level	Evidence and Relevance
		1993, p. 88; Brack and Whitaker 2001, p. 207), with diet composition differing geographically and seasonally (Brack and Whitaker 2001, p. 208). Lepidopterans and coleopterans (beetles) are most commonly found insects in NLEB diet (Brack and Whitaker 2001, p. 207; Lee and McCracken 2004, pp. 595–596; Feldhamer et al. 2009, p. 45; Dodd et al. 2012, p. 1122), with arachnids also being a common prey item (Feldhamer et al. 2009, p. 45)
Reproductive Phenology	Low	Copulation occurs in fall and winter. Females ovulate in the spring upon emergence from hibernacula and fertilization occurs soon after; duration of hibernation and timing of spring emergence is variable across the range. Copulation occasionally occurs again in the spring (Racey 1982, p. 73), and can occur during the winter as well (Kurta 2013, in litt.).
Life Span	Moderate/Low	Maximum NLEB lifespan is estimated to be up to 18.5 years (Hall et al. 1957, p. 407).
Fecundity	Low	A reproductive female can produce up to one offspring annually.

Appendix 3: Supplementary Results

A: Historical Condition

Table A-3A1. The historical number of states/provinces, spatial extent (Extent of Occurrence: EOO), winter abundance, and documented hibernacula rangewide.

# of States/Provinces	EOO (acres)	# of hibernacula	Abundance (max)
29/3	1.2 billion	737	38,131

Table A-3A2. The historical number of hibernacula and winter abundance by RPU.

RPU	# of Hibernacula	Abundance (max)
East Coast	8	1,460
Eastern Hardwoods	665	29,775
Midwest	9	1,218
Southeast	50	393
Subarctic	5	5,628

B: Current Condition

Table A-3B1. Projected yearly rangewide number of states, spatial extent (EOO in acres), number of hibernacula, and median abundance under ***current*** *conditions.*

Year	# of States	EOO (ac)	# of hibernacula	Abundance (median)
2020	18	644 million	139	19,356
2030	7	294 million	11	1,889
2040	2	0	1	540
2050	1	0	0	409
2060	0	0	0	230

Table A-3B2. Projected RPU-level number of hibernaculaand probability of population growth (λ)>1 (pPg) under ***current*** *conditions.*

RPU	Year	# of Hibs	pPg
Southeast	2020	1	0
	2030	1	0.24
	2040	0	0.04
	2050	0	0.03
	2060	0	0.02
Subarctic	2020	5	0.11
	2030	0	0.30

RPU	Year	# of Hibs	pPg
	2040	0	0.30
	2050	0	0.20
	2060	0	0.21
Eastern Hardwoods	2020	115	0
	2030	10	0.19
	2040	1	0.47
	2050	0	0.47
	2060	0	0.64
East Coast	2020	1	0.07
	2030	1	0.36
	2040	0	0.09
	2050	0	0.03
	2060	0	0.01
Midwest	2020	5	0.08
	2030	0	0.24
	2040	0	0.14
	2050	0	0.06
	2060	0	0.07

Table A-3B3. Projected RPU median abundance (90% CI) under current conditions.

RPU	2020	2030	2040	2050	2060
Southeast	298 (CI 298 – 298)	2 (CI 0 – 20)	0 (CI 0 – 6)	0 (CI 0 – 0)	0 (CI 0 – 0)
Subarctic	5,630 (CI 5,630 – 5,630)	16 (CI 0 – 4,118)	0 (CI 0 – 3,328)	0 (CI 0 – 6,459)	0 (CI 0 – 10,601)
Eastern Hardwoods	13,119 (CI 13,076 – 13,162)	1,576 (CI 149 – 6,151)	390 (CI 0 – 3,673)	252 (CI 0 – 2,482)	130 (CI 0 – 1,554)
East Coast	187 (CI 186 – 188)	4 (CI 0 – 50)	0 (CI 0 – 14)	0 (CI 0 – 0)	0 (CI 0 – 0)
Midwest	122 (CI 108 – 136)	4 (CI 0 – 72)	0 (CI 0 – 70)	0 (CI 0 – 44)	0 (CI 0 – 42)

Table A-3B4. Summary of recent NLEB population trends from multiple data types and analyses. Winter Colony analysis –(Chapter 5); Summer Occupancy analysis –Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire). [1] *No data available.*

Representation Unit	Winter colony	Summer occupancy	Summer capture	Summer mobile acoustic
Southeast	-24%	-85%	-47%	-50%
Eastern Hardwoods	-56%	-78	-43%	-87%
Subarctic	-0%	-63%	-[1]	-[1]
Midwest	-90%	-87%	-77%	-99.9%
East Coast	-87%	-79%	-43%	-69%
Rangewide	-49%	-80%	-43% – 77%	-79%

C**: Future Condition**

Table A-3C1. Projected rangewide number of states and known hibernacula with 1 or more bats persisting, spatial extent (EOO), number of hibernacula, and population abundance under ***future*** *scenarios.*

Year	# of States	EOO (ac)	# of hibernacula	Abundance (median)
2030	6	294 million	9	1,801
2040	4	0	1	460
2050	0	0	0	324
2060	0	0	0	201

Table A-3C2. Projected RPU-level number of hibernacula and probability of population growth (λ)>1 (pPg) over time under ***future*** *scenarios.*

RPU	Year	# of Hibs	pPg
Southeast			
	2030	0	0.19
	2040	0	0.06
	2050	0	0.03
	2060	0	0.02
Subarctic			
	2030	0	0.28
	2040	0	0.33
	2050	0	0.23

RPU	Year	# of Hibs	pPg
	2060	0	0.22
Eastern Hardwoods			
	2030	8	0.20
	2040	1	0.50
	2050	0	0.52
	2060	0	0.63
East Coast			
	2030	1	0.28
	2040	0	0.10
	2050	0	0.01
	2060	0	0.00
Midwest			
	2030	0	0.27
	2040	0	0.19
	2050	0	0.10
	2060	0	0.12

Table A-3C3. Projected RPU median abundance (90% CI) under future scenarios.

RPU	2020	2030	2040	2050	2060
Southeast	298 (CI 298 – 298)	2 (CI 0 – 18)	0 (CI 0 – 2)	0 (CI 0 – 0)	0 (CI 0 – 0)
Subarctic	5,630 (CI 5,630 – 56,30)	22 (CI 0 – 6,199)	0 (CI 0 – 5,199)	0 (CI 0 – 9,974)	0 (CI 0 – 11,830)
Eastern Hardwoods	13,119 (CI 13,076 – 13,162)	1,358 (CI 107 – 5,040)	294 (CI 0 – 3,385)	174 (CI 0 – 2,486)	66 (CI 0 – 2,297)
East Coast	187 (CI 186 – 187)	6 (CI 0 – 46)	0 (CI 0 – 12)	0 (CI 0 – 0)	0 (CI 0 – 0)
Midwest	122 (CI 108 – 136)	2 (CI 0 – 74)	0 (CI 0 – 73)	0 (CI 0 – 83)	0 (CI 0 – 65)

D: Qualitative/Comparative Threat Analysis

To estimate the proportion of NLEB's range with wind mortality risk in 2020, we took the following approach:

1. Buffer extant (known) hibernacula by avg. migration distance (89 km)
2. Buffer summer points by avg. migration distance (89 km)

3. Merge & dissolve buffered hibernacula and summer shapefiles into a “NLEB occupied” area, clip NLEB range by contiguous U.S. border for “NLEB U.S. range”, and clip NLEB occupied area by NLEB U.S. range.
4. Buffer & dissolve current turbines (Hoen et al. 2018) by avg. migration distance for “wind threat” area (89 km)
5. Clip wind threat area by NLEBs occupied area for “NLEB wind risk” area
6. Compare NLEB wind risk area with range area in U.S.: NLEB = 3,378,317 km^2 and 2020 wind risk area (U.S.): NLEB = 1,650,889 km^2 (49% of U.S. range) (Figure A-3D1)

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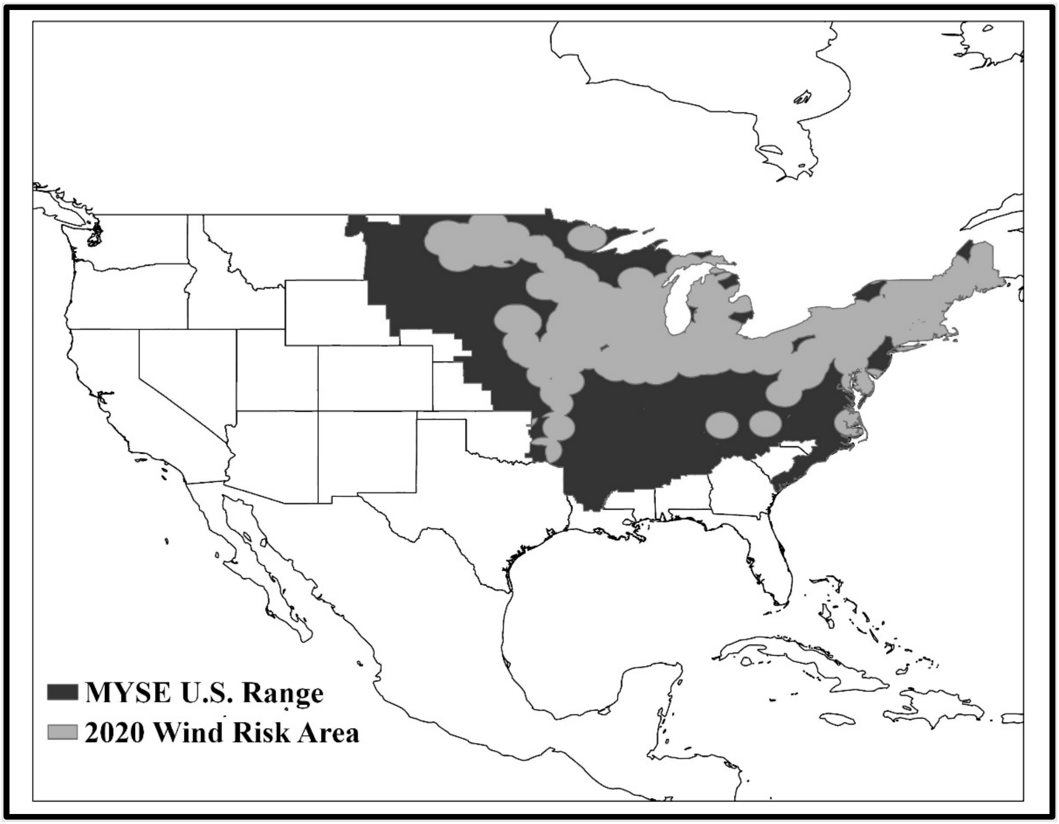

Figure A-3D1. Estimated extent of NLEB's U.S. range with wind mortality risk.

To estimate the proportion of NLEB's range with wind mortality risk in 2050 (per low and high build-out scenarios), we took the following approach:

1. 2050 Low Build-out Scenario:
    a. Buffer & dissolve 2050 High Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance for "future wind threat: area. *Note: Future MW summed by 11x11-km NREL grid cell so does not capture actual distribution of turbines on landscape* (89 km)
    b. Clip wind threat area by NLEB occupied areas for "NLEB 2050 low wind risk" area (U.S.)
    c. Compare NLEB 2050 low wind risk areas with range area in U.S.
        i. Range area (U.S.): 3,378,317 km^2
        ii. 2050 low wind risk areas = 937,019 km^2 (28% of U.S. range) (Figure A-3D2*)*
2. 2050 High Build-out Scenario:
    a. Buffer & dissolve 2050 Low Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance for "future wind threat" area. *Note: Future MW summed by 11x11-km NREL grid cell so does not capture actual distribution of turbines on landscape*: 89 km

b. Clip wind threat area by NLEB occupied areas for "NLEB 2050 high wind risk" area (U.S.)
c. Compare NLEB 2050 high wind risk areas with range area in U.S.
    i. Range area (U.S.): 3,378,317 km^2
    ii. 2050 high wind risk areas (U.S.): 2,374,707 km^2 (70% of U.S. range) (Figure A-3D3*)*

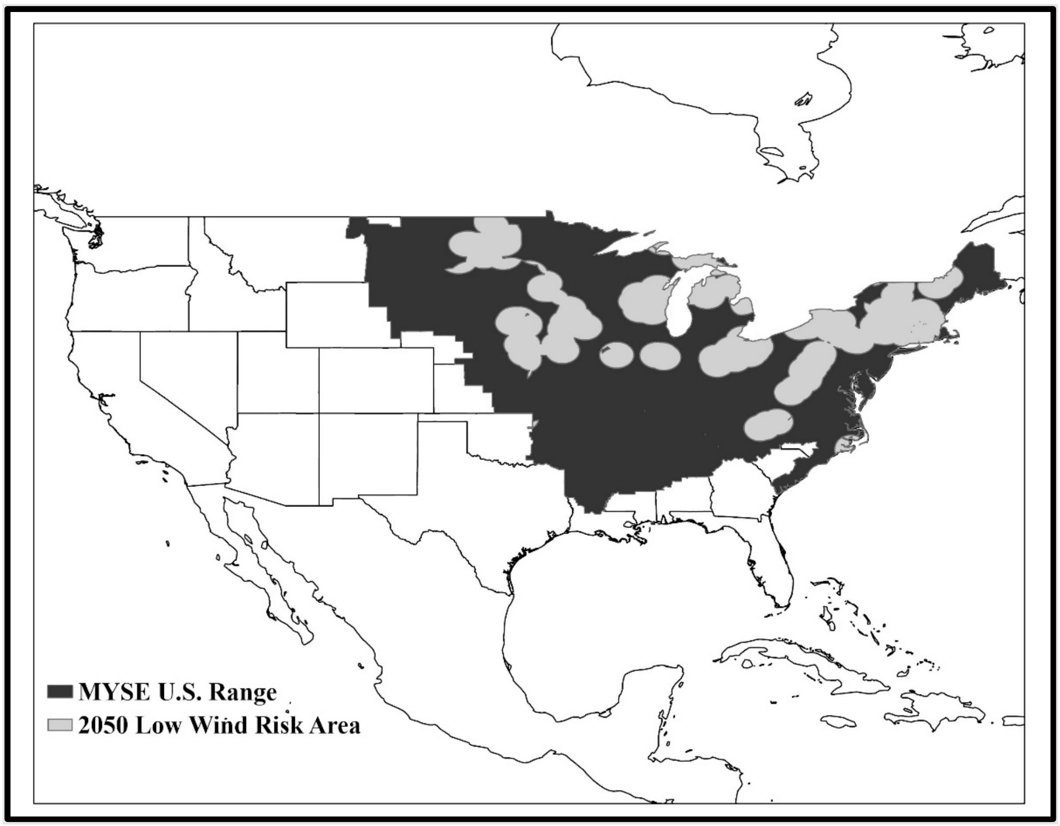

Figure A-3D2. Estimated extent of NLEB's U.S. range with wind mortality risk in 2050 low build-out scenario.

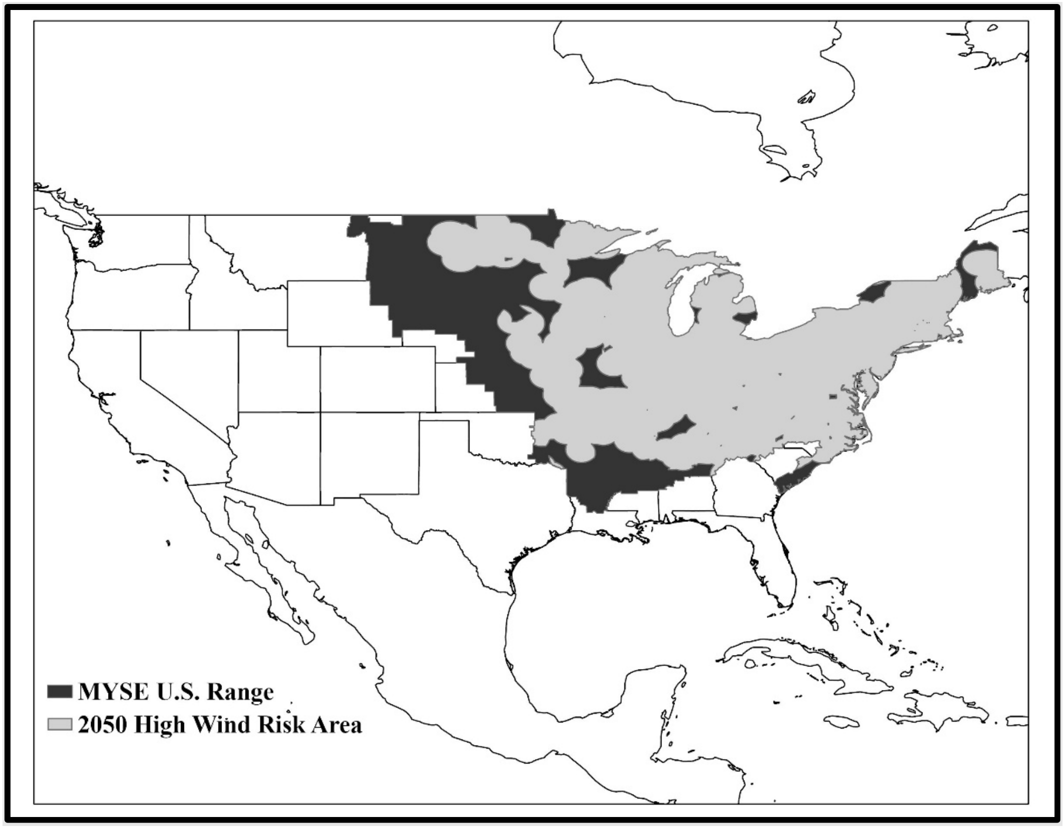

Figure A-3D3. Estimated extent of NLEB's U.S. range with wind mortality risk in 2050 high build-out scenario.

To estimate the severity of impact from wind energy related mortality, we compared scenarios to baseline scenarios without wind energy mortality. The results are presented in Tables A-3D1 and A-3D2.

Table A-3D1. Projected median rangewide abundance given wind energy mortality under 4 current conditions scenarios: 1) Pd model 1 and current wind energy related mortality, 2) Pd model 1 and no wind energy related mortality, 3) Pd model 2 and current wind energy related mortality, 4) Pd model 2 and no wind energy related mortality.

Scenario	2030	2040	2050	2060
Pd Model 1 – Current mortality	804	409	1,071	2,241
Pd Model 1 – No mortality	538	152	446	858
% change	-33%	-63%	-58%	-62%
Pd Model 2 – Current mortality	4,785	2,064	1,108	643
Pd Model 2 – No mortality	3,615	1,055	526	176
% change	-24%	-49%	-53%	-73%

Table A-3D2. Projected median rangewide abundance given wind energy mortality under 4 future conditions scenarios: 1) Pd model 1 and future wind energy related mortality, 2)

Pd model 1 and no wind energy related mortality, 3) Pd model 2 and future wind energy related mortality, 2) Pd model 2 and no wind energy related mortality.

Scenario	2030	2040	2050	2060
Pd Model 1 – low impact mortality	546	108	201	340
Pd Model 1 – future no mortality	719	383	1,026	1,831
% change	-24%	-72%	-80%	-81%
Pd Model 2 – high impact mortality	3960	1197	397	142
Pd Model 2 – future no mortality	4,938	2,210	1,278	857
% change	-20%	-46%	-69%	-83%

Appendix 4: Supplemental Threat and Future Scenario Information

A: WNS

Background

White-nose syndrome (WNS) is a disease of bats that is caused by the fungal pathogen *Pseudogymnoascus destructans (Pd)* (Blehert et al. 2009, entire; Turner et al. 2011, entire; Lorch et al. 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2016, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first observed in eastern New York in 2007 (with photographs showing presence since 2006; Meteyer et al. 2009, p. 411), although it is likely the pathogen existed in North America for a short time prior to its discovery (Keller et al. 2021, p. 3; Thapa et al. 2021, p. 17). Since then, *Pd* and WNS have spread to 39 states and 7 provinces, with lesions indicative of disease confirmed in 12 species of North America bats, including NLEB (Figure A-4A1, www.whitenosesyndrome.org; accessed May 13, 2021; Hoyt et al. 2021, Suppl. Material). *Pd* invades the skin of bats, leading to significant morbidity and mortality that causes drastic declines in multiple species of hibernating bats.

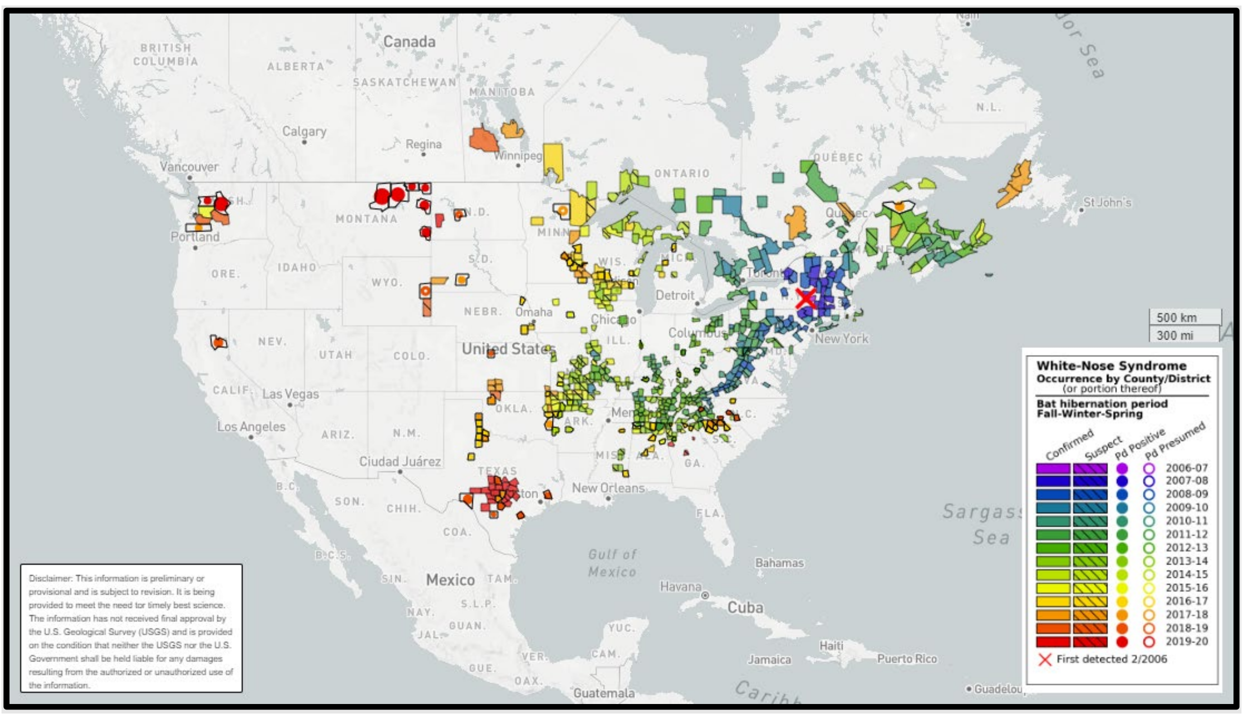

Figure A-4A1. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles) (www.whitenosesyndrome.org, accessed online: May 13, 2020).

White-nose Syndrome

As with any disease, there are three critical elements necessary for WNS to manifest: the pathogen, *Pd*; the host, hibernating bats; and a favorable environment for them to interact, the mainly subterranean hibernacula of bats (Turner et al. 2011, pp. 20–21).

- The *pathogen* that causes WNS, *Pseudogymnoascus destructans* (Gargas et al. 2009, pp. 151–152, Lorch et al. 2011, entire, Minnis and Lindner, 2013, p. 644) grows at cold temperatures ranging from 0–21 degrees C, with optimal growth temperature of 12–16 degrees C (Verant et al. 2012, p. 3), thus it is adapted to grow in conditions characteristic of bat hibernacula. It grows by invading the epidermis and underlying tissues of the face, ears and wings of bats (Meteyer et al. 2009, entire).
- The *hosts*, hibernating bats, are susceptible to infection by *Pd* in part because the physiological, physical and behavioral attributes associated with prolonged use of torpor present the opportunity for this cold-loving fungus to invade their tissues (Lorch et al. 2011, p. 2; Langwig et al. 2012, p. 4; Reeder et al. 2012, p. 4). In particular, hibernating bats overwinter in alternating states of torpor and euthermia (i.e., arousal) to survive prolonged periods without eating (McNab, 1982, p. 171). To use limited fat stores efficiently, metabolic rates are greatly reduced, along with immune functioning and other physiological processes (Moore et al. 2011, p. 8).
- The *environment* where *Pd* and bats interact to cause disease is typically a winter roost location where bats engage in fall swarming and hibernation. The conditions of these locations overlap with the suitable growth requirements for *Pd* (Verant et al. 2012, p. 4). Hibernacula are often assumed to be caves and mines that provide overwinter shelter for large aggregations of hibernating bats, but these essential habitats take many forms and are used by individual bats to large, multi-species colonies. In North America, bats have been documented overwintering in caves, mines, rock crevices, talus, tunnels, bunkers, basements, bridges, aqueducts, trees, earthen burrows, leaf litter, and a variety of other roosts. For bats to hibernate successfully, the most important conditions are relatively stable- low temperatures, but generally above freezing, and high humidity (Perry, 2013, p. 28). Notably, many North American hibernating bats select winter roosts that range between –4 and 16 degrees C (0.6 degrees C to 13.0 degrees C for NLEB; summarized in Webb et al. 1996, p. 763). The overlap of these roost conditions and suitable growth conditions for *Pd* (reported above), combined with the behavioral and physiological characteristics of their torpid state, are the primary factors making hibernating bats so susceptible to infection by *Pd*.

WNS is diagnosed histologically with the identification of "cup-like erosions" as *Pd* invades the skin tissue causing dehydration (Meteyer et al. 2009, p. 412). This fungal invasion destroys the protective skin tissue and disrupts water and electrolyte balance that is important to sustaining homeostasis through hibernation (Cryan et al. 2010, pp. 3–4; Warnecke et al. 2013, pp.3–4). Likely in response to the homeostatic imbalance and irritation of the skin, *Pd* infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves (Reeder et al. 2012, p. 5; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). As a result, WNS leads to starvation as sick bats run out of fat needed to support critical biological functions.

Bats suffering from WNS may exhibit a variety of behavioral changes that can alter the course of morbidity from the disease. In addition to altered arousal patterns, bats have been observed relocating to different areas of hibernacula where conditions may be advantageous for hibernation or disadvantageous for *Pd* growth (Turner et al. 2011, p. 22; Langwig et al. 2012, p. 2; Johnson et al. 2016, p. 189). Observed changes in clustering behavior such that a greater proportion of bats in a colony are seen hibernating solitarily after WNS is present rather than huddled with roost mates may point to a behavioral factor that affects severity of WNS (Langwig et al. 2012, p. 2; Kurta and Smith 2020, p. 769), but may also be a maladaptive response to experiencing symptoms of WNS (Wilcox et al. 2014, p. 162). In many situations, infected bats have been documented exiting hibernacula earlier than usual and prior to when surface conditions are suitable for spring emergence. Early emergence has also been observed during daylight hours when diurnal predators such as hawks and ravens can take advantage of bats weakened by disease. It is possible that bats may find water to drink and insects to prey upon at this time, especially in more moderate climates, thus supplementing depleted energy reserves (Bernard and McCracken, 2017, p. 1492–1493), but in much of NLEB's range, exposure to winter conditions and predation pose a significant threat to animals evacuating from hibernacula. Whether within the roost or on the landscape, WNS causes high rates of mortality during the hibernation season for multiple species (Turner et al. 2011, entire; Cheng et al. 2021, entire).

The weeks following emergence from hibernation also mark a critical period when bats incur energetic costs of clearing infection and recovering from over-winter sickness (Reichard and Kunz 2009, p. 461; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8). Meteyer et al. (2012, p. 3) proposed that bats with WNS can also suffer from immune reconstitution inflammatory syndrome, or IRIS. In this potentially fatal condition, deep or systemic infections that developed during hibernation while immune function was down-regulated trigger an excessive inflammatory response as immune function is upregulated in the spring (Meteyer et al. 2012, p. 5). Additionally, heavily compromised wing conditions resulting from overwinter infections and healing processes are likely to further limit foraging efficiency as the integrity of flight membranes is altered (Reichard and Kunz 2009, p. 462; Fuller et al. 2012, p. 6). These post-emergence complications can lead directly to mortality in addition to impacting reproductive success as a result of energetic constraints and trade-offs (Reichard and Kunz 2009, p. 462; Frick et al. 2010, p. 131; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8).

Transmission of Pd among bats

The fungus is spread via bat-bat and bat-environment-bat movement interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). Transmission occurs primarily in the fall and winter months when bats aggregate in hibernacula (Langwig et al. 2015a, p. 4). In spring, bats that survive a winter exposed to *Pd* can rid themselves of the fungus such that individuals are largely free of *Pd* at summer roosts (Dobony et al. 2011, p. 193; Langwig et al. 2015a, p. 4). However, it is not uncommon for some bats to be found carrying viable *Pd* later into summer (Dobony et al. 2011, p. 193; Ineson, 2020, p. 104) and *Pd* is capable of remaining viable in hibernacula without bats for extended periods (Lorch et al. 2013, p. 1298). The cool, humid conditions of hibernacula likely serve as environmental reservoirs for the fungal pathogen where it can survive and even proliferate until bats return in the fall (Reynolds et al. 2015, p. 320; Hoyt et al. 2020, p. 7259). Generally, bats return to winter roosts in the fall and engage in social interactions that lead to rapid spread of *Pd* from the environmental reservoir to the population (Hoyt et al. 2020, p. 7256). However, because hibernacula may be used throughout the year by males and non-reproductive females who hibernate there, as well as by other species that are more transient, including long distance migrants, some transmission is likely to occur year round and by other mechanisms.

Expansion of Pd in North America

Since it was first detected in New York, the range of *Pd* in North America has increased steadily via bat to bat transmission, although activities of humans, including scientific research, recreational activity, and shipping are also likely to contribute to some short and long distance movements (Bernard et al. 2020, p. 5–6). Simply, *Pd* has spread from just a small number of sites in New York in 2007 to hundreds of locations across the continent in just 14 years. Several predictive models have identified biological, geological, climatic, ecological and behavioral variables correlated with the patterns and timing of its expansion (Hallam and Federico, 2012, p. 2; Maher et al. 2012, p. 3; Alves et al. 2014, p. 2; Hefly et al. 2020, pp. 10–11). Putative barriers to *Pd* expansion have been hypothesized, but these generally have provided very short-term delays in *Pd's* steady progression into uncontaminated areas (Miller-Butterworth et al. 2014, p. 9; Hoyt et al. 2021 p. 3). While these obstacles to natural disease spread may delay arrival of *Pd*, when the fungus does pass them either via dispersing bats or via inadvertent transport by humans, it has led to disease and continued spread of the fungus on the other side (Miller-Butterworth et al. 2014, p. 9; Lorch et al. 2016, p. 4). Because the above published models have fallen behind reality in their predictions, we used two models to describe past occurrence of *Pd* and to predict its future expansion in North America (see *Figure A-2A4, methods described above).*

Establishment of Pd

With the arrival of *Pd* at a new location, progression of the disease proceeds similarly to many emerging infectious diseases through stages of invasion, epidemic, and establishment (Langwig et al. 2015b, p. 196; Cheng et al. 2021, entire). During *invasion* (years 0–1), the fungus arrives on a few bats and spreads through the colony until most individuals are exposed to and carry it. As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7) in the population. Then 8 years after its arrival, *Pd* is determined to be *endemic* (Langwig et al. 2015b, p. 196; Cheng et al. 2022, p. 205). Although methods for detecting *Pd* have changed over time, it is apparent with few exceptions that morbidity and mortality associated with WNS occurs within a year or two after *Pd* has been observed in a population (Frick et al. 2017, pp. 627–629; Hoyt et al. 2020 p. 7259). With the publication by Muller et al. (2013, entire), the use of polymerase chain reaction (PCR) to confirm the presence of *Pd* became the gold standard for diagnosing WNS. This technique provided greater confidence in *Pd* detection and improved our understanding of the disease progression.

Langwig et al. (2015a, pp. 3–4) and Hoyt et al. (2020, p. 7257) quantified the proportion of bats on which *Pd* is detected (prevalence) and the amount of *Pd* on bats (load) in the years after *Pd* invades and establishes itself in a site. In general, when *Pd* is first detectable (by PCR), a relatively small number of bats carry the fungus in low loads. These values increase throughout the first winter at varying rates among species. By the end of the first winter, *Pd* is detectable both on bats and on surfaces of the roost. In the second year after detection, *Pd* loads and prevalence pick up near where they were the previous year; prevalence and load are at significantly higher levels in the fall and early winter, and prevalence approaches 1 (i.e., all bats are infected) by mid-winter for NLEB (Frick et al. 2017, p. 627).

There are a few exceptions in which evidence of *Pd* has been detected in a site and then not detected at that site in subsequent years. These occurrences may represent failed invasions by *Pd.* In Iowa, for example, molecular tests revealed evidence suggestive of *Pd* being present, but WNS was not confirmed at that location for several more years. In California, *Pd* has not been detected in two subsequent years after initial evidence was detected (S. Osborne 2021, California Department of Fish and Game, personal communication). There are also examples that do not fit the expected disease progression described above. At Tippy Dam in Michigan, *Pd* has been present for over 5 years without indication of WNS in little brown bats, although NLEB are no longer observed at this location (Kurta et al. 2020, p. 584). The factors contributing to this atypical scenario are under investigation. It has also been posited that WNS may have a southern limit where disease is less likely to impact populations (Hallam and Federico 2012 p. 9; Hoyt et al. 2021, pp. 6–7). Nevertheless, the overwhelming pattern has been that WNS develops in a population soon after the arrival of *Pd*. Still, because environmental reservoirs of the pathogen play an important role in its transmission, hibernacula that become unsuitable for *Pd* during summer (e.g., too warm or dry) may reduce the amount of fungus in the environment between hibernation seasons, leading to lesser or delayed development of WNS (Hoyt et al. 2020, pp. 7257–7258). To date, these exceptions where colonies experience less severe impacts from WNS compared to the majority of colonies are not reliably predictable based on geographic or biological features, although see *"Persistence of impacted populations"* below.

Impacts of WNS

The impacts of white-nose syndrome are severe among species that were the first observed with the disease. This pattern has remained true over a large area as *Pd* has continued to expand its range affecting previously unexposed colonies of hibernating bats. Four years after the discovery of WNS, Turner et al. (2011) estimated total declines of 98% for NLEB at 42 sites with WNS in Vermont, New York, and Pennsylvania. Later, with data from six states (Vermont, New York, Pennsylvania, Maryland, Virginia, West Virginia), Frick et al. (2015) estimated that median colony size decreased by 90% and NLEB was extirpated from 69% of historical hibernacula (Frick et al. 2015. P. 5). Hoyt et al. (2021, p. 7) summarized overall declines from WNS to be "drastic" for NLEB in both the Northeast and Midwest regions. Using data from 27 states and 2 provinces, the most complete dataset available at the time, Cheng et al. (2021, entire) reported similar patterns. They estimated that WNS has caused 97–100% decline in NLEB across 79% of their range (Figure 4.4; Cheng et al. 2021, entire). Although there are ecological and environmental differences across the currently affected regions of North America, WNS has consistently caused significant declines in populations of NLEB (Figure 4.4), with very few examples of colonies that are avoiding the impacts (Figure 4.6).

Conservation Measures Associated with WNS

There are multiple national and international efforts underway in an attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts.

Efforts associated with the national response to WNS were initially aimed at determining the cause of the disease and reducing or slowing its spread. The response broadened and was formalized by the *National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-nose Syndrome in Bats* which provides the strategic framework for implementation of a collaborative, national response to WNS by State, Federal, Tribal and non-governmental partners (USFWS 2011, entire). The U.S. plan integrates closely with a sister plan for Canada, assuring a coordinated response across much of North America. Implementation of the WNS National Plan is overseen by executive and steering committees comprising representation from the Department of Interior, Department of Agriculture, Department of Defense, and state wildlife agencies under the authority of a multi-species recovery team under the ESA, with the USFWS serving the lead coordinating role. In 2021, the WNS National Plan is being revised to reflect current state of knowledge and identify key elements to continue to effectively respond to this disease. Goals and actions address the greatest needs and knowledge gaps to be pursued, including: coordinated disease surveillance and diagnostic efforts; inter-programmatic data management; development and implementation of disease management, conservation and recovery strategies; and communication and outreach among partners and with the public. These efforts are also supported by the North American Bat Monitoring Program (NABat), which is co-led by USGS and USFWS, to integrate data across jurisdictional borders in support of population level information that supports management decisions at different scales. Actions under the National Plan are intended to be supported through multiple funding programs in different agencies. For several years, many state, Federal, Tribal, and private partners have annually provided funding and physical efforts or both toward WNS research. For its part, the USFWS supports management activities of many partners, research to address key information needs, and development and application of management solutions. The USFWS maintains a website (www.whitenosesyndrome.org) and social media accounts to address many of the communication needs for both internal and external audiences.

Over 100 state and Federal agencies, Tribes, organizations and institutions are engaged in this collaborative work to combat WNS and conserve affected bats. Partners from all 37 states in NLEB's range, Canada, and Mexico are engaged in collaborations to conduct disease surveillance, population monitoring, and management actions in preparation for or response to WNS.

B: Wind

Background

Wind power is a rapidly growing portion of North America's clean energy sector due to its small footprint, lack of carbon emissions, changes in state's renewable energy goals and recent technological advancements in the field allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often have tens or hundreds of turbines installed in a given area, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

Wind related NLEB mortality, while often overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and regional levels. The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[8] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat mortalities continue to be documented at wind power installations across North America.

Mechanism behind bat mortality

Most bat mortality at wind energy projects is caused by direct collisions with moving turbine blades (Grodsky et al. 2011, p. 920; Rollins et al. 2012, p. 365). Barotrauma--a rapid air pressure change causing tissue damage to air-containing structures such as the lungs—may also contribute to bat mortality (Baerwald et al. 2008, pp. 695–696; Cryan and Barclay 2009, p. 1331; Rollins et al 2012, p. 368–369; Peste et al. 2015, p. 11), although impact trauma is likely the cause of most wind-related bat mortality (Lawson et al. 2020; entire).11). Grodsky et al. (2011, 924) further hypothesize that direct collision with turbine blades may cause delayed lethal effects (i.e., injured bats may leave the search area before succumbing to injuries; turbines may damage bats' ears, negatively affecting their ability to echolocate, navigate, and forage), thus causing an underestimation of true bat mortality.

Bats may be attracted to turbines (Solick et al. 2020, entire; Richardson et al. 2021, entire), though support for this is limited. Some hypotheses for bat attraction to wind turbines include the sound of moving blades, blade motion, insect aggregations near these structures, turbines as potential roost structures, and turbines as mating locations (Kunz et al. 2007, pp. 317–319, 321; National Research Council 2007, p. 97; Cryan and Barclay 2009, pp. 1334–1335, Cryan et al. 2014 p. 15128). Horn et al. (2008a, p. 14; 2008b, p. 126) observed bats flying within the turbine blade's rotor swept zone at wind projects in New York and West Virginia and noted that bats were actively feeding and foraging around moving and non-moving blades (2008b, p. 130), while Cryan et al. (2014, p. 15127) observed bats altering course towards turbines using thermal imagery.

Bat mortality tends to exhibit a seasonal pattern, with mortality peaking generally in the late summer and early fall (Erickson et al. 2002, p. 39; Arnett et al. 2008, p. 65; Taucher et al. 2012, pp. 25–26; Bird Studies Canada et al. 2018, pp. 28, 32, 33, 46). Based on our analysis, 6.5, 25.5, and 68.0% of bat fatalities occur during the spring, summer, and fall periods, respectively (USFWS 2016, pp. 4-12, 4-15). Temperature and wind speed may also indirectly influence bats risk of collision risk with wind turbines. Bat activity is higher during nights of low wind speed and warmer temperatures (Arnett et al. 2006, p. 18), and is lower during periods of rain, low temperatures, and strong winds (Anthony et al. 1981, 154–155; Erkert 1982, pp. 201–242; Erickson and West 2002, p. 22; Lacki et al. 2007, p. 89).

[8]Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, p. 32 and 69).

Bat Mortality

Bat mortality varies across wind facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise the majority of all known bat fatalities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other non-listed bat species. However, there is notable spatial overlap between NLEB occurrences and wind facilities along with NLEB mortality documented (Figure 4.7). Based on October 2020 installed MW capacity (Hoen et al. 2018, USFWS unpublished data), we estimated 122 NLEB are annually killed at wind facilities (Table 4.1; Udell et al. 2022, entire). Data from Whitby et al. (2022, entire) analyses suggest that the impact of wind related mortality is discernible in the ongoing decline of NLEB. We compared a no wind baseline scenario to current and future wind scenarios. The percent change in abundance relative to the baseline no wind scenario ranges from a 24% decrease by 2030 under the current wind scenario to a 83% decrease by 2060 under the future high impact wind scenarios (see Tables A-3D1–2). Whitby et al. (2022, entire) found a decline in the predicted relative abundance of NLEB as wind energy risk index increased.

Conservation Measures

To reduce bat fatalities, some facilities “feather” turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein et al. 2021, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed or at a higher threshold, typically referred to as curtailment. The effectiveness of feathering below various cut-in speeds differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with raising cut-in speeds by 1.0–3.0 meters per second (m/s) above the manufacturer’s cut-in speed (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for NLEB has not been documented.

Our wind threat analysis incorporated available curtailment data for existing facilities, and to a limited degree, accounted for future curtailment (see Appendix 2-A). Although effective, curtailment results in energy and revenue losses, which may limit the viability of widespread implementation (Hein and Straw 2021, p. 28). Based on available data (USFWS, unpublished data), most current curtailment is implemented as part of Habitat Conservation Plans developed to support Incidental Take Permits or Technical Assistance Letters detailing methods to avoid incidental take of Indiana bat, and these areas with risk to Indiana bat do not fully overlap with those where NLEB and other species may be susceptible to mortality.

However, there are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. For example, the use of ultrasonic acoustic bat deterrents mounted on turbine towers, blades, and nacelles is an emerging research field showing some promise at reducing bat fatalities (Arnett et al. 2013, entire; Romano et al. 2019, entire; Schirmacher et al. 2020, entire; Weaver et al. 2020, entire; Berthinussen et al. 2021, pp. 88–91). Acoustic-activated “smart” curtailment aims to focus operational curtailment when bat activity is detected in real time (e.g., Hayes et al. 2019, entire; Berthinussen et al. 2021, pp. 105–106; Hein and Straw 2021, pp. 29–30). Additionally, USGS is testing whether illuminating turbines with dim ultraviolet light may deter bats from approaching them (Cryan et al. 2016, entire; Berthinussen et al. 2021, p. 91; Hein and Straw 2021, pp. 23–24). Further, researchers have tested applying a textured coating to the surface of the turbine to alter bats’ perception of the turbine (Bennett and Hale 2019, entire; Berthinussen et al. 2021, pp. 87–88; Hein and Straw 2021, p. 24). These and other methods of reducing bat mortality are still in the research phase, and to date, there are no broadly proven and accepted measures to reduce the severity of impacts beyond various operational strategies (e.g., feathering turbine blades when bats are most likely to be active).

C: Climate Change

Background

There is growing concern about impacts to bat populations in response to climate change (for example, Jones et al. 2009, entire; Jones and Rebelo 2013, entire, O’Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats including changes in hibernation, mortality from extreme drought, cold, or rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed potential impacts of climate change on foraging, roosting, reproduction, and biogeography of bats and also discussed extreme weather events and indirect effects of climate change. However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). In particular, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success. While there may be a variety of ways that climate change directly or indirectly effects NLEB, here we summarize information on the effect of increasing temperatures and changes in precipitation.

Increased annual temperature

Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., annual average temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986 to 2016 relative to 1901 to 1960 (Hayhoe et al. 2018, p. 86). At a regional scale, each National Climate Assessment region also increased in temperature during that time with the largest changes in the west with average increases of more than 1.5 degrees F (0.8 degrees C) in Alaska, the Northwest, the Southwest and the Northern Great Plans and the least change in the Southeast (Hayhoe et al. 2018, p. 86).

Increased annual temperatures are likely to change bat activity and phenology. For example, increased winter temperatures may reduce hibernation period due to longer fall activity or earlier spring emergence (Jones et al. 2009, p. 99). Rodenhouse et al. (2009, p. 250) suggest that hibernation may be shortened by 4 to 6 weeks by the end of this century. Reduced hibernation periods may decrease the duration that an individual bat is exposed to *Pd* and effects of WNS (Langwig et al. 2015a, p. 5).

With increasing temperatures, earlier spring emergence has been documented for cave-roosting bats in Virginia (Muthersbaugh et al. 2019, p. 1). After earlier arrival to summer habitat, if spring weather remains favorable (warm, dry and calm nights providing suitable foraging conditions for bats), this could result in earlier parturition (Racey and Swift 1981, pp. 123–125; Jones et al. 2009, p. 99; Linton and MacDonald 2018, p. 1086) and increased reproductive success (Frick et al. 2010, p. 133; Linton and MacDonald 2018, p. 1086). However, earlier emergence increases the risk of exposure to lethal cold snaps (Jones et al. 2009, p. 99).

Increased temperatures may expand the suitable window for nightly foraging opportunities thereby increasing per night caloric intake. Low ambient temperatures reduce flying insect activity and bat foraging (Anthony et al. 1981, p. 155), while higher average temperatures may result in more frequent suitable foraging nights, particularly during the pre-hibernation fattening period.

Bats that hibernate in temperate regions require temperatures above freezing but cool enough to save energy through torpor (Perry 2013, p. 28). Increased ambient surface temperatures change hibernacula temperatures which then influences their ability to meet the needs of hibernating bats. However, increased ambient surface temperatures will not affect all hibernacula or all parts of a given hibernaculum equally. Hibernaculum microclimate is influenced by a variety of factors including the size, complexity, and location of the site (Tuttle and Stevenson 1978, pp. 109–113). In addition, temperatures of microsites near entrances are strongly correlated to external ambient temperatures compared to microsites deep within hibernacula (Dwyer 1971, p. 427; Boyles 2016, p. 21). Therefore, changes in ambient temperatures are anticipated to result in the greatest changes to portions of hibernacula nearest entrances.

In warmer regions, caves and mines that trap cold air produce beneficial conditions for hibernacula, while in colder regions sites that trap warm air will be more suitable (Perry 2013, p. 33; Kurta and Smith 2014, p. 595). Consequently, a northern site that is suitable today in part for its ability to trap warm air while surface temperatures are very low may become unsuitable as mean annual surface temperature increases.

Indiana bats have been documented to use a wide variety of microclimates within hibernacula and Boyles (2016, p. 34) suggests that the most valuable caves for protection might be the ones with the widest variety of microclimates available. Briggler and Prather (2003, p. 411) similarly found that more tricolored bats were found in caves with wide temperature gradients available. These more complex hibernacula will be less influenced by changes in surface ambient temperatures.

Variations in ambient temperature increase energy expenditure of hibernating bats (Boyles and McKechnie 2010, p. 1645); therefore, stable microsites may be advantageous. Increased ambient temperatures may reduce reliance on relatively stable temperatures associated with underground hibernation sites (Jones et al. 2009, p. 99). However, variation in ambient temperature (e.g., increases in spring) may decrease the energetic costs of arousing from hibernation and serve as a signal that surface conditions are suitable for emergence and foraging (Boyles 2016, p. 36).

Increased hibernacula temperatures may influence overwinter survival rates. If more frequent bat arousals occur, bats will burn through fat reserves more quickly. While insect abundance may also increase in winter, it is unknown whether they will become sufficiently abundant to offset the increased energetic costs associated with more frequent arousal by bats (Rodenhouse et al. 2009, p. 251; Jones and Rebelo 2013, p. 464). Changes to hibernacula temperatures could potentially alter the severity of WNS in these sites (Martínková et al. 2018, p. 1747). For example, a hibernaculum with temperature below the optimal growth rate for *Pd* could shift into the optimal temperature range, thus increasing infection at the site.

Lastly, increased temperatures may result in range shifts of the bats, forest communities, and invasive species. With increasing temperatures, a poleward range expansion of temperate-zone species is predicted (Humphries et al. 2004, p. 154). Kuhl's pipistrelle (*Pipistrellus kuhlii*) has already undergone a substantial northward range shift over the past 15 years (Jones et al. 2009, p. 100), and Lundy et al. (2010, entire) suggested that the migratory Nathusius' pipistrelle (*Pipistrellus nathusii*) has expanded its range in the United Kingdom in response to climate change and will continue to do so. The ranges of European bats are forecasted to show considerable shifts, with species in the Boreal Zone experiencing the greatest change and risk of extinction (Rebelo et al. 2010, p. 568). Many species have little or no overlap between their current and predicted range and face enhanced extinction risk (Rebelo et al. 2010, p. 572). Loeb and Winters (2012, pp. 5–8) found the suitability of an area for Indiana bat maternity colonies declines once the average summer maximum temperature reaches 27.4 degrees C (81.3 degrees F) and predicts a range contraction and northward shift based on climate projections.

Any northern range shifts, however, will be limited based on availability of suitable hibernacula and energetic requirements for hibernation and migration. Humphries et al. (2002, p. 315) predicted that minimum accumulated fat stores of little brown bats are currently inadequate for surviving hibernation throughout the northern portions of the Canadian provinces and the maximum possible fat stores are inadequate for most of Alaska and Canadian territories. When considering a predicted increase of 6 to 8 degrees C (10.8 to 14.4 degrees F), the region of suitable hibernation is expected to expand with a northward shift of approximately 6 km (3.7 mi) per year over the next 80 years (Humphries et al. 2002, pp. 315–316) (Figure A-4C1).

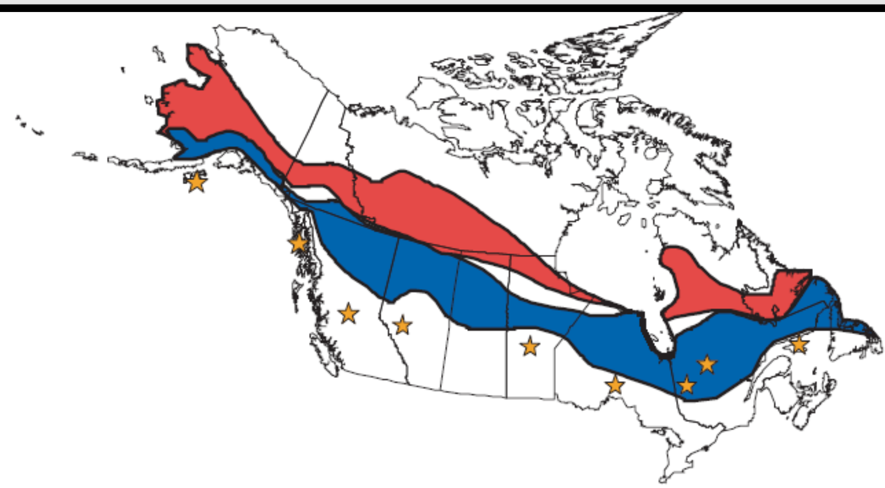

Figure 3 Observed and predicted *M. lucifugus* range distributions in northern North America. Confirmed northern locations of hibernacula (refs 10, 14, 19 and 29, and M. Gauthier, personal communication) are indicated with yellow stars, and the predicted distributional limit of hibernacula is indicated by the blue area; the width of the area represents variation in predictions resulting from cave temperatures exceeding average annual temperature by 0–2 °C, fat stores varying from 1.5 to 3.5 g, and all other parameters varying by up to ±5%. Thus, the lower margin of the blue zone indicates the limit beyond which to the south almost all bats could successfully hibernate, and the upper margin indicates the limit beyond which to the north almost no bats could successfully hibernate. Observed hibernacula approach and only marginally exceed the lower margin of this zone and there is a significant positive correlation between the latitude of observed hibernacula and the lower margin of this zone (predictions generated for each hibernaculum longitude, $n = 9$, $r = 0.93$, $P < 0.001$; intercept not significantly different from 0). The red area represents the predicted 2080 winter range limit of *M. lucifugus* due to global warming. Climate projections are based on the Hadley Centre coupled global climate model (HadCM2—GAX)[12]. The width of the red zone represents the same as does the blue zone.

Figure A-4C1. Observed and predicted little brown bat range distributions in northern North America (from Humphries et al. 2002, Figure 3).

While more northerly sites may become suitable for hibernation, there may be other constraints on successful recruitment at higher latitudes. The active season is shorter in higher latitudes or elevations which may be particularly important for juveniles. Juvenile little brown bats take longer than adults to gain sufficient fat stores for hibernation and shorter active seasons limit their capacity to grow and fatten before their first winter (Kunz et al. 1998, pp. 10–13; Humphries et al. 2002, p. 315). Higher elevations have similar climatic influences as higher latitudes and significantly fewer reproductive female little brown bats are captured at higher elevations in Pennsylvania, West Virginia and Virginia with a similar pattern for tricolored bats in West Virginia (Brack et al. 2002, pp. 24–26).

While bats may be more flexible than other mammals in shifting their ranges, given their ability to fly, the ability of individuals to reach new climatically suitable areas will be impacted by loss and fragmentation of habitat (Thomas et al. 2004, p. 147). The availability of ample suitable roosts may be one of the most limiting resources for bats (Scheel et al. 1996, p. 453). This may be of special concern for tree-dwelling bats since the rate of climate change may be too fast to allow the development of mature forests in the new climatically suitable areas in the north (Rebelo et al. 2010, p. 573).

Changes in Precipitation

Increased temperatures interact with changes in precipitation patterns and results may differ regionally. Annual average precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the Southwest and Southeast (Easterling et al. 2017, p. 208; Hayhoe et al. 2018, p. 88) (Figure A-4C2). The frequency and intensity of heavy precipitation events across the U.S. have increased more than increases in average precipitation (Hayhoe et al. 2018, p. 88).

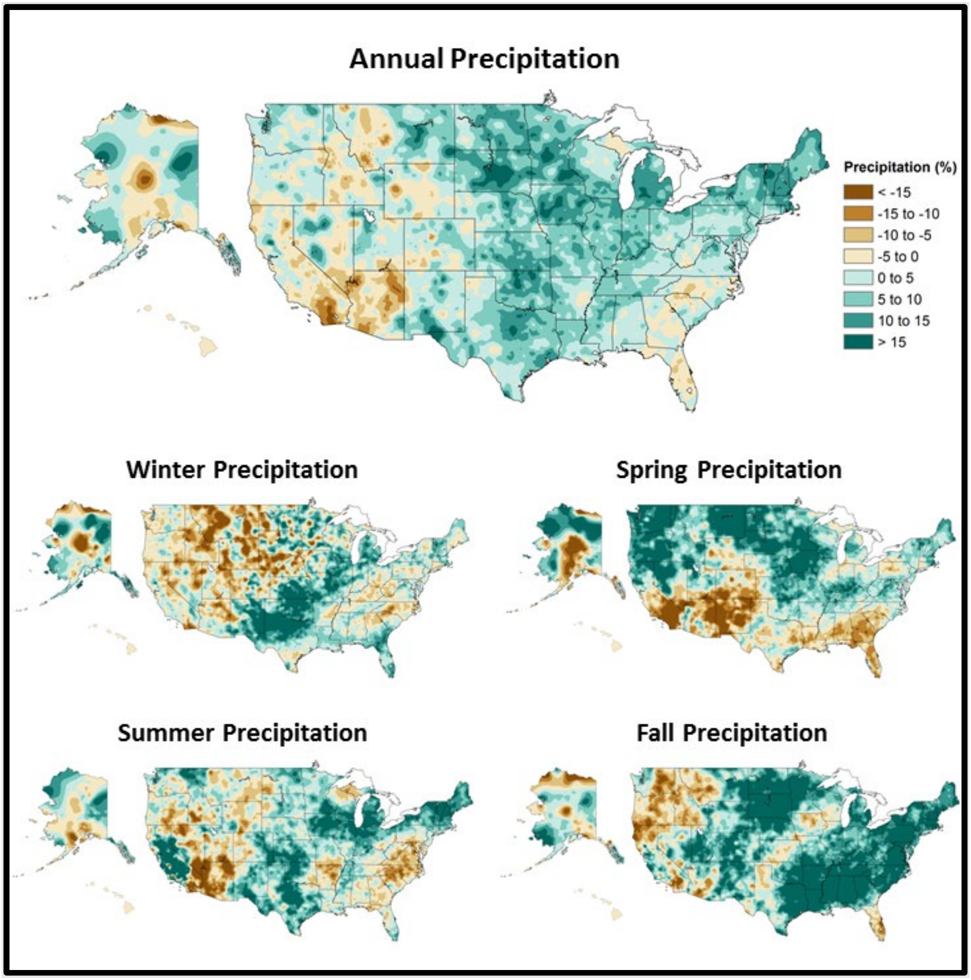

Figure A-4C2. Annual and seasonal changes in precipitation over the U.S. Changes are the average for present-day (1986–2015) minus the average for the first half of the last century (1901–1960 for the contiguous U.S., 1925–1960 for Alaska and Hawai'i) divided by the average for the first half of the century (Easterling et al. 2017, Figure 7.1).

In arid regions, any further reductions in water availability from human uses, reductions in snowpack, or droughts will amplify existing constraints. Spring snow cover extent and maximum snow depth has declined in North America and snow water equivalent and snowpack has declined in the western U.S. (Hayhoe et al. 2018, p. 90). Bats rely on access to free water for thermoregulation, foraging, and reproduction (Adams and Hayes 2008, pp. 1117–1119). In the Rocky Mountains, drought and reduced standing water appears associated with decreased reproduction in bats (Adams 2010, entire). Years that were hotter and drier had a higher incidence of non-reproductive females for all species and 64% of adult female little brown bats were non-reproductive in the drought years of 2007 and 2008 (Adams 2010, pp. 2440–2442) (Figure A-4C3). While cooler and wetter springs resulted in shifts in parturition dates (Grindal et al. 1992, p. 342; Linton and MacDonald 2018, p. 1086), drought years resulted in an overall reduction in the percentage of bats that were reproductive at all (Adams 2010, p. 2442). Readily available water sources appear to be particularly important during lactation (Adams and Hayes 2008, pp. 1117–1120).

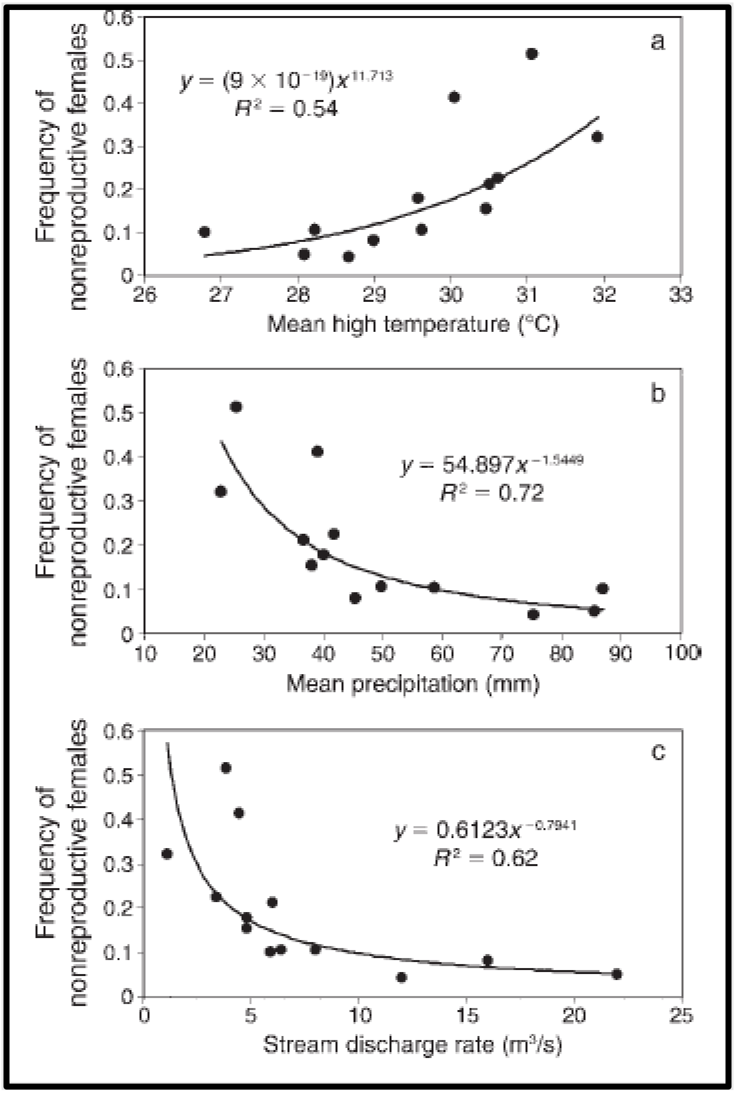

Figure A-4C3. Relationships between the frequency of non-reproductive females captured from 1996 through 2008 and (a) mean high temperature (R -0.74, P = 0.001), (b) mean precipitation (R -0.85, P = 0.0001), and (c) stream discharge rate (R - 0.79, P = 0.001(Adams 2010, Figure 2).

In temperate regions, increased cumulative annual rainfall may lead to increases in the abundance of insects such as dipterans and lepidopterans and is correlated with higher little brown bat survival rates (Frick et al. 2010, pp. 131–133). They suggest that increased insect abundance associated with higher moisture availability was the likely driver and this relationship may vary based on the timing of precipitation (Frick et al. 2010, p. 133). Drying summer conditions may negatively impact aquatic insect prey and little brown bats in the northeastern U.S. (Rodenhouse et al. 2009, p. 250; Frick et al. 2010, p. 133). Small mammals with high energy demands like bats, may be particularly vulnerable to changes in food supply (Rodenhouse et al. 2009, p. 250).

More precipitation has been falling as rain rather than snow in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 90). For example, increased winter temperatures are associated with decreases in Great Lakes ice cover and increases in winter precipitation occurring as rain. The extent and duration of lake ice on the Great Lakes are two of the principal factors controlling the amount of lake-effect snow (provided the air temperatures are sufficiently cool). When large areas of the lakes are covered with ice, the moisture cycle that generates lake-effect snow systems is greatly diminished (Brown and Duguay 2010, p. 692). During the first half of the 20th century there was an increase in snowfall in the Great Lakes Basin; however, recent studies have shown a decline through the latter half of the 20th and early 21st century (Baijnath-Rodino et al. 2018, p. 3947). Similarly, Suriano et al. (2019, pp. 4) found a reduction in snow depth in the Great Lakes Basin of approximately 25% from 1960 to 2009. Trends in snowfall and snow depth during this timeframe are variable by subbasin (Suriano et al. 2019, pp. 5–6) and there was a significant increase of the number of ablation events (i.e., snow mass loss from melt, sublimation, or evaporation) in many areas (Suriano et al. 2019, pp. 6–7). These events are associated with rapid snow melt and often lead to localized flooding. Hibernacula that already faced periodic flooding would be expected to have an increased risk in these areas.

While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795). Given these consequences, bats are likely to reduce their foraging bouts during these heavy rain events.

There is a balancing act that insectivorous bats perform, balancing the costs of flight, thermoregulation and reproduction versus energetic gains from foraging. When bats arrive at maternity areas in the spring, they are stressed after a lengthy hibernation period, a potentially long migration, and the demands of early pregnancy. During this period when their energetic and nutritional requirements are highest, food (flying insects) is relatively scarce, due to cool and wet weather (Kurta 2005, p. 20). Adverse weather, such as cold spells, increases energetic costs for thermoregulation and decreases availability of insect prey (the available energy supply). Bats may respond to a negative energy balance by using daily torpor which conserves consumed and stored energy, and probably minimizes mortality. This has significant implications for their survival or reproduction.

Also, as mentioned above, increased rainfall during pregnancy and lactation may delay parturition or reduce reproductive success (Racey and Swift 1981, pp. 123–125; Grindal et al. 1992, p. 128; Burles et al. 2009, pp. 135–136; Linton and MacDonald 2018, p. 1086). Some females may not bear a pup in years with adverse weather conditions (Barclay et al. 2004, p.

691). Young bats who are born and develop later in the season have less time to develop to successfully forage and to build the fat stores needed to meet the energy demands of migration and hibernation (Humphrey 1975, p. 339). Frick et al. (2010, pp. 131–132) found that little brown bats born even a few weeks later in the summer have significantly lower first-year survival rates and are significantly less likely to return to the maternity colony site to breed in their first year.

Early in the summer, females are under heavy energy requirements to supply their developing fetuses. After giving birth, the adult females experience increased energy needs due to the requirements of lactation and the need to return to the roost during night foraging times to feed their non-volant pups (Murray and Kurta 2004, p. 4). Later in the summer as the pups become volant, these inexperienced and relatively inefficient flyers must expend increased levels of energy as they are growing and learning to feed. Once weaned, young-of-the-year bats must consume enough on their own to migrate to hibernacula and store sufficient fat for the coming winter.

Interaction with WNS-affected Bats

Regardless of the source of increased stress (e.g., reduced foraging, reduced free standing water), because of WNS, there are additional energetic demands for bats. Because WNS causes premature fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, p. 2–3). In addition, WNS-affected bats have wing damage (Meteyer et al. 2009, entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging and results in increased energetic demands associated with the healing process (Davy et al. 2017, pp. 619–612; Meierhofer et al. 2018, p. 487; Fuller et al. 2020, p. 8).

Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. Bats may use torpor to conserve energy during cold, wet weather when insect activity is reduced and increased energy is needed to thermoregulate. However, use of torpor reduces healing opportunities as immune responses are suppressed (Field et al. 2018, p. 3731).

Dobony et al. (2011, entire) observed a little brown bat colony prior to and after onset of WNS impacts and found evidence of lower reproductive rates in the years immediately after WNS was first documented to affect the colony. Francl et al. (2012, p. 36) observed a reduction in juveniles captured pre- and post-WNS in West Virginia, suggesting similarly reduced reproductive rates. Meierhofer et al. (2018, p. 486) found higher resting metabolic rates in WNS-infected (vs. uninfected) little brown bats, suggesting additional energy costs during spring in WNS survivors.

Future climate conditions

Over the next few decades, annual average temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C), relative to 1985 to 2015 regardless of future scenario (Hayhoe et al. 2018, p. 86; Figure A-4C4). Larger increases are projected by late century of 2.3 to 6.7 degrees F (1.3 to 3.7 degrees C) under RCP4.5 and 5.4 to 11.0 degrees F

(3.0 to 6.1 degrees C) and 5.4 to 11.0 degrees F (3.0 to 6.1 degrees C) under RCP8.5, relative to 1986 to 2015 (Hayhoe et al. 2018, p. 86). For the period of 2070 to 2099 relative to 1986 to 2015, precipitation increases of up to 20 and 30% are projected in winter and spring for north central U.S. and Alaska, respectively, with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al. 2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

NLEB’s responses to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as little brown bat (see above). This includes reduced reproduction in the Rocky Mountains due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years in the Northeast (Frick et al. 2010, pp. 131–133). However, the timing of rain events is also important as reduced reproduction has been observed during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will likely vary throughout NLEB’s rangedepending on how much the annual temperature actually rises in the future.

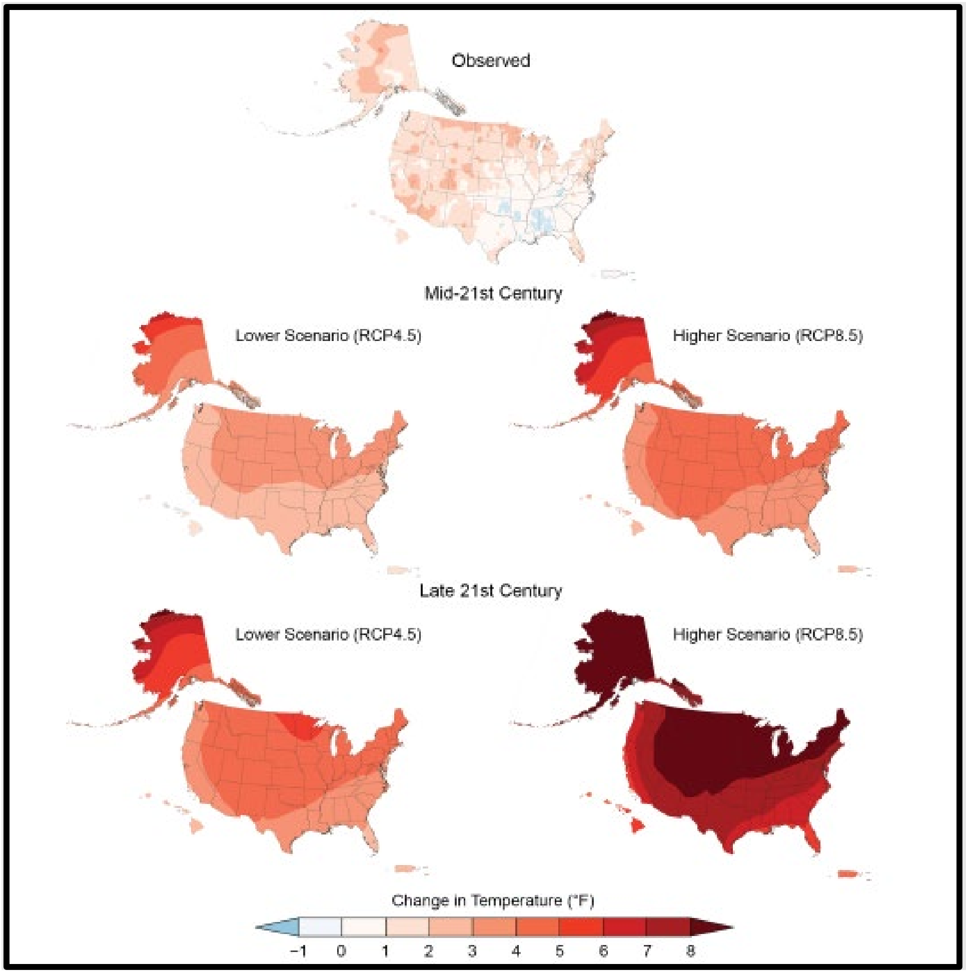

Figure A-4C4. Observed and Projected Changes in Annual Average Temperature (from Hayhoe et al. 2018, Figure 2.4, p. 87).

Climate change may additionally impact these bats in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid, southern pine beetle) (Haavik 2019).

Climate change has also resulted in a rise of global sea level by about 7 to 8 inches (16 to 21 centimeters) since 1993 and relative to the year 2000, sea level is very likely to rise 1 to 4 feet (0.3 to 1.3 meters) (Hayhoe et al. 2018, p. 83). Relative sea level rise is projected to be greater than the global average along the coastlines of the U.S. Northeast and western Gulf of Mexico

(Hayhoe et al. 2018, p. 99), which may reduce access to cave roost along low-lying coastal areas (Jones et al. 2009, p. 101).

Additionally, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success.

D: Habitat Loss

Background

As discussed in Chapter 2, NLEB require suitable habitat for roosting, foraging, and commuting between those habitats during spring, summer, and fall. Forest is a primary component of all of these habitat types, except for the far western portion of the range. Wetlands and water features are important foraging and drinking water sources.

There are a variety of reasons for roosting, foraging, and commuting habitat loss within the range of the NLEB. Hammerson et al. (2017, entire) assessed scope and severity of threat to bats with the highest projected threat impact including invasive species and diseases (particularly WNS); energy production and mining, especially wind energy; human intrusions and disturbance of primarily cave- or mine-dwelling species; and biological resource use, such as tree cutting and forestry practices. Tree cutting and wetland loss can occur from a variety of sources (e.g., development, energy production and transmission, transportation projects). As discussed in Chapter 4, these are increasing across much of the range of the NLEB (USFWS 2015, p. 17991; Oswalt et al. 2019, p. 17) and may result in impacts to the NLEB.

Past and Current

The USFS (2014, p. 7) summarized U.S. forest trends and found a decline from 1850 to the early 1900s, and a general leveling off since that time; therefore, conversion from forest to other land cover types has been fairly stable with conversion to forest (cropland reversion/plantings). In addition, the USFS reviewed U.S. forest trends through 2017 and found forest area trended upward from 1987 to 2012, but since 2012 appears to have reached a plateau (Oswalt et al. 2019, p. 4). About 9.6 million acres (1.4%) of U.S. forest land are affected by tree cutting and removal each year and on an average annual basis, twice as much forest land area (~19 million acres) is affected by natural disturbances that cause either mortality or damage to trees (Oswalt et al. 2019, p. 7). These forest disturbances are attributable to insects and disease (34 percent), fire (21%), weather (16%), and other causes (30%), with importance of disturbance agents varying greatly among geographic regions (Oswalt et al. 2019, p. 7).

In addition to reviewing these reports, we examined more recent (2006 to 2016) change in various NLCD landcover classes within each RPU in the continental U.S. Overall, forest landcover was fairly stable in all RPUs with slight annual increases (27,000 to 50,000 acres/year) in all but Midwest RPU (loss of 23,000 acres/year) (Table A-4D1). However, deciduous forest landcover decreased across all RPUs by 1.4 million acres for an average loss of 140,000 acres per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 1.4 million acres.

Table A-4D1. Changes in land cover types in acres (NLCD 2006-2016) by NLEB RPU occurring within the continental U.S. (Subarctic RPU not included).

	NLEB Representative Units – Change (in acres)				
Land Cover Type	**Southeast RPU**	**Eastern Hardwoods RPU**	**Midwest RPU**	**East Coast RPU**	**All Units**
No Data	0	0	0	0	0
Open Water	-53228	-15513	645390	16451	593100
Developed, Open Space	86718	136193	58923	24639	306472
Developed, Low Intensity	133223	226024	90183	51348	500778
Developed, Medium Intensity	162539	300223	106341	64206	633309
Developed, High Intensity	64748	135120	43896	23717	267481
Barren Land	16701	-767	65283	-3608	77609
Deciduous Forest	-717517	-638191	-24698	-49555	-1429962
Evergreen Forest	920674	-36455	-215544	214328	883003
Mixed Forest	218377	245548	15009	21098	500032
Shrub/Scrub	253128	971856	46115	-114649	1156451
Grassland/Herbaceous	-532118	-520196	-2944844	-164519	-4161676
Pasture/Hay	-888122	-1676022	-2000851	-19983	-4584978
Cultivated Crops	325615	788149	4498629	-37950	5574443
Woody Wetlands	77534	876487	-7942	88299	1034379
Emergent Herbaceous Wetlands	-68273	-792457	-375890	-113822	-1350443
Forest change over 10 years	499068	447390	-233175	274170	987453
Annual average forest change	49906	44738	-23317	27416	98745

Forest ownership varies widely across the species' range in the U.S. As of 2017, private landowners owned approximately 60% of forests (Oswalt et al. 2019, p. 7). Private lands may carry with them a higher risk for conversion than do public forests (since they do not support the same level of regulatory certainty as public lands) a factor that must be considered when assessing risk of forest loss now and in the future (USFWS 2015, p. 17990). Private land ownership is approximately 81% in the East and 30% in the western U.S. (USFS 2014, p. 15). Of the timber harvested annually in the U.S., 89% comes from private lands (Oswalt et al. 2019, p. 9).

Future

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources (we have nothing comparable for Canada). This assessment was influenced by a set of future scenarios with varying assumptions regarding global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010 to 2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million hectares (16–34 million acres or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million hectares (9–21 million acres) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8–7). However, regenerating young forests temporarily lack roosts until suitable tree sizes are reached to provide space and thermal needs for NLEB colonies. In addition, NLEB is not uniformly distributed across the landscape. Loss of essential population needs of roosts and foraging and commuting habitat within NLEB home range where they remain is the issue.

Impacts to bats

These changes in land cover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality (during active season tree removal).

Bats may be directly affected by forest habitat loss by removal of occupied roost trees or loss of roosting and foraging habitat (Farrow and Broders 2011, p. 177). While roosting bats can sometimes flee during tree removal, removal of occupied roosts (during spring through fall) is likely to result in direct injury or mortality to some bats (Belwood 2002, p. 193; McAlpine et al. 2021, p. 2). This is particularly likely during cool spring months (when bats enter torpor) and if flightless pups or inexperienced flying juveniles are also present.

Removal of trees any time of year, including winter, can result in additional impacts depending upon the scope of the action (e.g., acreage of tree removal, locations, and landscape context of the projects) and current understanding or well-supported inferences regarding NLEB presence and use of the area.

Loss of roosts → colony fragmentation → smaller colonies → reduced thermoregulation, reduced information sharing → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival

- reduced adult survival

Loss of roosts, foraging habitat, or travel corridors → displacement → increased flights → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival
- reduced adult survival

Displacement from optimal roosts can also lead directly to increased energy expenditure.

For temperate bats, the requirements for roosting are more restricted and habitat suitable for roosting is rare relative to foraging habitat (Pauli et al. 2015, p. 16); therefore, removal of roosting habitat is more impactful than foraging habitat to these species.

For these species, although loss of a roost is a natural occurrence that temperate bat species must cope with regularly due to the ephemeral nature of tree roosts, the loss of many roosts or an entire home range may result in impacts at the colony level. Bats switch roosts for a variety of reasons, including temperature, precipitation, predation, parasitism, sociality, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264; Barclay and Kurta 2007, p. 34). NLEB is known to switch roosts; therefore, NLEB can tolerate some loss of roosts, provided suitable alternative roosts are available (see Chapter 2). However, loss of central or important roosts can result in colony fragmentation. For example, Silvis et al. (2015, pp. 6–12) found a loss of approximately 17% of roosts may begin to cause colony fragmentation in NLEB. One of the most prominent advantages of colonial roosting is the thermoregulatory benefit (Humphrey et al. 1977, pp. 343–344). Therefore, smaller colonies are expected to provide fewer thermoregulatory benefits for adults in cool spring temperatures and for non-volant pups at any time.

If bats are required to search for new roosting or foraging habitat and to find the same habitats as the rest of their colony finds in the spring, it is reasonable to conclude that this effort places additional stress on pregnant females at a time when fat reserves are low or depleted and they are already stressed from the energy demands of migration and pregnancy. In addition, removal of roosting or foraging habitat may result in longer travel distances between sites used for roosting and foraging. The increased energetic cost of longer commuting distances may result in maternity colony disruption and may be particularly important for pregnant and lactating females and therefore, reproductive success (Lacki et al. 2007, p. 89). NLEB emerge from hibernation with their lowest annual fat reserves, and return to their summer home ranges. Loss or alteration of roosting or foraging habitat puts additional stress on species such as NLEB with strong summer site (i.e., roosting area) fidelity (Foster and Kurta 1999, p. 665; Patriquin et al. 2010, p. 908; Broders et al. 2013, p. 1180), when returning to summer roosting or foraging areas after hibernation. Reproduction is one of the most energetically demanding periods for temperate-zone bats (Broders et al. 2013, p. 1174). Female NLEB produce a maximum of one pup per year; therefore, loss of just one pup results in loss of that entire year's recruitment for females. Limited reproductive potential severely limits the ability of bat populations to respond quickly to perturbations.

Interaction with WNS-affected Bats

Similar to climate change, there are interacting effects of habitat loss with effects from WNS. Regardless of the source of increased stress on bats (roost or foraging habitat removal), because of WNS, there are additional energetic demands for bats associated with healing (Fuller et al. 2020, p. 7). Because WNS causes more frequent arousals (Reeder et al. 2012, pp. 6–9) and fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, p. 7001) and have wing damage (Meteyer et al. 2009, entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging. Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. With increased flights to find suitable habitat or between roosting and foraging habitat comes a trade-off for sufficient energy for survival, recovering from WNS, successful pregnancy or successful rearing of pups.

Roosting/Foraging/Commuting Habitat Loss Conservation Measures

All states have active forestry programs with a variety of goals and objectives. Several states have established habitat protection buffers around known Indiana bat hibernacula that will also serve to benefit other bat species by maintaining sufficient quality and quantity of swarming habitat. Some states conduct some of their own forest management activities in the winter within known listed bat home ranges, as a measure that would protect maternity colonies and non-volant pups during summer months. The USFWS routinely works with project sponsors and Federal agencies to minimize the amount of forest loss associated with their projects and to provide mitigation for impacts associated with forest loss within the range of the federally listed Indiana bat. Examples of largescale efforts to address impacts associated with habitat loss include the rangewide transportation consultation for Indiana bats and NLEB, NiSource Habitat Conservation Plan, and rangewide in-lieu fee program for Indiana bats. Many of the beneficial actions associated with these and similar efforts may benefit other bats if they occur in overlapping ranges. Depending on the type and timing of activities, forest management can be beneficial to bat species (e.g., maintaining or increasing suitable roosting and foraging habitat).

Forest management that results in heterogeneous (including forest type, age, and structural characteristics) habitat may benefit tree roosting bat species (Silvis et al. 2016, p. 37). For example, creation of small canopy openings could increase solar exposure to roosts, leading to warmer conditions that result in more rapid development of NLEB young (Perry and Thill 2007, p. 224). In central Arkansas, female NLEB roosts were more often located in areas with partial harvesting than males, with more male roosts (42%) in unharvested stands than female roosts (24%) (Perry and Thill 2007, pp. 223–224). Silvicultural practices can meet both male and female NLEB roosting requirements by maintaining large-diameter snags in early stages of decay, while allowing for regeneration of forests (Lacki and Schwierjohann 2001, p. 487). Although loss of a roost is a natural phenomenon that bats must deal with regularly, the loss of multiple roosts due to a variety of reasons likely stresses individual bats, as well as the social structure of the colony. Therefore, maintaining roost networks is essential for maternity colony dynamics as colonies may fragment (split into multiple colonies) temporarily with the loss of a primary (central node) roost or multiple alternate roosts (Silvis et al. 2014, pp. 287, 289).

Summary

In summary, U.S. forest area trends have remained relatively stable with some geographic regions facing more loss than others in the recent past. In the future, forest loss is expected to continue, whether from commercial or residential development, energy production, or other pressures on forest lands. Impacts from forest habitat removal to individuals or colonies would be expected to range from minor (e.g., removal of a portion of foraging habitat in largely forested areas with no removal of roosts in areas with robust NLEB populations) to significant (e.g., removal of roosts, removal of a large percentage of summer home range, highly fragmented landscapes, areas with WNS impacts). In areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), impacts would be more likely with a higher probability of removing roosts or causing loss of connectivity between roosting and foraging habitat.

Conservation Measures addressing winter roost loss and disturbance

Protecting these species from disturbance during winter is essential because any additional arousal from hibernation will require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable. This is even more important for sites where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Willis and Boyles 2012, p. 96).

One method of reducing this disturbance is through installation of bat-friendly gates that allow passage of bats while reducing disturbance from human entry as well as changes to the cave microclimate from air restrictions (Kilpatrick et al. 2020, p. 6). Many state and Federal agencies, conservation organizations, and land trusts have installed bat-friendly gates to protect important hibernation sites. The National Park Service has proactively taken steps to minimize effects to underground bat habitat resulting from vandalism, recreational activities, and abandoned mine closures (Plumb and Budde 2011, unpublished data). Further, all known hibernacula within national grasslands and forestlands of the Rocky Mountain Region of the USFS are closed during the winter hibernation period, primarily due to the threat of WNS, although this will reduce disturbance to bats in general inhabiting these hibernacula (USFS 2013, unpaginated). Because of concern over the importance of bat roosts, including hibernacula, the American Society of Mammalogists developed guidelines for protection of roosts, many of which have been adopted by government agencies and special interest groups (Sheffield et al. 1992, p. 707). Also, regulations, such as the Federal Cave Resources Protection Act (16 U.S.C. 4301 *et seq*.), protects caves on Federal lands. Finally, many Indiana bat hibernacula have been gated and some have been permanently protected via acquisition or easement, which provides benefits to other bats that also use the sites.

Appendix 5. Supplemental Future Scenario Descriptions

A summary of the low and high impact scenarios is described below and summarized in Table A-5.1.

Table A-5.1. NLEB composite plausible future scenarios. Pd rate refers to whether % species composition was reduced following Pd arrival.

Plausible Scenario	WNS Spread	WNS Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition	*Pd* rate
Low impact	*Pd* occurrence model 1	15-yr species-specific survival rates	Lower build-out	Regional- specific	U.S. - combined, Canada - regional-specific	No
High impact	*Pd* occurrence model 2	40-yr species-specific survival rates	Higher build-out	Regional- specific	U.S. - combined, Canada - regional-specific	No

WNS

For current projections, we used the two *Pd* occurrence models (see Appendix 2) to assign a WNS stage to all known hibernacula. Table A-5.2 provides the current (2020) number of winter colonies in each of the five WNS stages.

Table A-5.2. Number of NLEB colonies in 2020 per WNS stage under Pd occurrence models 1 and 2.

Model	Pre-arrival	Invasion	Epidemic	Established	Post-established
Pd occurrence model 1	1 (0.1%)	0 (0%)	23 (3%)	320 (44%)	389 (53%)
Pd occurrence model 2	3 (0.4%)	11 (2%)	60 (8%)	140 (19%)	507 (69%)

The difference between the low and high impact scenarios is based on past year of arrival of *Pd* and future rate of *Pd* spread. We used *Pd* occurrence model 1 (Wiens et al. 2022, pp. 226–229) in our low impact scenario and *Pd* occurrence model 2 (Hefley et al. 2020, entire) in our high impact scenario. As *Pd* expands its range, we expect bat populations to be impacted similarly across the species' range. Thus, we apply the same WNS impacts schedule in low and high impact scenarios. Each hibernaculum's population abundance trajectory is divided into three segments with differing λ values: a pre-*Pd*-arrival λ typically $\gtrsim 1$, a *Pd*-arrival λ typically <1, and a post-established λ that can be less than, greater than, or approximately equal to 1. From years since arrival (YSA) 0 to 6, λ varied annually based on results of the status and trends model. We used site specific estimates to the extent possible, although relatively few colonies had sufficient data from counts more than 6 YSA. Therefore, for YSA >6, λ was estimated as the average predicted rate of change in that time period and is held constant through YSA=15 (low impact scenario) and through YSA=40 (high impact scenario). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts under the high impact scenario to 40 years (i.e., the time throughout which WNS will affect survival in the population). To understand the sensitivity of the results to the duration of the disease dynamic and to fully capture the uncertainty, we used the shortest reasonable disease dynamic duration in the low impact scenario. Based on current data (i.e., data from hibernacula documented with WNS in 2008 continue to show impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect populations after *Pd* arrives. After YSA=15 (low impact) or YSA=40 (high impact), λ is assumed to return to pre-WNS rates (i.e., no further WNS impacts applied).

Wind

U.S. Current and Future Wind Capacity

We obtained current wind capacity data for the U.S. from the USWTDB (version 3.2) (Hoen et al. 2018) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we considered projections for 2030, 2040 and 2050 from four potential sources: (1) the U.S. Department of Energy (USDOE) April 2015 Wind Vision report (USDOE 2015, entire) & downloadable data for 2020; (2) the U.S. Energy Information Administration (USEIA) January 2020 Annual Energy Outlook (AEO) report (USEIA 2020, entire) and downloadable data; (3) the USFWS April 2016 Draft Midwest Wind Multi-Species Habitat Conservation Plan (USFWS 2016, Appendix B); and (4) the National Renewable Energy Laboratory (NREL)'s 2020 Standard Scenarios Report (Cole et al. 2020, entire) and downloadable data.

After exploring these data sets and their stated purposes and underlying assumptions and consulting with experts from the USEIA, USDOE, and NREL, we ultimately decided that the NREL Standard Scenarios would serve best for the purposes of our analysis. According to the Standard Scenarios report, it is "o*ne of a suite of National Renewable Energy Laboratory (NREL) products aiming to provide a consistent and timely set of technology cost and performance data and define a scenario framework that can be used in forward-looking electricity analyses by NREL and others. The long-term objective of this effort is to identify a range of possible futures for the U.S. electricity sector that illuminate specific energy system issues. This is done by defining a set of prospective scenarios that bound ranges of technology, market, and macroeconomic assumptions and by assessing these scenarios in NREL's market models to understand the range of resulting outcomes, including energy technology deployment and production, energy prices, and emissions*" (Cole et al. 2020, p. iii).

In addition to a Mid-case Scenario, which uses the reference, mid-level, or default assumptions for all scenario inputs, represents a reference case, and provides a useful baseline for comparing scenarios and evaluating trends, the NREL's 2020 report presents 46 power sector scenarios for the contiguous U.S. (CONUS) that consider the present day through 2050. The NREL report notes, "*the Standard Scenarios are not "forecasts," and we make no claims that our scenarios have been or will be more indicative of actual future power sector evolution than projections made by others*" (Cole et al. 2020, p. 1); however, our experts advised that although the NREL report doesn't calculate a level of probability associated with any given scenario, the Mid-case Scenario is a justifiably reasonable baseline scenario for future wind deployment to use in our analysis.

After further exploring the NREL Standard Scenarios data, we discussed with USDOE and NREL experts the option of using high and low deployment bounds rather than, or in addition to, a reasonable central projection (i.e., Mid-case Scenario). Our experts agreed that this approach would help to capture some of the uncertainty associated with modeled projections; however, we were cautioned not to simply use the lowest and highest deployment scenarios since some scenarios might best be thought of as edge cases intended to show the sensitivity of the model to tweaks in assumptions rather than realistic characterizations of future deployment. Instead, we were advised to use the High and Low Onshore Wind Cost Scenarios as a reasonable combination of scenarios for our SSA analysis, and ultimately decided to apply them as lower and upper bounds, respectively, for the U.S. projections.

The Mid-case, High Wind Cost, and Low Wind Cost Scenarios each implement a slightly different set of assumptions for electricity demand, fuel prices, electricity generation and technology costs, financing, resource and system conditions and more. Under the High Onshore Wind Cost Scenario (our lower bound or "Low Build-out Scenario"), other energy technologies become more cost competitive compared to new wind energy facilities or repowering existing sites. As wind turbines reach their end of life, more are retired than are replaced with newer machines, condensing where wind energy is deployed to only the most optimal sites that present the fewest barriers and the greatest return on investment (B. Straw 2021, personal communication). Therefore, under this scenario, the distribution of wind turbines across the species' range by 2050 is reduced compared to 2020 build-out and total wind capacity decreased for several regions (Table A-5.3), although total U.S. wind capacity is projected to increase slightly. Under the Low Onshore Wind Cost Scenario (our upper bound, or "High Build-out Scenario"), repowering existing wind energy facilities or installing new wind facilities is more cost competitive compared to other energy technologies, resulting in a broader future distribution of wind turbines across the U.S. and higher overall capacity compared to 2020 build-out (Table A-5.3, Figures 4.9–4.11). For a summary of input assumptions used in the Standard Scenarios see Appendix A.1 from the 2020 Standard Scenarios report (https://cambium.nrel.gov/). We assumed total curtailed MW per NREL grid cell would remain unchanged into the future unless MW capacity declined; in these cases, we reduced grid cell curtailment proportionally (e.g., if MW capacity is projected to decline from 10 to 1 MW and currently there is curtailment on 9 MW, there would be 0.9 MW with curtailment and 0.1 MW without curtailment; Udell et al. 2022, entire).

Canada Current and Future Wind Capacity

We obtained current wind capacity data for Canada from the Canadian Wind Turbine Database (CWTD) (Government of Canada 2020, entire). To obtain current and future wind capacity for

Canada, the SSA wind team considered current buildout and projections for 2030, 2040 and 2050 from two sources: (1) The Canadian Wind Energy Association (CanWEA) (CanWEA undated, entire); and (2) The Canada Energy Regulator (CER) Canada's Energy Future 2019 Report (CER 2019, entire). We decided that the CanWEA data would not serve well for our analysis because adequate projections were lacking through the future decades (2020–2050) for most provinces as well as the entire country.

The CER Canada's Energy Future 2019 (EF 19) report is an annual report published by the Government of Canada starting in 2013 and presents projections for wind energy buildout and future capacity through 2040 through updated baseline projections from previous years. According to the report "*the Energy Futures series explores how possible energy futures might unfold for Canadians over the long term. Energy Futures uses economic and energy models to make these projections. They are based on assumptions about future trends in technology, energy and climate policies, energy markets, human behavior and the structure of the economy*" (CER 2019, p. 1). The baseline projections EF 19 are based on one future projection scenario called the Reference Case. According to the report, the Reference Case is "*based on a current economic outlook, a moderate view of energy prices and technological improvements, and climate and energy policies announced and sufficiently detailed for modeling at the time of analysis*" (CER 2019, p. 1).

After we had selected the EF 2019 data for our analysis, the CER published an updated report (EF 20 report) in November 2020 (CER 2020, entire). Similar to previous reports, the EF 20 report presents projections for wind energy buildout and future capacity through updated baseline projections from previous years. Unlike its predecessors, the EF 20 projects buildout scenarios through 2050, 10 years longer than previous years. Additionally, unlike previous reports, the EF 20 Report analyzes two buildout scenarios rather than one: the Evolving Scenario and the Reference (baseline) Scenario. According to the report, the Evolving Scenario "*considers the impact of continuing the historical trend of increasing global action on climate change throughout the projection period. Globally, this implies lower demand for fossil fuels, which reduces international market prices. Advancements in low carbon technologies lead to improved efficiencies and lower costs. Within Canada, we assume a hypothetical suite of future domestic policy developments that build upon current climate and energy policies.*" (CER 2020, p. 4). The 2020 Reference Scenario "*provides an update to what has traditionally been the baseline projection in the Energy Futures series, the Reference Scenario. The scenario considers a future where action to reduce GHG emissions does not develop beyond measures currently in place. Globally, this implies stronger demand for fossil fuels, resulting in higher international market prices compared to the Evolving Scenario. Low carbon technologies with existing momentum continue to improve, but at a slower rate than in the Evolving Scenario*" (CER 2020, p. 4).

In addition to being more up-to-date than the 2019 data, the dual buildout scenarios included in the 2020 Update presented an opportunity to analyze a range of scenarios rather than a single projection and set of assumptions. Therefore, we assigned the Evolving Scenario as an upper bound buildout scenario and the Reference Scenario as a lower bound scenario for our analysis.

Table A-5.3. Wind capacity (MW) by USFWS Region and Canadian Province under 2020 and 2050 low and high scenario build-out.

Location	Wind Capacity (MW)		
	2020 Build-out	2050 Low Build-out (% change)	2050 High Build-out (% change)
Region 3	27,387	15,198 (-45%)	141,573 (+417%)
Region 6	21,280	40,944 (+92%)	83,033 (+290%)
Region 5	6,116	7,252 (+19%)	68,946 (+1027%)
Region 1	7,459	1,422 (-81%)	19,102 (+156%)
Region 8	2,466	1,414 (-43%)	20,624 (+736%)
Region 4	240	391 (+63%)	38,083 (+15768%)
Region 2	39,964	40,511 (+1%)	116,346 (+191%)
U.S. Total	104,912	107,132 (+2%)	487,707 (+365%)
Alberta	1,746	6,699 (+284%)	10,286 (+489%)
British Columbia	732	1,252 (+71%)	1,967 (+169%)
Manitoba	258	476 (+85%)	851 (+230%)
Ontario	5,436	5,646 (+4%)	12,300 (+126%)
Quebec	4,330	5,830 (+35%)	6,930 (+60%)
Atlantic Canada	873	1,408 (+61%)	2,394 (+174%)
Saskatchewan	221	3,256 (+1373%)	5,781 (+2516%)
Canada Total	13,597	24,569 (+81%)	40,510 (+198%)
U.S. + Canada	118,509	131,701 (+11%)	528,217 (+346%)

Species Status Assessment (SSA) Report
for the
Tricolored Bat
(*Perimyotis subflavus*)

Version 1.1

Tricolored bats (Photo credit: Tim Krynak, Cleveland Metroparks)

December 2021
U.S. Fish and Wildlife Service
Northeast Region
Hadley, MA

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[1] This SSA was developed in tandem with analyses and reports for the northern long-eared bat (NLEB; *Myotis septentrionalis*) and little brown bat (LBB; *Myotis lucifugus*).

EXECUTIVE SUMMARY

This report summarizes results of a species status assessment (SSA) conducted for the tricolored bat (TCB; *Perimyotis subflavus*). TCB is a widely distributed small insectivorous bat of eastern North America. Readily identifiable by its tricolored fur, TCB primarily roost in foliage of live and dead trees in the spring, summer, and fall, and hibernate in caves and other subterranean habitats during the winter.

In conducting our status assessment, we first considered what TCB needs to ensure viability. We then considered factors that are currently influencing viability needs or expected to in the future. Based on the species' viability needs and current influences on those needs, we evaluated TCB's current condition. Lastly, we predicted TCB's future condition based on its current condition and expected future influences on viability.

For survival and reproduction at the individual level, TCB require suitable roosting and foraging habitat near abundant food and water resources in the spring, summer and fall; habitat with suitable microclimate conditions for prolonged bouts of torpor and shortened periods of arousal in the winter; and suitable habitat connectivity between summer and winter habitats. For TCB populations to be healthy, they require a population size and growth rate sufficient to withstand natural environmental fluctuations, habitat of sufficient quantity and quality to support all life stages, gene flow among populations, and a matrix of interconnected habitats that support spring migration, summer maternity colony formation, fall swarming, and winter hibernation.

At the species level, TCB require resiliency (demographic, physically, and genetically healthy populations across a diversity of environmental conditions), representation (genetic and ecological diversity to maintain adaptive capacity), and redundancy (multiple and sufficient distribution of populations within areas of unique variation). Resiliency is the ability of the species to withstand environmental and demographic stochasticity and, in the case of TCB, is best measured by the number, distribution, and health of populations across the species' range. Redundancy is an indicator of the ability of a species to withstand catastrophic events by "spreading the risk" and can be measured through the duplication and distribution of resilient populations across the species' range. Representation is an indicator of the ability of a species to adapt to changing environmental conditions and can be measured by the number and distribution of healthy populations across areas of unique adaptive diversity. For TCB, we identified three representation units (RPUs).

The primary factors influencing TCB's viability which have led to its current condition include white-nose syndrome (WNS), wind related mortality, effects from climate change, habitat loss, and conservation efforts.

- WNS is a disease of bats that is caused by the fungal pathogen *Pseudogymnoascus destructans* (*Pd*). *Pd* invades the skin of bats, initiating a cascade of physiological and behavioral processes that often lead to mortality.

- Wind related mortality of TCB is also proving to be a consequential stressor at local and regional levels. TCB are killed at wind energy projects primarily through collisions with moving turbine blades.

- Loss of roosting, foraging, and commuting habitat may lead to minor or significant impacts to TCB depending on the timing, location, and extent of the removal. Loss or modification of winter habitats may also result in negative impacts to TCB, especially given the species' high site fidelity and narrow microclimate requirements for hibernation. Additionally, disturbance (e.g., human entry) during hibernation results in increased arousals in TCB, which leads to increased energy expenditure at a time when food and water resources are scarce or unavailable.

- Changing climatic variables including changes in temperature and precipitation influence TCB's resource needs, such as suitable summer and winter roosting habitat, foraging habitat, and prey availability. Although pervasive across TCB's range, the magnitude, direction, and seasonality of climate change will vary geographically (e.g., some regions will experience more frequent droughts which may lead to reduced TCB survival or reproductive success; alternatively, some regions will experience heavier and more frequent precipitation events that may lead to decreased foraging bouts and insect availability).

- Conservation efforts include multiple national and international initiatives underway in an attempt to reduce the impacts of WNS (to date, however, there are no proven measures to reduce the severity of impacts). Additionally, some wind facilities within TCB's range are implementing curtailment (e.g., feathering turbine blades during low wind periods) to reduce bat fatalities.

We used the best available data to assess TCB viability over time. Winter hibernacula counts provide the most consistent, long-term, reliable trend data, and provide the most direct measure of WNS impacts and thus were used to assess TCB current and future viability. The availability and quality of summer data substantially vary temporally and spatially but were useful for evaluating past population trends. We relied upon the data derived from North American Bat Monitoring Program (NABat) analyses of all available winter and summer data. Current demographic conditions based on past declines indicate TCB's rangewide winter abundance and number of extant winter colonies have declined by 52% and 29%, respectively. TCB winter abundance has declined across all RPUs but varies spatially (24–89%). Declining trends in TCB occurrence and abundance is also evident from summer data: 1) TCB rangewide occupancy declined 28% from 2010–2019; 2) mobile acoustic detections decreased 53% from 2009–2019; and 3) summer mist-net captures declined 12% compared to pre-WNS capture rates. Based on current conditions, future projections of TCB abundance, number of hibernacula, and spatial extent will continue to decline. By 2030, rangewide abundance declines by 89%, the number of winter colonies declines by 91%, and TCB's spatial extent declines by 65%. Projected declines in TCB's abundance, number of winter colonies, and spatial extent are widespread across all RPUs under current conditions.

To assess TCB's future viability, we determined how WNS occurrence and wind energy capacity is likely to change into the future. We described two scenarios that bound our uncertainty on WNS spread and wind energy capacity: 1) WNS spread under Hefley et al. (2020, entire) model and lower wind energy capacity (low impact scenario) and 2) WNS spread under Wiens et al. (2022, pp. 215–248) model and higher wind energy capacity (high impact scenario). Using these scenarios and NABat data, we projected the species' abundance and distribution. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. Under the future scenarios, TCB declines worsen precipitously, with rangewide and RPU-level declines predicted in abundance, number of winter colonies, and spatial extent.

WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future. Wind energy related mortality is also proving, especially in light of the steep declines stemming from WNS impacts, to be a pervasive and consequential driver to TCB's viability, with an estimated 3,327 TCB killed annually at wind facilities across the species' range. Although we consider habitat loss pervasive across TCB's range, severity has likely been low given historical abundance and spatial extent; however, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer winter and summer colonies) negative impacts (e.g., loss of a hibernaculum or maternity colony) may be significant. Lastly, although challenging to describe for such a wide-ranging species, climate change will continue and negative impacts are anticipated in the future.

In summary, TCB abundance has declined significantly and winter abundance, number of occupied hibernacula, spatial extent, and summer habitat occupancy are decreasing. Since the arrival of WNS, TCB abundance steeply declined. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, TCB's ability to recover from these low abundances is limited given their low reproduction output (two pups per year). Therefore, TCB's resiliency is greatly compromised in its current condition and is projected to worsen under future stressor conditions. Additionally, because TCB's spatial extent is projected to decline, TCB will become more vulnerable to catastrophic events. Lastly, the steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reducing TCB's ability to adapt to changes in its biological and physical environments. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit TCB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen TCB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further WNS spread and additional wind energy development, TCB's viability is likely to rapidly decline over the next 10 years.

There is currently uncertainty associated with progression of WNS within TCB winter colonies at road-associated culverts used as hibernacula in the southern U.S. No *Pd* has been detected at culverts in Louisiana and although *Pd* has been detected since 2014 at several culverts in Mississippi, no disease, mortality, or population impacts have been documented. Whether environmental (e.g., shorter and milder winters) or biological factors (e.g., shorter torpor bouts, winter foraging opportunities) contribute to the differences observed at culverts is currently unknown.

Contents

ABBREVIATIONS AND ACRONYMS

%Sp – Percent Species Composition
AC – Adaptive Capacity
AEO – Annual Energy Outlook
AWEA – American Wind Energy Association
AWWI – American Wind Wildlife Institute
Bfat – Bat Fatality
BWEC – Bat Wind Energy Association
C – Celcius
CanWEA – Canadian Wind Energy Association
CC – Climate Change
CE – Catastrophic Event
CER – Canadian Energy Regulator
CI – Confidence Interval
CONUS – Continental United States
CWTD – Canada Wind Turbine Database
DFW – Department of Fish and Wildlife
ESA – Endangered Species Act
F – Fahrenheit
GRTS – Generalized Random-Tessellation Stratified
Hibs – Hibernacula
IUCN – International Union for Conservation of Nature
km – Kilometers
LBB – Little brown bat (*Myotis lucifugus*)
MAST – Mean Annual Surface Temperature
mi – Miles
MLRC – Multi-Resolution Land Characteristics
MW – Megawatts
MYLU – *Myotis lucifugus*
MYSE – *Myotis septentrionalis*
N – Abundance
NABat – North American Bat Monitoring Program
NCSL – National Conference of State Legislatures
NLCD – National Land Cover Database
NLEB – Northern long-eared bat (*Myotis septentrionalis*)
NPS – National Park Service
NREL – National Renewable Energy Laboratory
Pd – Pseudogymnoascus destructans
PESU – *Perimyotis subflavus*
pPg – Probability of Population Growth
RPA – Resources Planning Act
RPU – Representation Unit
SSA – Species Status Assessment

TCB – Tricolored bat (*Perimyotis subflavus*)
USDOE – U.S. Department of Energy
USEIA – U.S. Energy Information Administration
USFS – U.S. Forest Service
USFWS – U.S. Fish and Wildlife Service
USGS – U.S. Geological Survey
USWTDB – U.S. Wind Turbine Database
WNS – White-Nose Syndrome
YOA – Year of Arrival
YSA – Years since Arrival
λ (Lambda) – Population Growth Rate
λavg – Average Population Growth Rate
λtot – Total Population Growth Rate

CHAPTER 1 – INTRODUCTION

This report summarizes the results of a species status assessment (SSA) conducted for the tricolored bat (*Perimyotis subflavus*; TCB). It delivers the best available scientific and commercial information available on TCB in a transparent and defensible peer reviewed report for immediate and future Endangered Species Act (ESA) related decisions. Therefore, while the report is not a decisional document, it does serve as a synthesis of the best available information on the biological status, helpful in promoting the current and future conservation of the species. For this reason, after reviewing this document relative to all relevant laws, regulations, and policies, the U.S. Fish and Wildlife Service (USFWS) plans to utilize the results of this report to make and publish a listing determination in the *Federal Register*.

This chapter describes the analytical framework and methods used to assess TCB's viability over time. Chapter 2 summarizes the ecological requirements for survival and reproduction at the individual, population, and species levels. Chapter 3 summarizes the historical condition of TCB. Chapter 4 describes the key drivers that led to TCB's current condition and the anticipated plausible change in the primary drivers (referred to as influences) over time. Chapter 5 summarizes the current condition assuming no change in influences. Chapter 6 describes the species' future conditions given the plausible projections of the key influences. Lastly, Chapter 7 synthesizes the above analyses and describes how the consequent change in the number, health, and distribution of populations influence TCB viability over time as well as the sources of uncertainty and the implications of this uncertainty. Appendices 1–5 provide further information on uncertainty and sensitivity, supplemental methodology information, supplemental results, supplemental threat background information, and supplemental data.

Analytical Framework

Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308–311). Meaning, to sustain populations over time, a species must have a sufficient number of populations distributed throughout its geographic range to withstand:

(1) environmental and demographic stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

Viability is a measure of the likelihood of sustaining populations over time. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is generally better able to adapt to future changes and to tolerate catastrophes, environmental stochasticity, and stressors, and thus, typically has high viability.

Resiliency is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature and rainfall), periodic disturbances within the normal range of variation (fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

Resiliency is multi-faceted. First, it requires having healthy populations demographically (robust survival, reproductive, and growth rates), genetically (large effective population size, high heterozygosity, and gene flow between populations), and physically (good body condition). Second, resiliency also requires having healthy populations distributed across heterogeneous environmental conditions (referred to as spatial heterogeneity; this includes factors such as temperature, precipitation, elevation, and aspect). Spatial heterogeneity is particularly important for species prone to spatial synchrony (regionally correlated fluctuations among populations). Populations can fluctuate in synchrony over broad geographical areas (Kindvall 1996, pp. 207, 212; Oliver et al. 2010, pp. 480–482) because environmental stochasticity can operate at regional scales (Hanski and Gilpin 1997, p. 372). Spatial heterogeneity induces asynchronous fluctuations among populations, thereby guarding against concurrent population declines. Lastly, resiliency often requires connectivity among populations to maintain robust population-level heterozygosity via gene flow among populations and to foster demographic rescue following population decline or extinction due to stochastic events.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health (Mangel and Tier 1993, p. 1083). For all species, a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000, pp. 307, 309–310; Groves et al. 2002, p. 506). Reducing the risk of extinction due to a single or series of catastrophic events requires having multiple populations widely distributed across the species' range, with connectivity among groups of locally adapted populations to facilitate demographic rescue following population decline or extinction. Redundancy provides a margin of safety to reduce the risk of losing substantial portions of genetic diversity or the entire species to a single or series of catastrophic events.

Representation is the ability of a species to adapt to both near-term and long-term novel or extraordinary changes in the conditions of its environment, both physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (novel pathogens, competitors, predators, etc.). This ability to adapt to changing and novel conditions—referred to as adaptive capacity—is essential for viability as environmental conditions are continuously changing (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either 1) moving to new, suitable environments or 2) by altering (via plasticity or genetic change) their physical or behavioral traits (phenotypes) to match the new environmental conditions (Nicotra et al. 2015, p. 1270; Beever et al. 2016, p. 132).

Maintaining a species' *ability to disperse* and colonize new environments fosters adaptive capacity by allowing species to move from areas of unsuitable conditions to regions with more favorable conditions. It also fosters adaptive capacity by increasing genetic diversity via gene flow, which is, as discussed below, important for evolutionary adaptation (Hendry et al. 2011, p. 173; Ofori et al. 2017, p. 1). Thus, maintaining natural levels of connectivity among populations is important for preserving a species' adaptive capacity (Nicotra et al. 2015, p. 1272).

Maintaining a species' *ability to adapt* to novel and extraordinary conditions requires preserving the breadth of genetic variation. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either genetic change or plasticity (see Text Box

1.1). For adaptation to occur, whether through plasticity or evolutionary adaptation, there must be genetic variation upon which selection can act (Hendry et al. 2011, pp. 164–165; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 326). Without genetic variation, the species cannot adapt and is more prone to extinction (Spielman et al. 2004, p. 15263; also see Text Box 1.1).

Text Box. 1.1. ***Species Adaptation***. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either *genetic change* or *plasticity* (Chevin et al. 2010, pp. 2–3; Hendry et al. 2011, pp. 162; Nicotra et al. 2015, p. 1270). *Genetic change,* referred to as evolutionary adaptation or potential, involves a change in phenotypes via an underlying genetic change (specifically, a change in allele frequency) in response to novel environmental cues (Nicotra et al. 2015, p. 1271; Ofori et al. 2017, p. 2). *Plasticity*, unlike evolutionary adaptation, involves a change in phenotypes (phenotypic plasticity) without undergoing changes in the genetic makeup (Nicotra et al. 2015, pp. 1271–1272). Plasticity is an important mechanism for species to adapt both in immediate and future time frames. In the immediate time frame, plasticity directly acts to allow species to persist despite novel changes in the environment. In the longer time frame, plasticity contributes to a species' adaptive capacity by buying time for adaptive evolution to occur through genetic changes (referred to as genetic assimilation, see Ghalambor et al. 2007, p. 395; Nicotra et al. 2015, p. 1271). Not all genetic and plastic induced changes are adaptive; changes must lead to improved fitness to be adaptive (Nicotra et al. 2015, pp. 1271–1272). Importantly, however, adaptive traits can vary over space and time; what is adaptive in one location may not be adaptive in another, and similarly, what is adaptive today may not be under future conditions and vice versa (Nicotra et al. 2015, pp. 1271–1272). Thus, maintaining the full breadth of variation in both plastic traits and genetic diversity is important for preserving a species' adaptive capacity.

Genetic variation that is adaptive is difficult to identify for a species and represents a significant challenge even when there is genetic information available. To denote variation as 'adaptive' we need to identify which loci are under selection, which traits those loci control, how those traits relate to fitness, and what the species' evolutionary response to selection on those traits will be over time (Hendry et al. 2011, pp. 162–163; Lankau et al. 2011, p. 316; Teplitsky et al. 2014, p. 190). Although new genomic techniques are making it easier to obtain this type of information (see Funk et al. 2019, entire), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation: (1) phenotypic variation; (2) neutral genetic variation; and (3) disjunct or peripheral populations. One of the easiest proxies to measure is variation in biological traits (also described as phenotypic variation). Phenotypic variation, which on its own can be a mechanism for adapting to novel changes, can be due to underlying adaptive genetic variation (Crandall et al. 2000, p. 291; Forsman 2014, p. 304; Nicotra et al. 2015, p. 3). A second proxy for adaptive genetic variation is neutral genetic variation, which is usually the type of genetic data first reported in species-specific genetic studies (see Text Box 1.2). A third, and more distant, proxy for adaptive genetic variation is disjunct or peripheral populations (Ruckelhaus et al. 2002, p. 322). These populations can be exposed to the extremes in habitat/ecological/climate conditions and thus harbor unique and potentially adaptive traits. Similarly, populations that occur across steep environmental gradients can be indicators of underlying adaptive genetic diversity because local adaptation is driven by environmental conditions, which are continually changing at different rates and scales (Sgro et al. 2011, pp. 330, 333).

Text Box. 1.2. ***Genetic diversity***. Genetic variation can be partitioned into two types: adaptive and neutral genetic diversity. Both types are important for preserving the adaptive capacity of a species (Moritz 2002, p. 243), but in different ways. Genetic variation under selection underlies traits that are locally adaptive and that determine fitness (Holderegger et al. 2006, pp. 801, 803; Lankau et al. 2011, p. 316); thus, it is the variation that underpins adaptive evolution (Sgro et al. 2011, p. 328). This type of genetic variation is referred to as adaptive genetic diversity and determines the capacity for populations to exhibit an adaptive evolutionary response to changing environmental conditions. Conversely, neutral genetic variation refers to regions of the genome that have no known direct effect on fitness (i.e., selectively neutral) and change over time due to non-deterministic processes like mutation and genetic drift (Sgro et al. 2011, p. 328). Although, by definition, neutral genetic variation is not under selection, it contributes to the adaptive capacity of a species in a couple of ways. First, neutral genetic variation that is statistically neutral in one environment may be under selection--and thus adaptive--in a different environment (Nicotra et al. 2015, pp. 1271-1272). Second, neutral markers can allow us to infer evolutionary lineages, which is important because distinct evolutionary lineages may harbor locally adaptive traits (Hendry et al. 2011, p. 167), and hence, serve as an indicator of underlying adaptive genetic variation. Thus, maintaining the full breadth of neutral and adaptive genetic diversity is important for preserving a species' adaptive capacity.

Lastly, preserving a species' adaptive capacity requires maintaining the processes that allow for evolution to occur; namely, natural selection and gene flow (Crandall et al. 2000, pp. 290–291; Zackay 2007, p. 1; Sgro et al. 2011, p. 327). Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population via differential survival or reproduction (Hendry et al. 2011, p. 169). To preserve natural selection as a functional evolutionary force, it is necessary to maintain populations across an array of environments (Shaffer and Stein 2000, p. 308; Hoffmann and Sgro 2011, p. 484; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 332). Gene flow serves as an evolutionary process by introducing new alleles (variant forms of genes) into a population, thereby, increasing the gene pool size (genetic diversity). Maintaining the natural network of genetic connections between populations will foster and preserve the effectiveness of gene flow as an evolutionary process (Crandall et al. 2000, p. 293). Preserving genetic connections among populations along with maintaining large effective population sizes will minimize the loss of genetic variation due to genetic drift (Crandall et al. 2000, p. 293). Maintaining large population abundance also fosters adaptive capacity as the rate of evolutionary adaptation is faster in populations with high diversity, which is correlated with population size (Ofori et al. 2017, p. 2).

General Methods

Below we describe our methods for assessing TCB viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and

5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability and extinction risk under both current and future conditions (Figure 1.1). We briefly explain these steps below and provide further details in Appendix 2. Because of the difficulty of delineating populations, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. Henceforth, the terms populations, winter colonies, and hibernacula are used interchangeably.

Like all species status assessments, we do not have perfect information. Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios and making reasonable assumptions based on the best available data. The key uncertainties and how we addressed these uncertainties are described in Appendix 1.

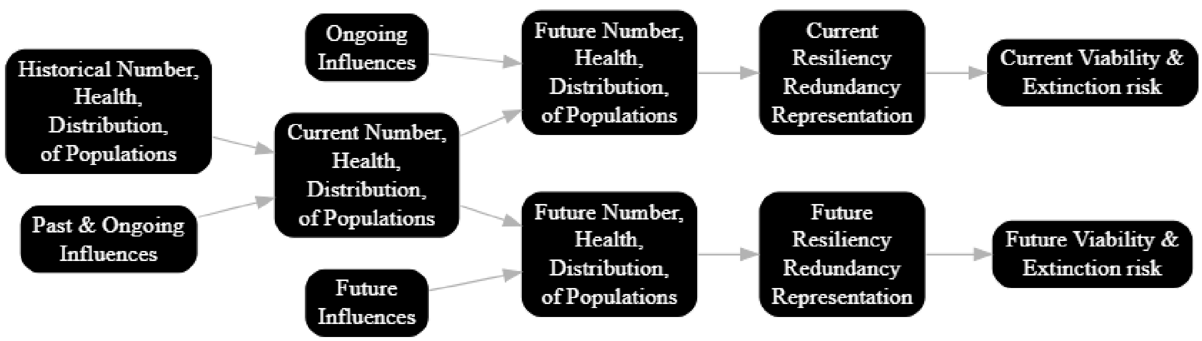

Figure 1.1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time given current and future conditions.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, state and others) across the range to garner the best available and useable data. The majority of these data were collected by state agencies and are now maintained in the North American Bat Monitoring Program (NABat) database, unless otherwise requested by the data contributor or data was not in a format compatible with NABat. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (NABat 2021; accessed February 10, 2021). Winter counts are conducted as internal surveys of caves, mines, tunnels, culverts and other accessible subterranean habitats. Winter counts are conducted in mid to late winter when bats are expected to be less likely to move between hibernacula and prior to spring emergence. Colony counts in hibernacula provide the best estimate of species abundance consistently available for TCB. Colony count data represents the largest amount (geographic and in amount of survey) of abundance data throughout the range of the species. Because not all hibernacula are known and accessible (including extremely scarce information from Mexico and Central America; Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10), we assume that hibernacula for which data are available are representative of all known and unknown hibernacula for the species. Additionally, to provide a non-model approach, we calculated historical abundances by summing the observed counts within each year. To account for missing data, we applied the last observed count. We refer to this third approach as "constant interpolation."

We measured population health as abundance within hibernacula (*N*) and population trend (λ). To estimate historical *N* and *λ*, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233). Using a linear mixed effects model (henceforth, status and trends model), Wiens et al. (2022, pp. 231–247) estimated the yearly population abundance (*N*) from 1990–2020. From these yearly abundances, *λ* was estimated over time for each hibernaculum. For sites with insufficient data-points, Wiens et al. (2022, pp. 231–247) applied λ values from the nearest neighbor. To capture uncertainty in the year of arrival of *Pseudogymnoascus destructans* (*Pd*), they calculated yearly abundance trajectories under two different *Pd* occurrence models (Wiens et al. 2022, pp. 226–229 and Hefley et al. 2020, entire).

Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, pp. 215–251) as described above. Additionally, because bats occupying a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led analyses of available summer capture records and acoustic records to garner additional insights on population trends at regional scales (see Appendix 2A).

Step 3. Identify the Primary Drivers (Influences)

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) drivers of population numbers. We identified white-nose syndrome (WNS), wind related mortality, habitat loss, and climate change as the primary drivers in TCB's abundance.

We qualitatively assessed the scope, severity, and impact of the four stressors using an approach adapted from Master et al. (2012, pp. 28–35) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. The criteria used to assign levels are shown in Figure 1.2.

SCOPE (% of range)	SEVERITY (% of population decline)			
	Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure 1.2. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012, pp. 28–35).

For WNS and wind related impacts, we quantitatively modeled the current and future severity of these stressors. We used an existing demographic population model (BatTool, Erickson et al. 2014) to estimate the impacts (severity) from WNS and wind related mortality (described below).

To assess the impact of WNS and wind related mortality into the future, we used published data, expert knowledge, and professional judgment to form plausible future scenarios. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible "lower" and "upper" impact scenarios. The future scenarios are described in Chapter 4.

To calculate the impact of WNS, Wiens et al. (2022, pp. 231–247) derived the yearly effects of WNS, referred to as "WNS impacts schedule" from winter counts at sites upon WNS arrival (see Appendix 2A for further detail). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and thus, we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts to 40 years (i.e., the time throughout which WNS will affect survival in the population). However, to understand the sensitivity of the results to the duration of disease dynamic and to fully capture the uncertainty, we also incorporated a shorter disease dynamic duration. Based on current data (i.e., data from caves documented with WNS in 2008 continue to show continued impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect a population after *Pd* arrives. Thus, our lower impact scenario assumes a 15-year impact duration (i.e., no further WNS impacts beyond year 15 since *Pd* arrival) and high impact scenario assumes a 40-year impact duration (i.e., the last and least severe WNS disease stage carries through to 2060) (see Appendix 5 for further detail).

To calculate the impact from wind related mortality, we estimated species-specific wind fatality rates as:

$$\text{TCB } per\ MW\ fat\ rate = Bfat * \%Sp$$

Where *Bfat* is the all-bat fatality rate per megawatt (MW) and *%Sp* is the species-specific percent composition of fatalities reported (see Appendix 2A for further details of how *Bfat* and *%Sp* were calculated).

Step 4. Project the Number, Health, and Distribution of Populations under Current and Future Influences

To project future abundance and trend given current and future state conditions for WNS and wind related mortality, we used the population model, BatTool (updated with TCB-specific demographic values). The BatTool projects hibernaculum abundance over time given starting abundance (*N*), trend (λ), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind mortality as specified by the wind capacity scenarios. Starting abundance (*N*) and trend (λ) were derived from Step 2 above. We projected abundance through 2060 to capture the colony response to the 2050 wind energy build-out. Given the species' generation time is 5–7 years, 10 years is sufficient to discern the impacts of the annual mortality levels associated with the 2050 wind capacity build-out.

Using these projected abundance estimates, we calculated various hibernaculum-level and Representation Unit (RPU) metrics to describe the species' historical, current, and future condition (number, health, and distribution of populations) given current and future influences. The results are summarized in chapters 3, 4, and 6. RPUs are further described in Chapter 2.

Step 5. Assess the Current and Future Viability

We evaluated how the change in the number, health, and distribution of populations from historical to present to future influences TCB's ability to withstand stochastic events, catastrophes, and novel changes in its environment, i.e., the 3Rs over time. Specifically, we used the change in the abundance and distribution of winter colonies over time--to evaluate TCB's resiliency to stochasticity, disturbances, and stressors. To assess redundancy, we qualitatively assessed how the current and projected abundance and distribution of colonies affect the risk of catastrophic losses due to extreme weather events and epizootics. To assess TCB's ability to adapt to novel changes in its physical and biological environment, we characterized TCB's adaptability relative to 12 recognized core adaptive capacity attributes (Thurman et al. 2020, entire) and assessed the likelihood of maintaining colonies across the breadth of adaptive diversity given geographic-specific influences and vulnerability to catastrophic events (Appendix 2B).

Summary of NABat Data Sources

Our analyses relied on existing information and upon the data and analyses conducted by NABat. Wiens et al. (2022, entire) provided estimates of past, current, and future abundance based on available winter count data (NABat 2021; accessed February 10, 2021). Deeley and Ford (2022, entire), Stratton and Irvine 2022, entire), and Whitby et al. (2022, entire) provided estimates of population trend since *Pd* arrival based on available summer data (NABat 2020; accessed November 18, 2020). Udell et al. (2022, entire) estimated hibernaculum-specific wind energy mortality estimates. How we used these data are briefly described in Table 1.1 and Figure 1.3, with more detail in Appendix 2. A conceptual model of the BatTool is provided in Figure 1.4. Using Wiens et al. (2022, entire) data, we calculated summary statistics at rangewide and RPU scales over time. For ease of reading, we do not cite the source of the data within the text of Chapters 3–7. In several cases, contributed data could not be utilized in these range-wide analyses due to incompatibility with the database structure of NABat or infeasibility of transferring data files, e.g., New York State Department of Environmental Conservation acoustic data. In these cases, we reviewed any data summaries and analyses provided by the contributing partner and assessed them alongside analyses from NABat.

Table 1.1. NABat analyses used in the SSA. Steps refer to the 5 steps of our analytical approach.

Citation	Data/Analyses	Step in Analytical Process	Chapter
Cheng et al. 2021	Impacts of WNS	Step 3: past WNS impacts	Chapter 4

Citation	Data/Analyses	Step in Analytical Process	Chapter
Cheng et al. 2022	Winter colony count analysis	Step 3: past WNS impacts	Chapter 4
Deeley and Ford 2022	Rangewide analysis of summer capture rates from 1999–2019	Step 2 - Current conditions	Chapter 5
Stratton and Irvine 2022	Rangewide change in occupancy from 2010–2019 based on summer acoustic & mist-net data	Step 2 – Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Whitby et al. 2022	Rangewide analysis of relative abundance based on summer mobile acoustic data from 2009–2019	Step 2 – Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Udell et al. 2022	Estimated wind related bat mortality & allocation to known hibernacula	Step 3. Define future scenarios for wind energy mortality	Chapter 4
Wiens et al. 2022, pp. 231–247	Status & trends linear effects model using winter colony count data	Steps 1 & 2 Historical & current abundance (*N*) and population trend (λ) over time Step 3 past WNS impacts, construct WNS impacts schedule	Chapter 3 Chapters 4, 5
Hefley et al. 2020	*Pd*-occurrence model 2	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future low impact scenario for Pd-spread	NA Chapter 4
Wiens et al. 2022, pp. 226–229	*Pd*-occurrence model 1	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future high impact scenario for Pd-spread	NA Chapter 4
Wiens et al. 2022, pp. 236–247	Future projections of *N* via BatTool	Step 4. Project abundance over time	Chapters 5, 6

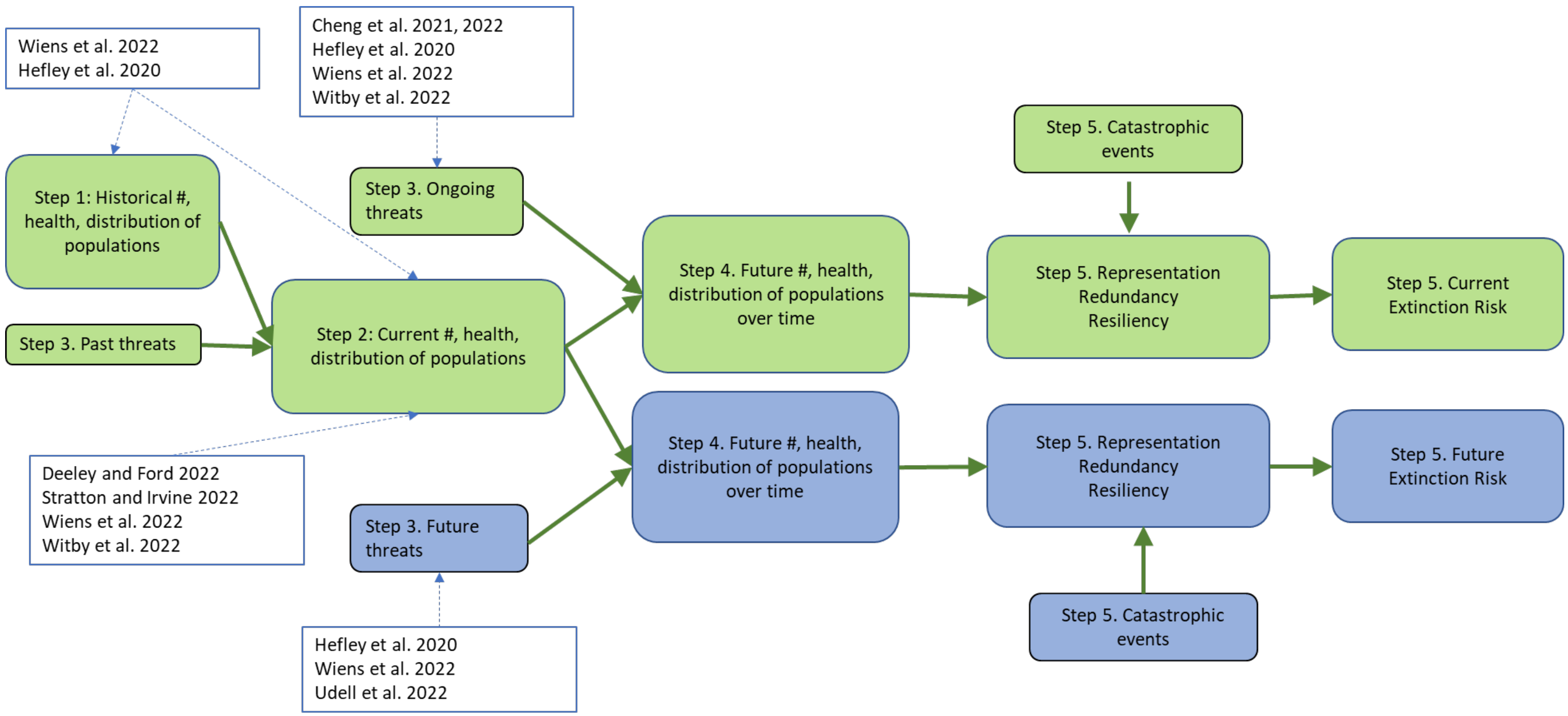

Figure 1.3. A conceptual diagram showing where the NABat data sources are used in our analytical process.

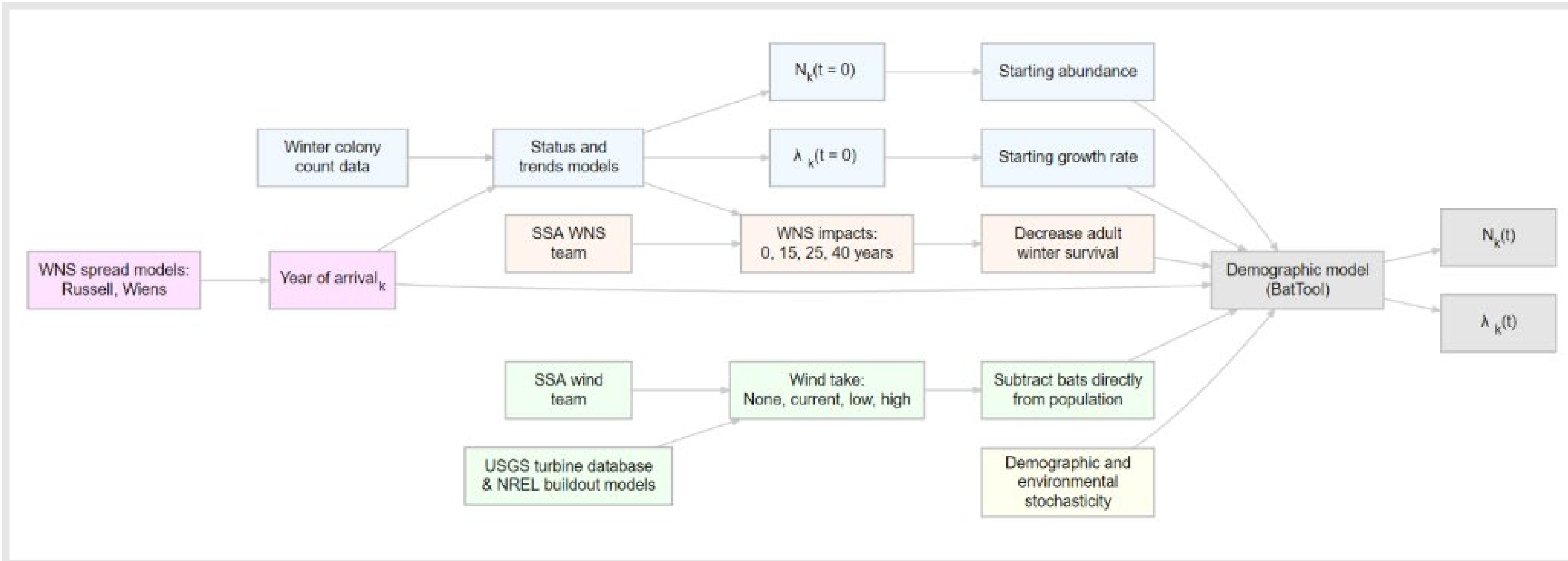

Figure 1.4. BatTool conceptual model. Top (blue boxes): raw data (winter colony) feeds into the status and trends model, which outputs current colony size (N) and population trend (λ) values to input into the BatTool. Middle (pink boxes): 2 Pd occurrence models give Pd year of arrival, which is used in both the status and trends model and BatTool. Middle (peach boxes): SSA core team derived WNS annual impacts schedule, which feeds into the BatTool as decreases in adult winter survival. Bottom (green boxes): SSA core team calculated species-specific bat fatality per MW and USGS projected allocation of this mortality are used to project colony specific mortality over time, which feeds into the BatTool as direct loss of adult females. Far right boxes (gray boxes): projected abundance (N) over time is the output, which is used to calculate colony and RPU level statistics, e.g., λ, number of extant sites, etc

CHAPTER 2 – SPECIES ECOLOGY AND NEEDS

Taxonomy and Genetics

The tricolored bat (*Perimyotis subflavus*; TCB) was first described by Cuvier in 1832 from specimens collected from the eastern U.S., likely Georgia (Fujita and Kunz 1984, p. 1). Various common names have been applied to TCB, including Georgian bat, pigmy bat, southern pipistrel, and most commonly: eastern pipistrelle (Fujita and Kunz 1984, p. 4). In addition, this species has been identified by different scientific names: *Vespertilio subflavus*, *V. erythrodactylus*, *V. monticola*, *Vesperugo veraecrucis* and *Pipistrellus subflavus* (Fujita and Kunz 1984, p. 1). In 1897, Miller (pp. 90–95) placed TCB into genus *Pipistrellus* where it remained until recent phylogenetic analyses confirmed TCB do not share a recent common ancestor with other *Pipistrellus*-like bats and consequently belong in their own genus, *Perimyotis* (Hoofer and Van Den Bussche 2003, pp. 32–34; Hoofer et al. 2006, pp. 982–983). Davis (1959, entire) described four subspecies (*Pipistrellus subflavus subflavus*, *P. s. clarus*, *P. s. floridanus*, and *P. s. veraecrucis*) based on geographic variation in color, size, and cranial measurements (Figure 2.1); an analysis of TCB genetics across its entire range has not been conducted. Furthermore, when the genus reclassification from *Pipistrellus* to *Perimyoti*s was completed, no separate subspecies were proposed. Consequently, we find this point to be more persuasive than the morphological information provided in Davis 1959 (entire). Therefore, for the purposes of this SSA, we considered TCB a valid taxon and monotypic (Hoofer and Van Den Bussche 2003, entire).

As we mentioned above, TCB genetics information is limited. In one study, Martin (2014, entire) examined mitochondrial and microsatellite markers to assess genetic structure across TCB's eastern and midwestern range. Mitochondrial markers separated by large geographic distances were more genetically distinct[2] and suggest two subpopulations across the sampled range (Figure 2.1) (Martin 2014, pp. 20 and 39). Microsatellites, however, suggested very little genetic differentiation (Martin 2014, p. 21). Martin (2014, p. 24) postulated this observed pattern of significant structure in maternally inherited markers (i.e., mitochondrial DNA) with a lack of structure in nuclear markers (i.e., microsatellites) may be the result of male-biased dispersal, but additional analyses are required. Unfortunately, large portions of TCB's range were not sampled. We are unaware of additional genetic studies that analyze TCB population genetics in greater depth and we suggest more research is needed.

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[2] One exception was TCB samples collected from Vermillion, Indiana, which were significantly genetically distinct from all other sites; however, there were haplotypes shared with locations in the *West* population supporting some low but non-negligible intra-regional female dispersal (Martin 2014, pp. 25–26).

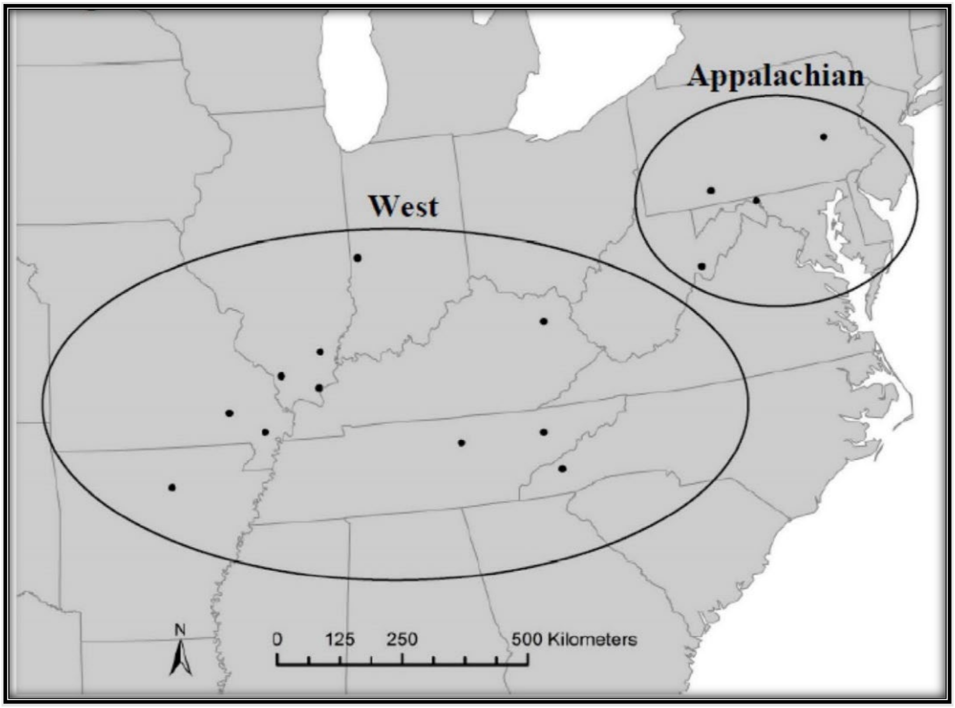

Figure 2.1. Mitochondrial markers suggest two subpopulations across the sampled range (from Martin 2014, p. 39).

Species Description

TCB (Figure 2.2) is one of the smallest bats in eastern North America and is distinguished by its unique tricolored fur that appears dark at the base, lighter in the middle, and dark at the tip (Barbour and Davis 1969, p. 115). TCB often appear yellowish (varying from pale yellow to nearly orange), but may also appear silvery-gray, chocolate brown, or black (Barbour and Davis 1969, p. 115). Males and females are colored alike, but females are consistently heavier than males (LaVal and LaVal 1980, p. 44). Newly volant young are much darker and grayer than adults (Allen 1921, p. 55). Other distinguishing characteristics include 34 teeth (compared with 38 teeth in eastern North American *Myotis* spp. for which it is sometimes confused), a calcar (i.e., spur of cartilage arising from the inner side of the ankle) with no keel, and only the anterior third of the uropatagium (i.e., the membrane that stretches between the legs) is furred (Barbour and Davis 1969, p. 115; Hamilton and Whitaker 1979, p. 85).

Figure 2.2. TCB bat with young (photo credit: Christopher E. Smith, Minnesota Department of Transportation).

Species Range

TCB are known from 39 States (Alabama, Arkansas, Colorado, Connecticut, Delaware, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Nebraska, New Hampshire, New Jersey, New Mexico, New York, North Carolina, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Texas, Vermont, Virginia, Wisconsin, West Virginia, Wyoming), Washington D.C., 4 Canadian Provinces (Ontario, Quebec, New Brunswick, Nova Scotia), and Guatemala, Honduras, Belize, Nicaragua and Mexico (Figure 2.3). The species current distribution in New Mexico, Colorado, Wyoming, South Dakota and Texas is the result of westward range expansion in recent decades (Geluso et al. 2005, p. 406; Adams et al. 2018, entire; Hanttula and Valdez 2021, p. 132) as well as into the Great Lakes basin (Kurta et al. 2007, p. 405; Slider and Kurta 2011, p. 380). This expansion is largely attributed to increases in trees along rivers and increases in suitable winter roosting sites, such as abandoned mines and other human-made structures (Benedict et al. 2000, p. 77; Geluso et al. 2005, p. 406; Slider and Kurta 2011, p. 380).

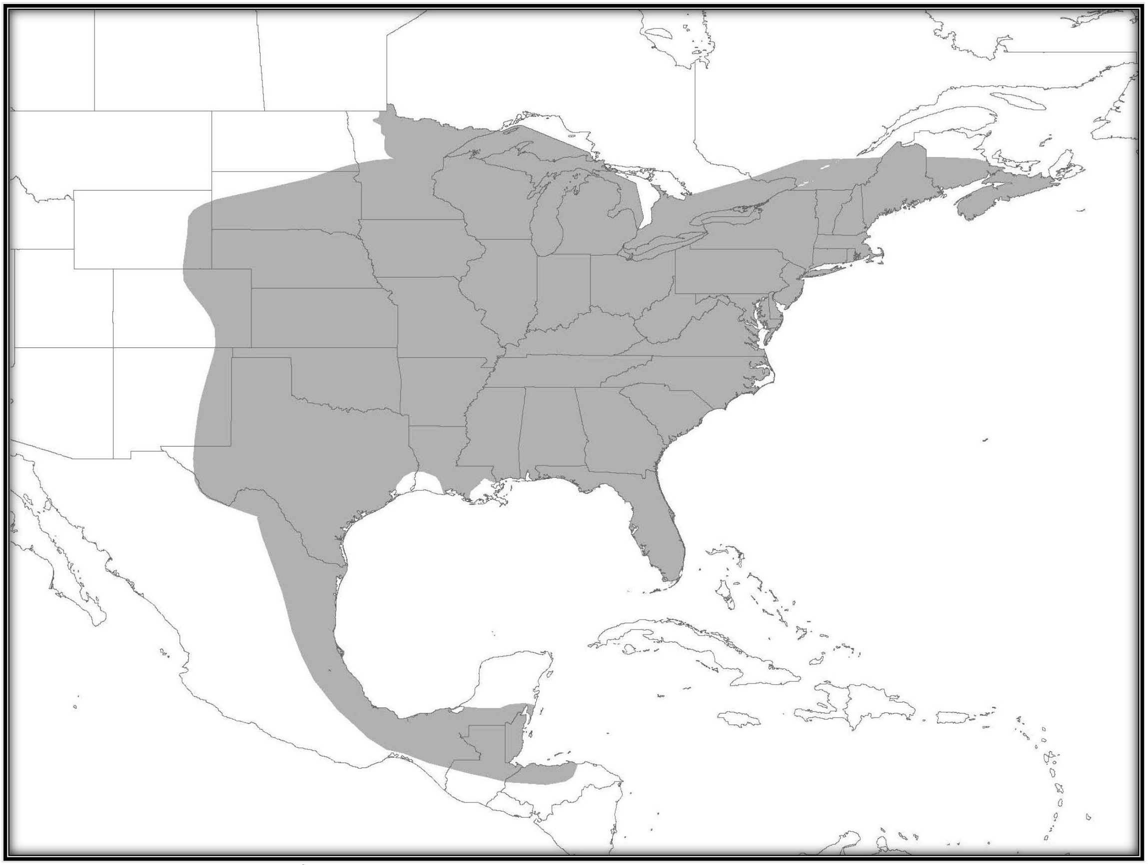

Figure 2.3. TCB range[3].

Individual-level Ecology and Needs

Below we describe the life history and ecological needs for TCB individuals to survive and reproduce; life history and ecological needs are summarized in Table 2.1 and Figure 2.5.

***Longevity*—**The oldest TCB on record is a male captured in Illinois 14.8 years after it was originally banded (Walley and Jarvis 1972, p. 305). Paradiso and Greenhall (1967, pp. 251–252) reported an 11.2 year-old female in West Virginia and 4 additional TCB living at least 10 years. Based on monitoring recaptures over twelve years at two caves in West Virginia, Davis (1966, p. 389) suggested TCB survival is low in the first year, peaks in the third year, and then decreases as maximum life span is approached (Davis 1966, p. 389); however, this study did not account for the possibility that some individuals dispersed to different hibernacula.

***Sheltering*—**During the spring, summer, and fall (i.e., non-hibernating seasons), TCB primarily roost among live and dead leaf clusters of live or recently dead deciduous hardwood trees (Veilleux et al. 2003, p. 1071; Perry and Thill 2007, pp. 976–977; Thames 2020, p. 32). In the southern and northern portions of the range, TCB will also roost in Spanish moss (*Tillandsia usneoides*) and *Usnea trichodea* lichen, respectively (Davis and Mumford 1962, p. 395; Poissant 2009, p. 36; Poissant et al. 2010, p. 374). In addition, TCB have been observed roosting during

[3] Note map does not include single TCB record from northwestern Nicaragua (Medina-Fitoria et al. 2015, p. 49).

summer among pine needles (Perry and Thill 2007, p. 977), eastern red cedar (*Juniperus virginiana*) (Thames 2020, p. 32), within artificial roosts (e.g., barns, beneath porch roofs, bridges, concrete bunkers) (Jones and Pagels 1968, entire; Barbour and Davis 1969, p. 116; Jones and Suttkus 1973, entire; Hamilton and Whitaker 1979, p. 87; Mumford and Whitaker 1982, p. 169; Whitaker 1998, p. 652; Feldhamer et al. 2003, p. 109; Ferrara and Leberg 2005, p. 731; Smith 2020, pers. comm.), and rarely within caves (Humphrey et al. 1976, p. 367; Briggler and Prather 2003 p. 408; Damm and Geluso 2008, p. 384). Female TCB exhibit high site fidelity, returning year after year to the same summer roosting locations (Allen 1921, p. 54; Veilleux and Veilleux 2004a, p. 197). Female TCB form maternity colonies and switch roost trees regularly (e.g., between 1.2 days and 7 days at roost trees in Indiana) (Veilleux and Veilleux 2004a, p. 197; Quinn and Broders 2007, p. 19; Poissant et al. 2010, p. 374). Males roost singly (Perry and Thill 2007, p. 977; Poissant et al. 2010, p. 374).

During the winter, TCB hibernate (i.e., reduce their metabolic rates, body temperatures, and heart rate) in caves and mines, although in the southern U.S., where caves are sparse, TCB often hibernate in road-associated culverts (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Limon et al. 2018, entire; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276) and sometimes tree cavities (Newman 2020, p. 14) and abandoned water wells (Sasse et al. 2011, p. 126). TCB exhibit high site fidelity with many individuals returning year after year to the same hibernaculum (Davis 1966, p. 385; Jones and Pagels 1968, p. 137; Jones and Suttkus 1973, p. 964; Sandel et al. 2001, p. 175).

TCB are one of the first cave-hibernating species to enter hibernation in the fall and one of the last to leave in the spring in Missouri and Pennsylvania (LaVal and LaVal 1980, p. 29; Merritt 1987, p. 102). In the southern U.S., hibernation length is shorter compared to northern portions of the range and some TCB exhibit shorter torpor bouts and remain active and feed during the winter (Layne 1992, pp. 43–44; Grider et al. 2016, p. 8; Limon et al. 2018, p. 219; Newman 2020, pp. 13–17; Stevens et al. 2020, p. 528). The number of hibernating TCB does not peak at caves and mines until December or later, suggesting some bats stay on the landscape or in alternate hibernacula and only move in to caves and mines when it gets colder (Barbour and Davis 1969, p. 119; Vincent and Whitaker 2007, p. 61), although, in some cases, TCB may remain on the landscape and hibernate in rock shelters (e.g., fissures in sandstone and sedimentary rock) (Johnson 2021, pers. comm.).

TCB are often found hibernating at warmer locations within caves and mines compared to other cave-hibernating bat species within these locations (Barbour and Davis 1969, p. 119; Raesly and Gates 1987, p. 17). TCB was observed hibernating at a mean temperature of 51.6 degrees Fahrenheit (F; 10.9 degrees Celsius (C)) (range 50.5 – 52.5 degrees F (10.3–11.4 degrees C)) at caves and mines in Pennsylvania, Maryland, and West Virginia (Raesly and Gates 1987, p. 18). TCB are also found in areas of caves and mines with high humidity (e.g., 99%; Mohr 1976, p. 97) and were not observed in caves where relative humidity was below 80% (Ploskey and Sealander 1979, p. 72).

Hibernating TCB do not typically form large clusters; most commonly roost singly, but sometimes in pairs, or in small clusters of both sexes away from other bats (Hall 1962, p. 29; Barbour and Davis 1969, p. 117; Mumford and Whitaker 1982, p. 169; Raesly and Gates 1987,

p. 19; Briggler and Prather 2003, p. 408; Vincent and Whitaker 2007, p. 62). TCB roost on cave walls (more often) and ceilings and are rarely found in cave crevices (Mumford and Whitaker 1982, p. 169). TCB will shift roosts from one to another during the winter but arouse less frequently than other cave-hibernating bat species (Barbour and Davis 1969, p. 119; Mumford and Whitaker 1982, p. 169); consequently, sometimes water beads will collect on their fur making them appear almost white (Hamilton 1943, p. 86; Barbour and Davis 1969, p. 119). In road associated-culverts in the southern U.S., however, TCB exhibit shorter torpor bouts and move within and between culverts throughout the winter (Anderson et al. undated).

TCB hibernate in more caves and mines than any other cave-hibernating bat species in eastern North America (Sealander and Young 1955, pp. 23–24; Barbour and Davis 1969, p. 117; Brack et al. 2003, p. 65). TCB may use small caves and mines that are unsuitable to other cave-hibernating bat species (Barbour and Davis 1969, p. 117; Mumford and Whitaker 1982, p. 168; Hamilton and Whitaker 1979, p. 87); however, hibernating TCB have been observed in greater numbers in hibernacula with stable temperatures (Briggler and Prather 2003, p. 411). Raesly and Gates 1987 (p. 19) found TCB hibernating in 80% of the 50 locations surveyed in Pennsylvania versus little brown bats (*Myotis lucifugus*), Indiana bats (*M. sodalis*), northern long-eared bats (*M. septentrionalis*), and big brown bats (*Eptesicus fuscus*) which were found in 56%, 16%, 16%, and 34% of potential hibernacula, respectively. Almost every cave in Indiana has contained at least one TCB (Mumford and Whitaker 1982, pp. 167–168); and small numbers of TCB have likely occupied most of Missouri's 6,400 caves (Perry 2021, pers. comm.).

Prior to the arrival of WNS (see Chapter 4), hibernating TCB colonies varied between 1 and 5,300 individuals; however, 40% of hibernacula had between just 1 and 10 individuals (Figure 2.4). The largest TCB hibernating colony (n = 5,300) was observed in Georgia in 2010 (NABat 2021).

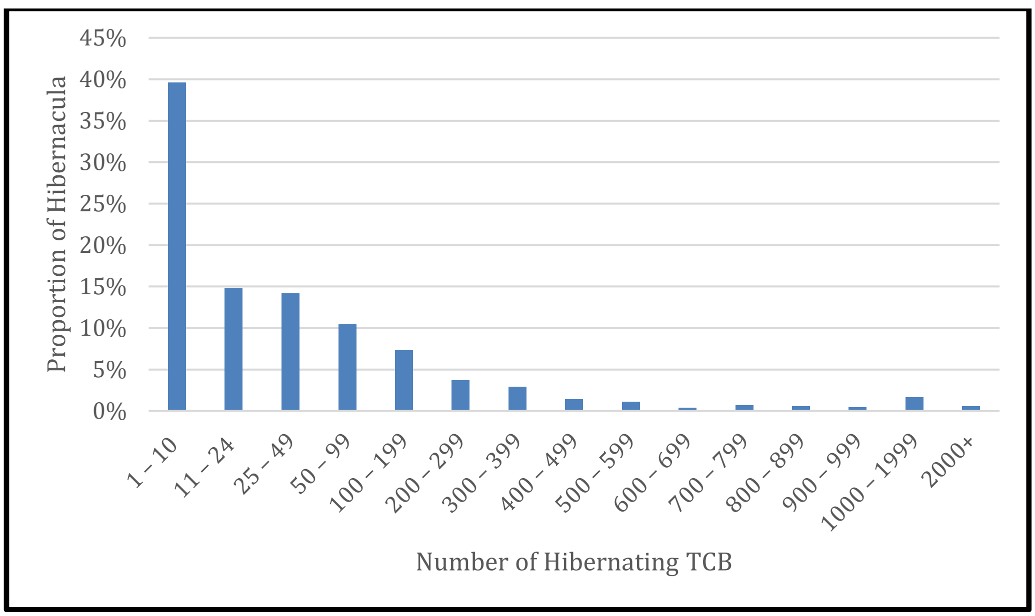

Figure 2.4. Total number of hibernating TCB observed during winter counts at hibernacula (n=1,236) prior to the arrival of white-nose syndrome (NABat 2021; accessed February 10, 2021).

***Feeding*—** TCB are opportunistic feeders and consume small insects including caddisflies (Trichoptera), flying moths (Lepidoptera), small beetles (Coleoptera), small wasps and flying ants (Hymenoptera), true bugs (Homoptera), and flies (Diptera) (Whitaker 1972, p. 879; LaVal and LaVal 1980, p. 24; Griffith and Gates 1985, p. 453; Hanttula and Valdez 2021, p. 132). TCB emerge early in the evening and forage at treetop level or above (Davis and Mumford 1962, p. 397; Barbour and Davis 1969, p. 116) but may forage closer to ground later in the evening (Mumford and Whitaker 1982, p. 170). TCB exhibit slow, erratic, fluttery flight while foraging (Fujita and Kunz 1984, p. 4) and commonly forage with eastern red bats (*Lasiurus borealis*) and silver-haired bats (*Lasionycteris noctivagans*) (Davis and Mumford 1962, p. 397; Mumford and Whitaker 1982, p. 169). TCB forage most commonly over waterways and forest edges (Barbour and Davis 1969, p. 116; Mumford and Whitaker 1982, pp. 170–171; Hein et al. 2009, p. 1204). Maximal distance traveled from roost areas to foraging grounds was 4.3 kilometers (km; 2.7 miles) for reproductive (pregnant or lactating) adult females in Indiana (Veilleux et al. 2003, p. 1074) and 24.4 km (15.2 miles) (mean=11.4 km; 7.1 miles) for male TCB in Tennessee (Thames 2020, p. 61).

***Reproduction*—**Male and female TCB converge at cave and mine entrances between mid-August and mid-October to swarm and mate. Adult females store sperm in their uterus during the winter and fertilization occurs soon after spring emergence from hibernation (Guthrie 1933, p. 209). Females typically give birth to two young, rarely one or three between May and July (Allen 1921, p. 55; Barbour and Davis 1969, p. 117; Cope and Humphrey 1972, p. 9). Young grow rapidly and begin to fly at 3 weeks of age and achieve adult-like flight and foraging ability at 4 weeks (Lane 1946, p. 59; Whitaker 1998, pp. 653–655). Adults often abandon maternity roosts soon after weaning, but young remain longer (Whitaker 1998, p. 653). TCB are considered juveniles (i.e., subadults) when entering their first hibernation and most probably do not mate their first fall (Fujita and Kunz 1984, p. 3).

Maternity colonies consist of 1 to 8 (mean = 4.4) females and pups at tree roosts in Indiana (Veilleux and Veilleux 2004b, p. 62). Perry and Thill 2007 (p. 977) observed an average of 6.9 adult females and pups per colony in Arkansas (range 3 to 13). Maternity colonies include up to 18 females in trees in Nova Scotia (Poissant et al. 2010, p. 374). Whitaker (1998, p. 652) found colonies in buildings averaged 15 adult females (range 7 to 29 adult females). Hoying and Kunz 1998 (p. 19) reported the largest colony on record in a Massachusetts barn (19 adult females and 37 young).

***Movement/Dispersal*—** TCB disperse from winter hibernacula to summer roosting habitat in the spring. Fraser et al. 2012 (p. 5) concluded that at least some TCB engage in latitudinal migration that is more typically associated with hoary bats (*Lasiurus cinereus*), eastern red bats, and silver-haired bats, and this behavior is more common for males than for females. The maximum migration distance on record is a female TCB who migrated a straight-line distance of 243 km (151 miles) from her winter hibernaculum in southern Tennessee to a summer roost in Georgia (Samoray et al. 2019, p. 17). Other migration records between winter hibernacula and summer habitat include less than 80 km (50 miles) (Barbour and Davis 1969, p. 117), 44 km (27 miles) (Samoray et al. 2019, p. 18), and 137 km (85 miles) (Griffin 1940, p. 237). Hibernaculum to hibernaculum movement up to 209 km (130 miles) has also been documented between two consecutive winters (Lutsch 2019, p. 38).

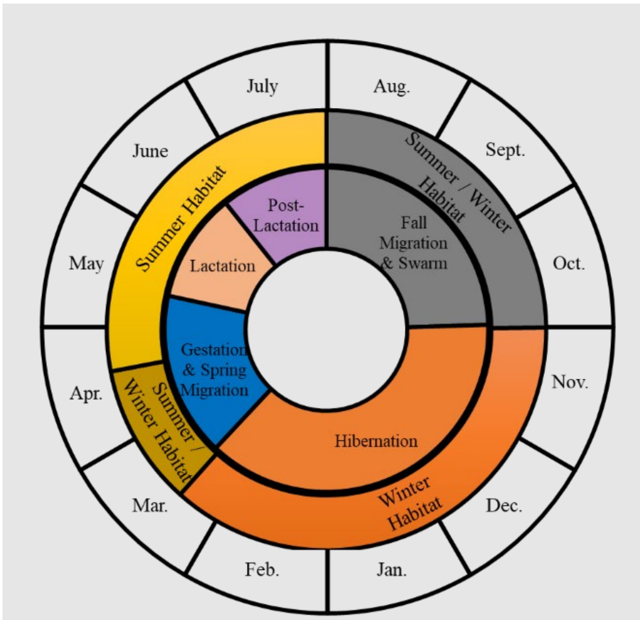

Figure 2.5. Generalized annual life history diagram for TCB (adapted from Silvis et al. 2016, p. 1).

Table 2.1. TCB individual-level needs for survival and reproduction.

LIFE STAGE	SEASON
Pups	<u>Summer</u> - roosting habitat with suitable conditions for lactating females and for pups to stay warm and protected from predators while adults are foraging.
Juveniles	<u>Summer</u> - other maternity colony members (colony dynamics, thermoregulation); suitable roosting and foraging habitat near abundant food and water resources. <u>Fall</u> - suitable roosting and foraging habitat near abundant food and water resources. <u>Winter</u> - habitat with suitable microclimate conditions.
Reproductive Females	<u>Summer</u> - other maternity colony members (colony dynamics); network of suitable roosts (i.e., multiple summer roosts in close proximity) near conspecifics and foraging habitat near abundant food and water resources.
All Adults	<u>Spring</u> - suitable roosting and foraging habitat near abundant food and water resources; habitat connectivity and open air space for safe migration between winter and summer habitats. <u>Summer</u> - roosts and foraging habitat near abundant food and water resources. <u>Fall</u> - suitable roosting and foraging habitat near abundant food and water resources; cave and/or mine entrances (or other similar locations, e.g., culvert, tunnel) for conspecifics to swarm and mate; habitat connectivity and open air space for safe migration between winter and summer habitats. <u>Winter</u> - habitat with suitable microclimate conditions.

Population-level Needs

To be self-sustaining, a population must be demographically, genetically, and physically healthy (see Redford et al. 2011, entire). Demographically healthy means having robust survival, reproductive, and growth rates. Genetically healthy populations have large effective population sizes (*Ne*), high heterozygosity, and gene flow between populations. Physically healthy means individuals have good body condition. The population-level ecological requirements of a healthy TCB population are discussed below and summarized in Figure 2.6 and Table 2.2.

Demography

For TCB populations to have a healthy demography, the population growth rate (lambda or λ) must be sufficient to withstand natural environmental fluctuations. At a minimum, λ must be at least one for a population to remain stable over time. To maintain a healthy λ and *Ne*, TCB, are dependent on their ability to select environments with ample prey and appropriate conditions at summer and winter roosting habitat. For example, TCB winter roosts require stable microclimates within narrow temperature and humidity ranges, and low levels of disturbance. During favorable hibernating conditions, TCB survival and therefore reproductive rates are high (increasing λ); conversely, when environmental conditions are unfavorable, survival and reproductive rates are low (decreasing λ). Growth rates are not expected to vary across TCB's range and population numbers generally do not experience extreme variation from year-to-year or successive generations.

Habitat Quality and Quantity

To support a strong growth rate, TCB populations benefit from large population sizes (which helps maintain genetic health via large *Ne*) and sufficient quality and quantity of habitat to accommodate all life stages. The required habitat quality to support healthy demographic rates and physical health is described under *Individual-level Ecology and Needs*. The quantity of habitat is likely to vary among populations, but will likely hinge on the availability of roosting habitat in the summer and suitable hibernacula in the winter. Limited research suggest the minimum summer roost area (not including foraging area) for individual adult female TCB ranges between 0.1 and 2.2 hectare (ha) (0.25 and 5.4 acre [ac]) (Veilleux and Veilleux 2004a, p. 197). Mean foraging area for 7 adult male TCB was 2,350 ha (5,807 acres) (range 234–9,655 ha; 578–23,858 acres) and 364 ha (899 acres) for a single non-reproductive female in Tennessee (Thames 2020, p. 61). Although TCB hibernate in more caves and abandoned mines than any other cave-hibernating bat species in eastern North America (see *Individual-level Ecology and Needs*), higher numbers of TCB have been observed in caves with stable temperatures (Briggler and Prather 2003, p. 411). More research is needed on the specific optimal quality of TCB habitat.

Connectivity

To support all life stages, TCB populations require a matrix of interconnected habitats that support spring migration, summer maternity colony formation, fall swarming, and winter hibernation. For these populations, movement among habitats is needed to maintain genetic diversity and to allow recolonization in the event of local extirpation. TCB migrate up to 243 km (151 miles) between winter hibernacula and summer roosting sites (Samoray et al. 2019, p. 17). TCB prefer landscapes with greater forest area, forest aggregation, and tree corridors and are less abundant among urban development (Duchamp and Swihart 2009, p. 855; Farrow and Broders 2011, p. 177). Thus, large stretches of urban development (i.e., less suitable habitat) may negatively influence connectivity between summer and winter habitats.

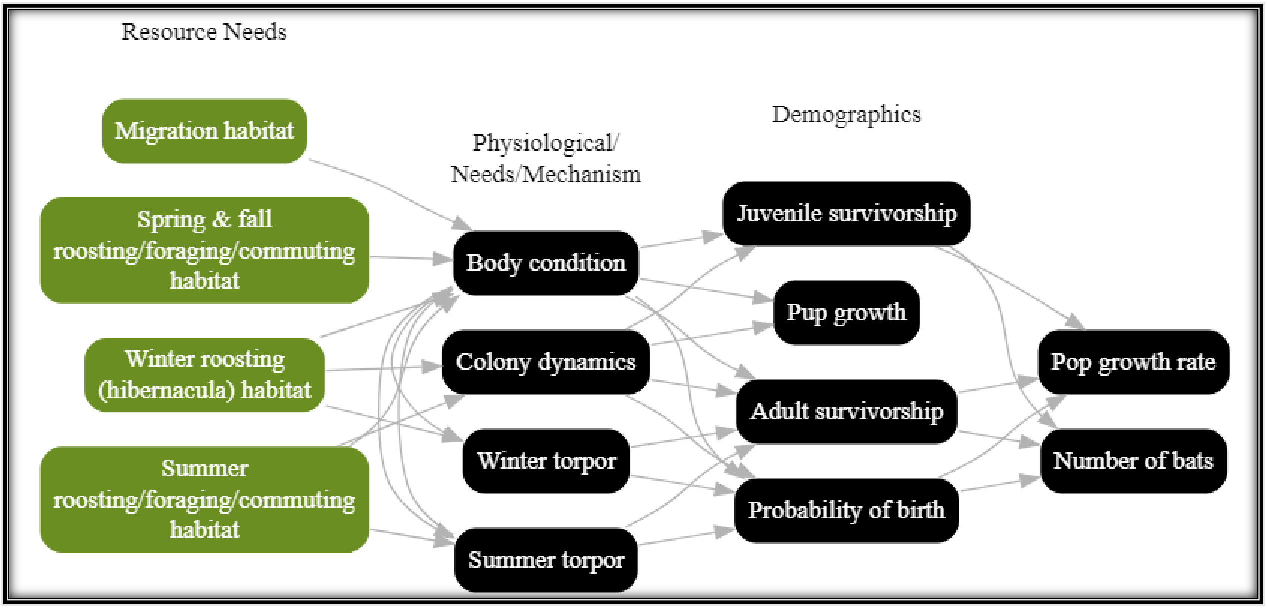

Figure 2.6. Conceptual model showing the connections between resource needs and the physiological needs and demographic rates of a TCB population (population-level resiliency).

Table 2.2. Population level requirements for a healthy population.

Parameter	**Requirements**
Population growth rate, λ	At a minimum, λ must be $\geq$1 for a population to remain stable over time.
Population size, *N*	Sufficiently large *N* to allow for essential colony dynamics and to be resilient to environmental fluctuations.
Winter roosting habitat	Safe and stable winter roosting sites with suitable microclimates.
Migration habitat	Safe space to migrate between spring/fall habitat and winter roost sites.
Spring and fall roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support bats as they exit hibernation (lowest body condition) or as they enter into hibernation (need to put on body fat).
Summer roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support maternity colonies.

Species-level Needs

The ecological requisites at the species level include having a sufficient number and distribution of healthy populations to ensure TCB can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). We describe TCB's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

TCB's ability to withstand stochastic events requires maintaining healthy populations across spatially heterogeneous conditions. Healthy populations—demographically, genetically, or physically robust—are better able to withstand and recover from environmental and demographic variability and stochastic perturbations. The greater the number of healthy populations, the more likely TCB will withstand perturbations and natural variation, and hence, have greater resiliency. Additionally, occupying a diversity of environmental conditions and being widely distributed helps guard against populations fluctuating in synchrony (i.e., being exposed to adverse conditions concurrently). Asynchronous dynamics among populations minimizes the chances of concurrent losses, and thus, provides species' resiliency. Lastly, maintaining the natural patterns and levels of connectivity between populations also contributes to TCB's resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

TCB's ability to withstand catastrophic events requires having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events. In addition to guarding against population extirpation, redundancy is important to protect against losses in TCB's adaptive capacity. Multiple, widely distributed populations within areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events, such as extreme winter events, epizootics, and hurricanes.

Representation

TCB's ability to withstand ongoing and future novel changes is influenced by its capacity to adapt (referred to as adaptive capacity). TCB may adapt to novel changes by either moving to new, suitable environments or by altering (via plasticity or genetic change) its physical or behavioral traits to match the new environmental conditions. There are multiple intrinsic factors that limit the species ability to adapt to a rapidly changing environment (see Appendix 2B). Below we describe TCB's ability to colonize new areas and to alter its physical traits.

TCB's capacity to colonize new areas (or track suitable conditions) is a function of its physical capability and behavioral tendencies to disperse. TCB flight capabilities and behavior allows for TCB to shift their summer locales in response to local novel changes. Also, dispersal ability will hinge on the availability of suitable summer and winter roosting habitat. TCB primarily roost in foliage during the summer (see *Individual-level Ecology and Needs*), so TCB need landscapes with forest habitat and tree corridors likely promote movement between forested patches. TCB is found in more hibernacula than any other North American cave-hibernating bat species (see

Individual-level Ecology and Needs) and are able to exploit human-made structures (e.g., road-associated culverts). As previously discussed above (see *Species Range*), in recent decades, TCB distribution has expanded westward and into the Great Lakes basin. This expansion signifies TCB's ability to disperse when increases in suitable summer habitat (e.g., forested areas) and suitable winter roosting sites (e.g., human-made structures) are available. Maintaining suitable habitat within local home ranges and beyond likely enables TCB to shift their range to track suitable conditions. However, despite their capacity to fly long distances, females show limited capacity to make large, abrupt shifts. Their limits are likely owing to the energetic demands of migration and reproduction at a point when their fat reserves are at their lowest after hibernation and strong philopatry (i.e., tending to return to or remain near a particular area) to both winter and summer locales. Thus, TCB adapt to changing conditions via small, local shifts but are unlikely to possess the capacity for rapid, large shifts in response to broad-scale novel changes.

TCB's capacity to alter its physical or behavioral traits (phenotypes) to match the new environmental conditions is driven by the breadth of adaptive genetic variation. Thus, maintaining populations across the breadth of variation preserves TCB's capacity to adapt to ongoing and future changes.

In addition to preserving the breadth of variation, it is also necessary to maintain the key evolutionary processes through which adaptation occurs, namely, natural selection, gene flow, and genetic drift. Maintaining healthy TCB populations across a diversity of environments and climatic conditions as well as keeping natural networks of genetic connections between populations allows for such adaptation, via natural selection or gene flow; and preserving large effective population abundances, ensures genetic drift does not act unduly upon the species (see Chapter 1 for further explanation).

For reasons explained in Chapter 1, we rely on proxies to identify species' adaptive genetic variation. We identified and delineated the genetic variation across TCB's range into geographical representation units using the following proxies: variation in biological traits, neutral genetic diversity, peripheral populations, habitat niche diversity, and steep environmental gradients. These representation units (RPUs) are described below and illustrated in Figure 2.7. Bailey's Eco-Divisions (Bailey 2016, entire) were overlayed on these proxies to identify approximate boundaries due to the associated climatic differences (i.e., precipitation levels, patterns and temperatures) that may be influential in driving the species' adaptive ability. By establishing these RPUs (a combination of proxies and Bailey's Eco-Divisions) the underlying adaptive variation of TCB (at a broad scale) is preserved.

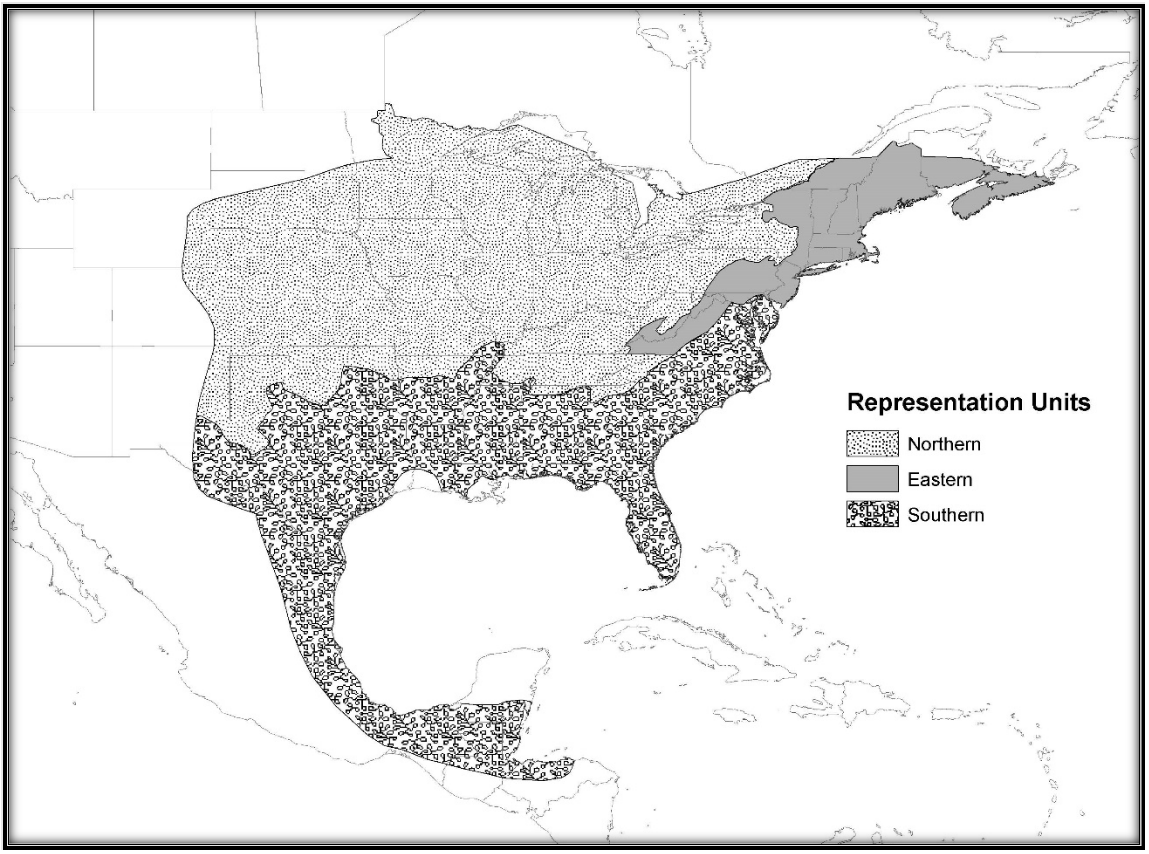

Figure 2.7. Range of TCB organized into three Representation Units.

1. Eastern RPU:

   Eastern TCB are identified as a RPU because they contribute unique genetic variation and behavioral traits and occupy unique ecological conditions.

   Eastern TCB have unique mitochondrial genetic variability that differentiates this population from other populations (Martin 2014, entire). The Appalachian Mountain range separates the Eastern and Northern RPUs and may serve as a barrier (although permeable) to maternal gene flow (Martin 2014, p. 26; Miller-Butterworth 2014, p. 361). The more robust investigation of little brown bats supports this concept. Miller-Butterworth et al. (2014, entire) found that the Allegheny front (i.e., escarpment along the eastern edge of the Appalachian plateau) likely influences movements of female little brown bats in Pennsylvania and West Virginia. They found that mitochondrial DNA was significantly different in hibernating little brown bats at three of five hibernacula west of the Allegheny front versus those hibernating east of the front and concluded that topography plays an important role in limiting female movements and maternal gene flow (Miller-Butterworth 2014, p. 361). In addition, to determine whether this pattern of population substructure extended beyond Pennsylvania and West Virginia, Miller-Butterworth et al. (2014, p. 360) analyzed genetic samples from a hibernating little brown bat colony in Vermont. They found that despite the geographic distance (up to 840 km (522 miles)), mitochondrial DNA was not significantly different from the bat colony in Vermont and colonies east of the Allegheny front in Pennsylvania and West Virginia.

The pattern of WNS (see Chapter 4) spread also supports the observed maternal gene flow patterns reported in Martin (2014, entire) and Miller-Butterworth et al. (2014, entire). The pattern and timing of spread of WNS may be partially explained by female latitudinal movements that are unobstructed by landscape features (Miller-Butterworth et al. 2014, p. 362). The initial spread of WNS between 2007 and 2009 followed the Appalachian Mountains through Pennsylvania, Maryland, Virginia, and West Virginia (Figure 4.3). All the Pennsylvania hibernacula infected with WNS during or prior to 2009 were located to the east of the Allegheny front (see Figure 1 in Miller-Butterworth et al. 2014, p. 356). Consequently, it took another 1 to 2 years for WNS to spread to hibernacula on the western side of the Allegheny front (Miller-Butterworth et al. 2014, p. 355).

The southern reach of the Eastern RPU is predominantly marked by hot summers, cool winters, and deciduous forests. The northern reach is predominantly marked by warm summers, cold winters, and coniferous forests. During the summer, Eastern TCB predominantly roost in foliage of live or recently dead deciduous hardwood trees; however, TCB in Nova Scotia are unique in their exclusive selection of *Usnea trichodea* lichen as summer roosting habitat (Poissant et al. 2010, p. 374).

2. Northern RPU:
   Northern TCB are identified as a RPU because they contribute unique genetic variation and behavioral traits and occupy unique ecological conditions.

   Northern TCB have unique mitochondrial genetic variability that differentiates this population from other populations (Martin 2014, entire; see *Eastern Unit* above for further discussion). The Northern RPU is predominantly marked by hot summers, cool or cold winters, deciduous forests to the east, prairies to the west, and coniferous forests to the north. Cooler winters have led Northern TCB to exhibit longer hibernation periods. Northern TCB generally emerge from hibernation between April and May, compared to Southern TCB who emerge from hibernation as early as March (USFWS unpublished data).

3. Southern RPU:
   Southern TCB are identified as a RPU because they contribute unique behavioral traits that include shorter hibernation duration, increased winter activity, and exploitation of road-associated culverts as hibernacula.

   Southern TCB exhibit shorter hibernation lengths and some remain active and feed year round (Grider et al. 2016, p. 8; Newman 2020, pp. 13–17). The Southern RPU is predominantly marked by subtropical climate conditions, high humidity (especially in summer), and the absence of harsh cold winters. Southern TCB may benefit from reduced physiological pressures associated with maintaining torpor during long harsh winters and in turn have higher survival rate (Fraser et al. 2012, p. 6). Southern TCB are also unique in their frequent exploitation of road-associated culverts as winter hibernacula in the southern U.S. As discussed in *Individual-level Ecology and Needs*, culverts account for the majority of hibernacula documented in Mississippi, Georgia, and Louisiana (Limon et al. 2018, entire; NABat 2021). Researchers have hypothesized that utilizing culverts coupled with sub-tropical climate conditions will lead to TCB exhibiting frequent arousal and foraging events during winter (Castleberry et al. 2019, p. 2). If TCB utilizing culverts are exhibiting increased winter activity related to foraging or otherwise, these euthermic bouts could significantly reduce their susceptibility to WNS (Cornelison et al. 2019, p. 3).

During the summer, Southern TCB predominantly roost in foliage of live or recently dead deciduous hardwood trees (see *Individual-level Ecology and Needs*); however, TCB will also roost in Spanish moss (Davis and Mumford 1962, p. 395). Note, TCB are considered rare and local in southeast Mexico and Central America (Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10); consequently, given limited data from this region, we were unable to include Guatemala, Honduras, Belize, Nicaragua, and Mexico in our analysis.

Table 2.3. Species-level ecology: Requisites for long-term viability (ability to maintain self-sustaining populations over a biologically meaningful timeframe).

3 Rs	**Requisites Long-term Viability**	**Description**
Resiliency (populations able to withstand stochastic events)	Demographic, physically, and genetically healthy populations across a diversity of environmental conditions	Self-sustaining populations are demographically, genetically, and physiologically robust, have sufficient quantity of suitable habitat
Redundancy (number & distribution of populations to withstand catastrophic events)	Multiple and sufficient distribution of populations within areas of unique variation, i.e., Representation units	Sufficient number and distribution to guard against population losses and losses in species adaptive diversity, i.e., reduce covariance among populations; spread out geographically but also ecologically
Representation (genetic & ecological diversity to maintain adaptive potential)	Maintain adaptive diversity of the species; Maintain evolutionary processes	Populations maintained across breadth of behavioral, physiological, ecological, and environmental diversity; Maintain evolutionary drivers--gene flow, natural selection--to mimic historical patterns

CHAPTER 3 – HISTORICAL CONDITION

This chapter describes the number, health, and distribution of TCB populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in TCB viability over time (Figure 3.1).

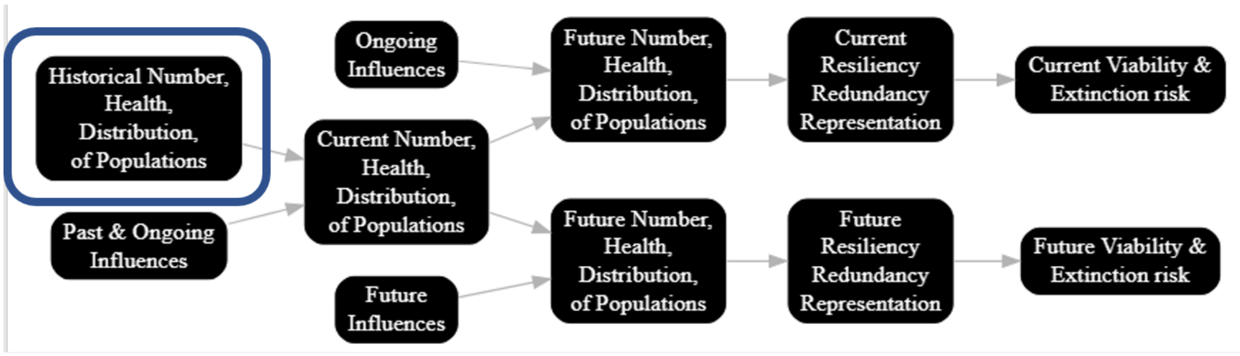

Figure 3.1. Highlighting (blue rectangle) the current step in our analytical framework.

Prior to 2006 (i.e., before WNS was first documented; see Chapter 4), TCB was highly abundant and widespread, with over 140,000 bats[4] observed hibernating in 1,951 known hibernacula spread across >1 billion acres in 34 states and 1 Canadian province (Figure 3.2, Table A-3A1). TCB numbers vary temporally and spatially, but abundance and occurrence on the landscape were generally stable (Cheng et al. 2022, pp. 204–205; Wiens et al. 2022, pp. 231–233). Although the majority of winter colony sizes were small (<100 individuals), the vast majority of individuals included in our dataset occupied a small subset of hibernacula; for example, in 2000, 32% (n=508) of the known winter colonies contained 90% of total known winter abundance (Figure 3.2).

Historically, of the known hibernacula, the Northern RPU contained approximately 58% of winter hibernacula (n=1,124) and 66% of the total TCB abundance. The Southern RPU contained approximately 32% of winter hibernacula (n=616) and 22% of the total abundance and the Eastern RPU was the smallest, comprising approximately 11% of winter hibernacula (n=211) and 11% of the total abundance (Table A-3A2).

TCB's range encompasses 39 states, 4 Canadian provinces, and Guatemala, Honduras, Belize, Nicaragua, and Mexico (Figure 2.3). In this SSA, we include occurrence records (i.e., TCB acoustic calls, captures, and hibernacula records) from 38 States, the District of Columbia, and 2 Canadian Provinces (Figure 3.3).

[4] This number only represents TCB that were observed during internal winter hibernacula surveys submitted to NABat for use in this SSA; we acknowledge historical TCB abundance and the number of hibernacula were higher.

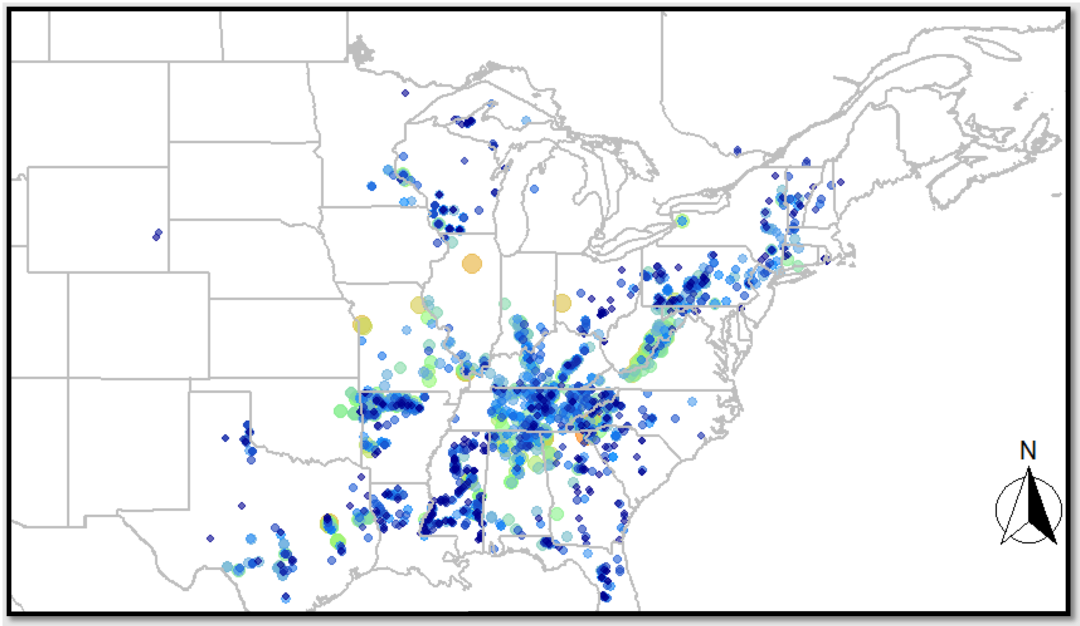

Figure 3.2. All known hibernacula and winter abundances for TCB in 2000. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

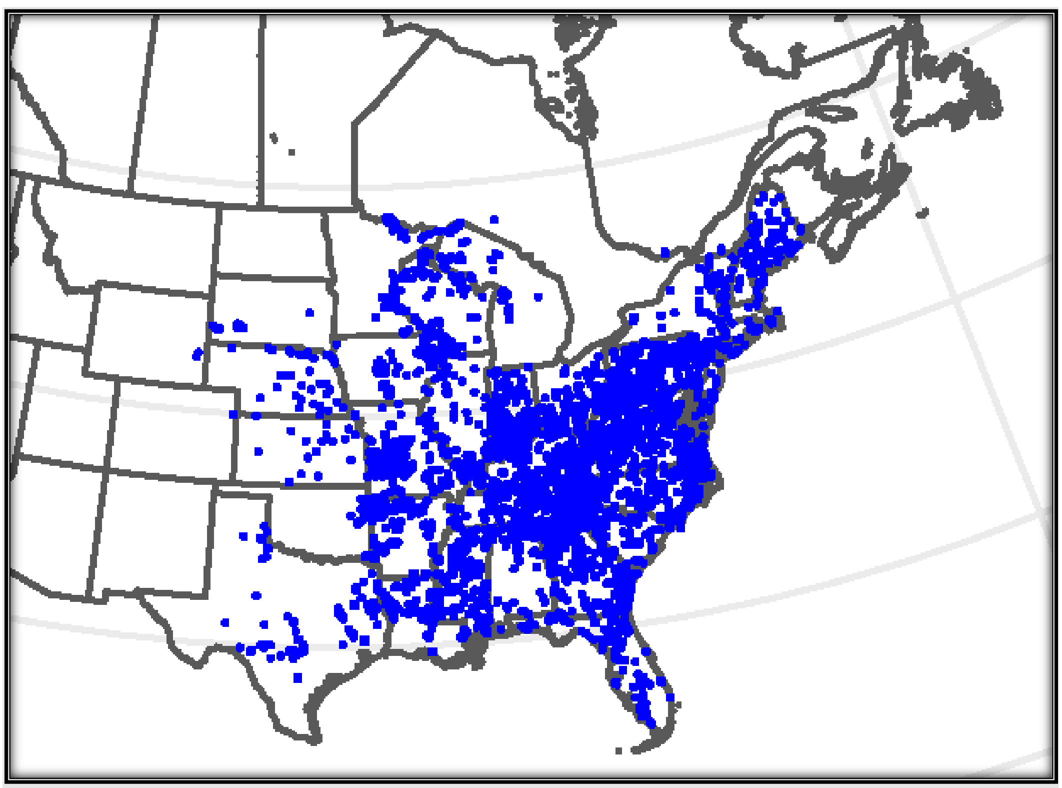

Figure 3.3. Documented range of TCB (blue dots), as known from available records (acoustic calls, captures, and hibernacula records) in the U.S. and Canada (Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center. Disclaimer: Provisional information is subject to revision). This map shows data provided to the SSA for TCB and does not replace the species range (Figure. 2.3).

CHAPTER 4 – PRIMARY INFLUENCES ON VIABILITY

Recognizing there are myriad influences operating on TCB, this chapter describes the primary threats that have most likely led to its current condition: WNS, wind related mortality, effects from climate change, and habitat loss (Figures 4.1 and 4.2). We similarly describe the primary past and ongoing conservation efforts that may be ameliorating these threats. Lastly, for WNS and wind related mortality, we describe the plausible future condition for each threat. To capture the uncertainty in our future projections, we identified the lowest plausible and highest plausible state for each primary threat. These lower and upper impact states for each threat were then combined to create composite plausible "low impact" and "high impact" scenarios. For climate change and habitat loss we provide qualitative assessments.

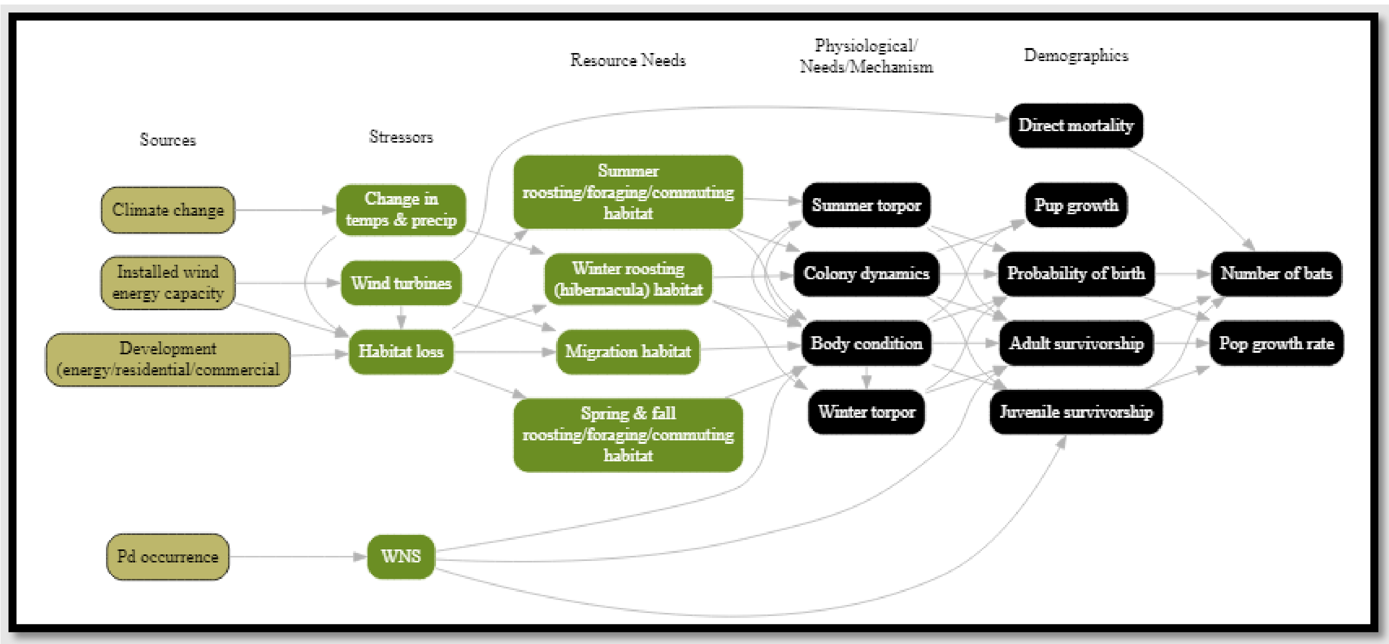

Figure 4.1. Visual diagram showing relationships between the primary threats and population needs.

Threats

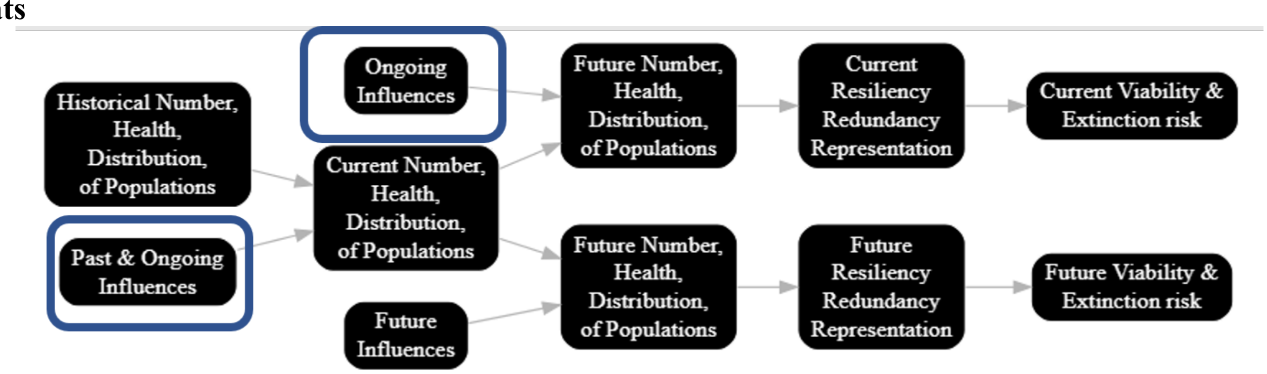

Figure 4.2. Highlighting (blue rectangle) the current step in our analytical framework.

White-nose Syndrome

For over a decade, WNS has been the foremost stressor on TCB. WNS is a disease of bats that is caused by the fungal pathogen *Pd* (Blehert et al. 2009, entire; Turner and Reeder 2009, entire; Lorch et al, 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2017, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first discovered in eastern New York in 2007 (with photographs showing presence since 2006) (Meteyer et al. 2009, p. 411), and since then have spread to 39 states and 7 provinces in North America (Figure 4.3). *Pd* invades the skin of bats, initiating a cascade of physiological and behavioral processes that often lead to mortality (Warnecke et al, 2013, p. 3; Verant et al. 2014, pp. 3–6). Infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves needed to survive winter (Turner et al. 2011, p. 15; Reeder et al. 2012, p. 5; Carr et al. 2014, p. 21; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). Bats that do not succumb to starvation in hibernacula often seek riskier roosting locations near entrances to roosts or emerge from roosts altogether, where they face exposure to winter conditions and scarce prey resources on the landscape (Langwig et al. 2012, p. 2). The weeks following emergence from hibernation also mark a critical period because prey availability is still limited, energetic costs of healing from WNS are high, and the potential for immune reconstitution inflammatory syndrome that can lead directly to mortality or impact reproductive success (Reichard and Kunz 2009, p. 461; Francl et al. 2012, pp. 35–36; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Reynolds et al. 2016, pp. 199–200; Fuller et al. 2020, pp. 7–8). As of May 2021, WNS has been confirmed in 12 species in North America, including TCB, and numerous other species in Europe and Asia (www.whitenosesyndrome.org, accessed online May 13, 2021; Hoyt et al. 2021, Suppl. material).

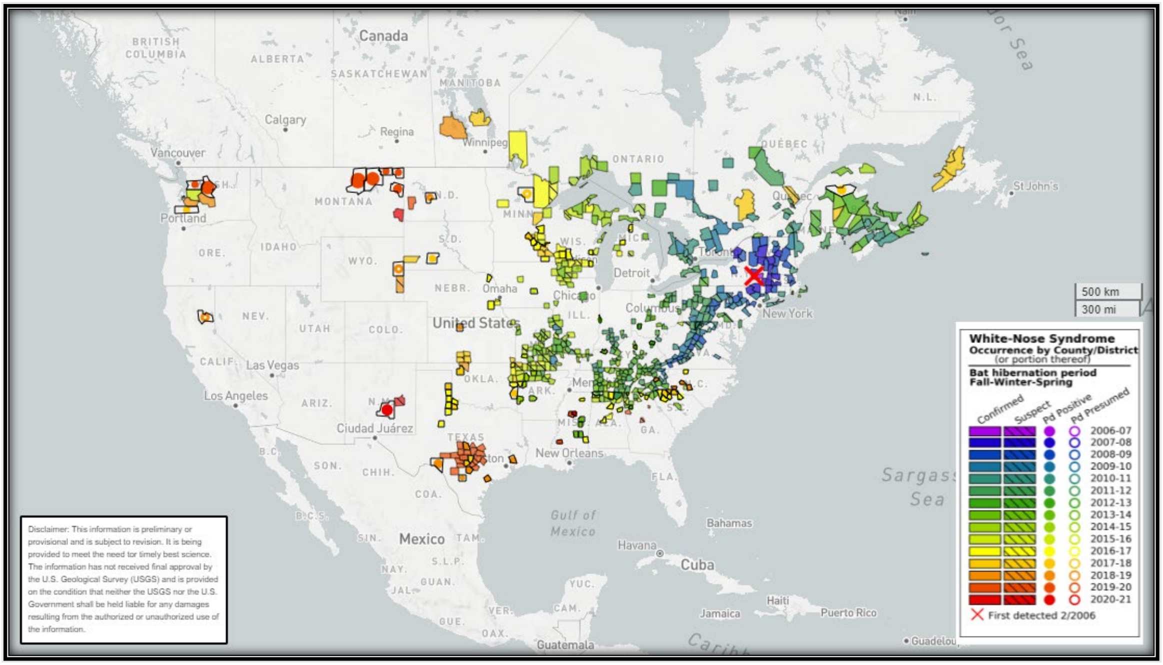

Figure 4.3. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles). Pd and WNS occurrence records generally reflect locations of winter roosts and are not representative of the summer distribution of affected bats (www.whitenosesyndrome.org, accessed online: May 13, 2021).

The fungal pathogen is spread primarily via bat-bat and bat-environment-bat movement and interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). With the arrival of *Pd* (year 0) to a new location, WNS progresses through "stages" similarly to many emerging infectious diseases: pre-invasion, invasion, epidemic, and establishment (Langwig et al. 2015a, p. 196; Cheng et al. 2021, p. 5). During *invasion* (years 0–1), the fungus arrives on a few bats and spreads through the colony as a result of swarming and roosting interactions until most individuals are exposed to the pathogen. Such interactions may occur in hibernacula or at nearby roosts where conspecifics engage in mating activity (Neubaum and Siemers, 2021, p. 2). As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7), and 8 years after its arrival, *Pd* is determined to be *endemic* in a population (Langwig et al. 2015a, p. 196; Cheng et al. 2021, p. 5).

The effect of WNS on TCB has been extreme, such that most summer and winter colonies experienced severe declines following the arrival of WNS. Just 4 years after the discovery of WNS, for example, Turner et al. (2011, pp. 18–19) estimated that TCB experienced a 75% decline in winter counts across 42 sites in Vermont, New York and Pennsylvania. Similarly, Frick et al. (2015, p. 5) estimated the arrival of WNS led to a 10–fold decrease in TCB colony size. Most recently, Cheng et al. (2021, p. 7) used data from 27 states and 2 provinces to conclude WNS caused estimated population declines of 90–100% across 59% of TCB range. Although variation exists among sites, the arrival of *Pd* caused marked decreases in populations during invasion, epidemic, and established stages of the disease (Figure 4.4), and lambda estimates less than 1 after the arrival of *Pd*, with few exceptions (Figure 4.5).

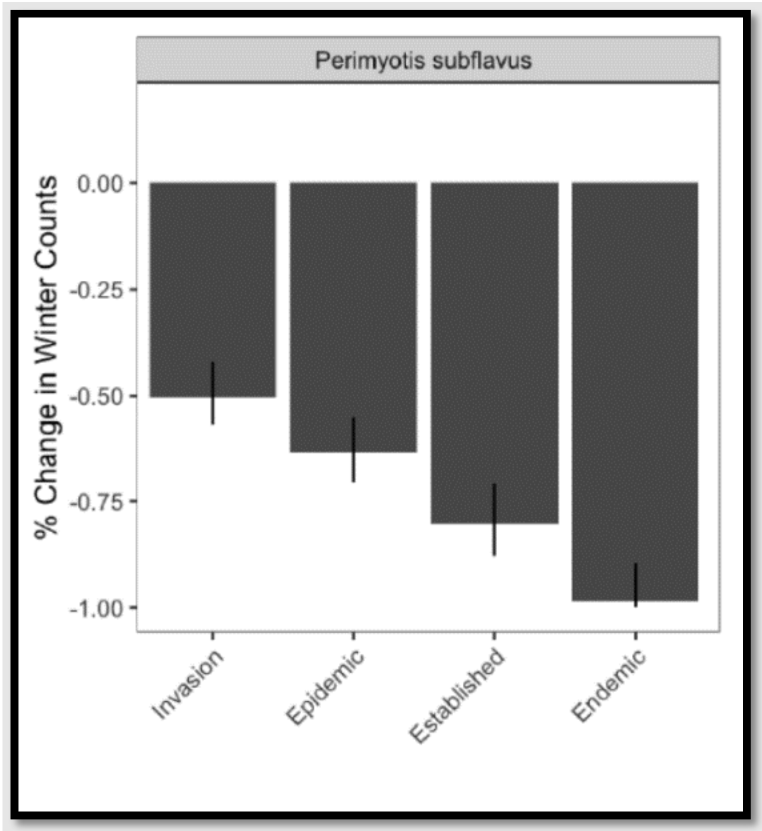

Figure 4.4. Percent change in TCB winter colony counts by disease stage relative to predicted median count prior to arrival of Pd (with 95% credible interval) (Cheng et al. 2022, Fig. D4).

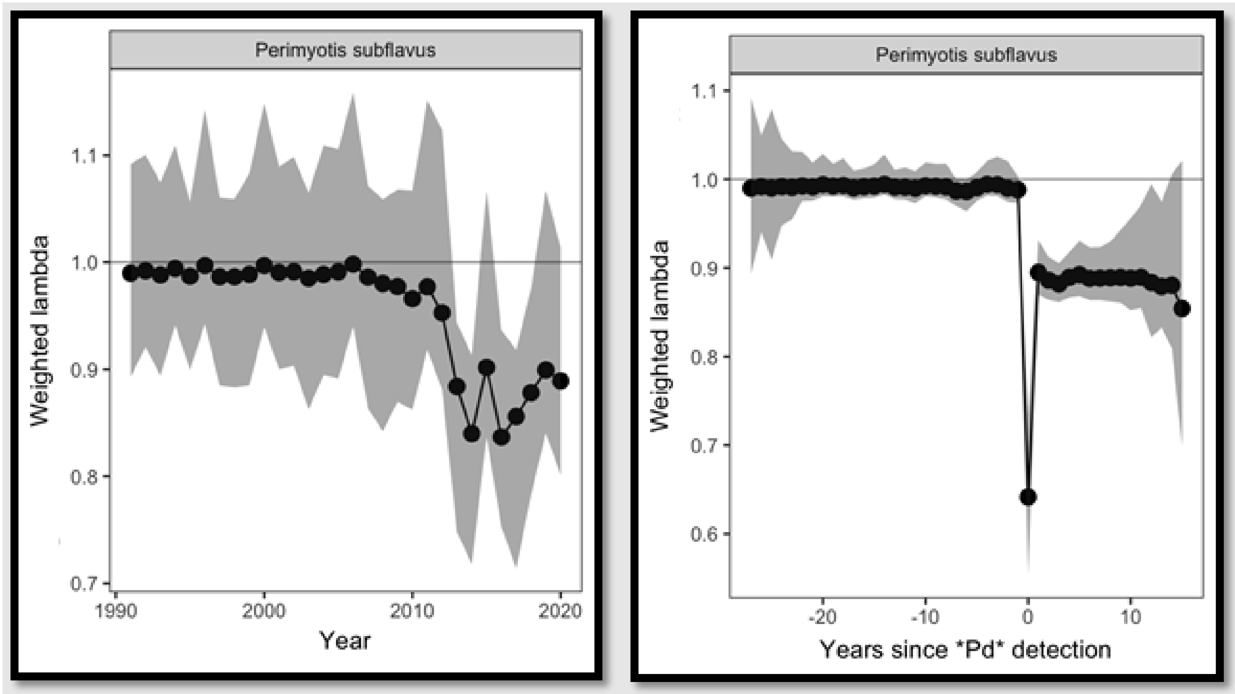

Figure 4.5. Estimated TCB weighted lambda (function of growth rate and colony size) by year (left) and by year since arrival of Pd (right) (Cheng et al. 2022, Fig. D3).

Building off work of Cheng et al. (2022, entire), Wiens et al. (2022, entire used available data from hibernacula surveys to estimate the annual impacts of WNS relative to the year of arrival of *Pd*, adding additional analysis of an endemic stage. Their analysis applied two models of *Pd* occurrence to interpolate WNS occurrence to all documented hibernacula. The analysis predicted *Pd* is present at 85–100% of documented TCB hibernacula (see Table A-5.2 for current WNS stage by hibernacula). Although variation exists among sites, an overwhelming majority of hibernating colonies of TCB have developed WNS and experienced serious impacts within 2–3 years after the arrival of *Pd* (Cheng et al. 2021, p. 8; Wiens et al. 2022, pp. 231–247) (Figures 4.4 and 4.6).

With respect to road-associated culverts used as hibernacula in the southern U.S., there is uncertainty associated with progression of WNS within these TCB winter colonies (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276). For example, *Pd* has been detected in several culverts that house overwintering TCB in Mississippi. Although *Pd* was first detected at these sites in 2014, no disease, mortality, or population impact has been documented (Cross 2019, entire). A variety of environmental and biological factors may contribute to the differences observed in culverts. Year-round temperature profiles may affect the environmental reservoir of *Pd*, thus reducing the source of reinfection when bats return to the locations each fall, which would be more likely to delay than preclude infection (Hoyt et al. 2020, pp. 7257–7258). However, it is important to acknowledge that bats likely encounter multiple subterranean environments during swarming activity, during which they can encounter reservoirs of *Pd* (Neubaum and Siemers, 2021, pp. 3–4). Winter length and climate may also affect the behavior and physiology of hibernating bats using culverts (e.g., shorter torpor bouts) or offer foraging opportunities that make it possible for them to avoid more serious infections, but these mechanisms have not been tested (Hayman et al. 2016, p. 5). Regardless, the vast majority of TCB colonies exposed to *Pd* have developed and are expected to continue to develop WNS and experience impacts from the disease (Cheng et al. 2021, Appendix S3; Wiens et al. 2022, pp. 231–247) (Figure 4.6).

Caves and cave-like hibernacula in this region do not appear to have the same uncertainty as culverts, although winter length and foraging opportunities may be similar to those experienced by colonies in culverts. Where *Pd* has been detected in caves and tunnels in the Southern RPU, these colonies have exhibited declines more in line with those documented farther north. Black Diamond Tunnel in northern Georgia declined from a high count of over 5,000 TCB in 2013 to about 200 TCB 3 years later, after WNS was confirmed there. Carleton Cave in Alabama had a max count of 1,794 TCB in 2013 and declined to 54 in 2018. There is also evidence of declines that are not associated with known arrival of *Pd.* For example, in Florida, Smith et al. (2021, p. 21) found TCB declined 73.9% at caves between 2015 and 2020, even though *Pd* has not been detected. Whether these losses represent bats contracting WNS at swarming sites and dying elsewhere is unknown, but current evidence does not support that being the case and consequently, the reason behind these declines is currently unknown. It is also plausible that changes in these colonies are the result of bats relocating to other hibernacula, many of which may not be counted.

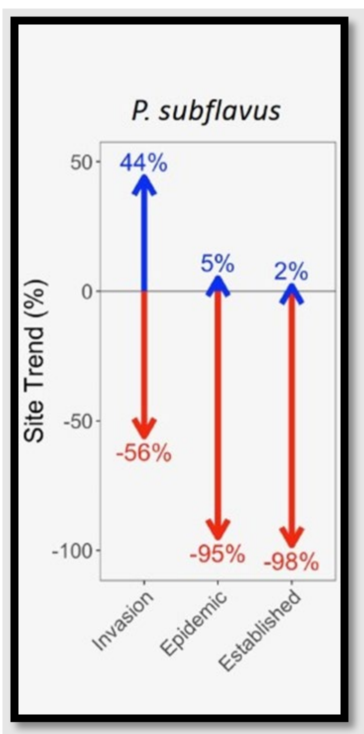

Figure 4.6. Percentage of winter colonies with increasing (blue) and decreasing (red), colony trend relative to WNS pre-arrival stage for invasion, epidemic, and established stages (Cheng et al. 2021, Appendix S3).

There are multiple national and international initiatives underway in an attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts. See Appendix 4A for more information regarding WNS impacts.

Wind Related Mortality

Wind related mortality, overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and RPU levels. Wind power is a rapidly growing portion of North America's energy portfolio in part due to changes in State energy goals (NCSL 2021, web) and recent technological advancements (Berkeley Lab 2020, web) and declining costs (Wiser et al. 2021, entire), allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association (AWEA) 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often encompass tens or hundreds of turbines, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[5] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat fatalities continue to be documented at wind power installations across North America and Europe. We describe mechanisms leading to bat fatalities in Appendix 4B.

Bat fatality varies across facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise the majority of all known bat fatalities at wind facilities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other non-listed bat species. However, there is notable spatial overlap between TCB occurrences and wind facilities (Figure 4.7) and notable TCB mortality documented. Based on October 2020 installed MW capacity (Hoen et al. 2018, entire; USFWS unpublished data), we estimated 3,227 TCB are killed annually at wind facilities (Table 4.1; Figure A-2A6; Udell et al. 2022, pp. 265–266). Analyses using data from Wiens et al. (2022, pp 236–247) and analyses by Whitby et al. (2022, entire) suggest that the impact of wind related mortality is discernible in the ongoing decline of TCB. Based on data from Wiens et al. (2022, pp. 236–247) comparing a no wind baseline scenario to current and future wind scenarios, the projected abundance decreases 19–21% by 2030 under the current wind scenario and up to 38% by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, pp. 151–153) found a decline in the predicted relative abundance of TCB as wind energy risk index increased. To reduce bat fatalities, some facilities "feather" turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein and Straw 2021, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed or at a higher threshold, typically referred to as curtailment. The

[5] Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, pp. 32 and 69).

effectiveness of feathering below various cut-in speeds (i.e., when turbine blades start rotating and generating power) differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with feathering below wind speeds of 4.0–6.5 meters per second (m/s) (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for TCB, however, has not been documented. Hereafter, we refer to feathering below the manufacturer's cut-in speed or higher wind speeds collectively as curtailment.

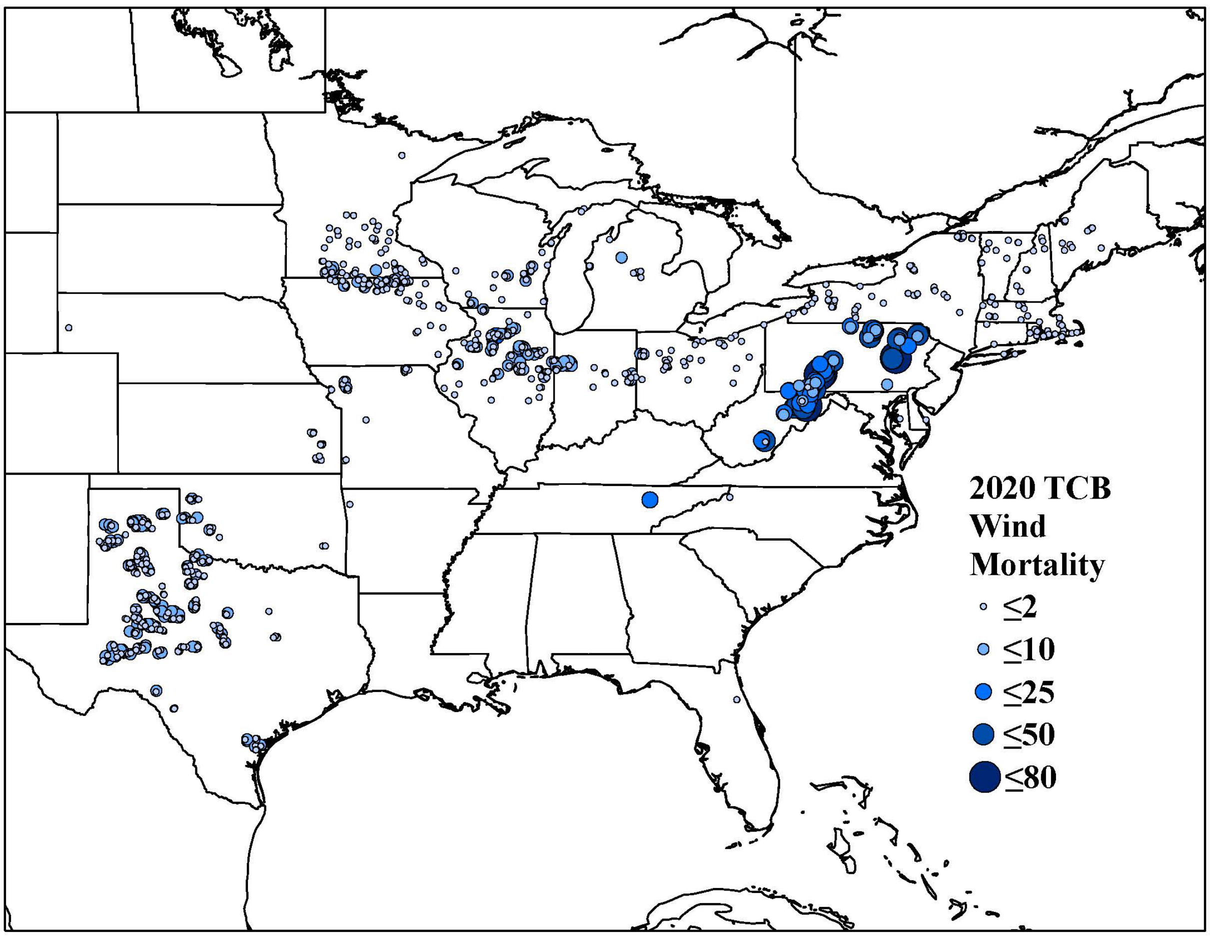

Figure 4.7. Estimated total annual TCB mortality at wind facilities in 2020. Mortality is shown at U.S. wind turbines as summed by 11x11-km NREL grid cell within the migratory range of extant NLEB hibernacula. Note that because MW were summed by Province centroid in Canada (and none were within the migratory range of hibernacula), the only TCB mortality that was allocated to Canadian hibernacula (Quebec) was that occurring at U.S. turbines within the migratory range. See Udell et al. 2022 pp. 265–266 and Appendix 2 for details on the wind mortality analysis.

Table 4.1. Estimated annual TCB mortality from wind facilities by USFWS Region (Figure A-2A6) and Canada, based on installed MW capacity in October 2020 (Udell et al. 2022, pp. 265–266).

Location	Mean Annual Mortality (n)	Lower CI	Upper CI
Region 2	912	301	929
Region 3	936	302	1,139
Region 4	159	48	197
Region 5	1,186	357	1,476
Region 6	30	12	31
Quebec	4	1	6
Total	**3,227**	**1,021**	**3,778**

There are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. To date, operational strategies (e.g., feathering turbine blades when bats are most likely to be active) are the only broadly proven and accepted measures to reduce the severity of impacts. See Appendix 4B for more information.

Climate Change

There is growing concern about impacts to bat populations in response to climate change (Jones et al. 2009, entire; Jones and Rebelo 2013, entire; O'Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats, including changes in hibernation, mortality from extreme drought, cold, or excessive rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed and discussed potential impacts of climate change, including effects to bat foraging, roosting, reproduction, and biogeography. Climate change is also likely to influence disease dynamics as temperature, humidity, phenology and other factors affect the interactions between *Pd* and hibernating bats (Hayman et al. 2016, p. 5; McClure et al. 2020, p. 2; Hoyt et al. 2021, p. 8). However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). Climate change may impact these bats in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid (*Adelges tsugae*), southern pine beetle (*Dendroctonus frontalis*)) (Haavik 2019).

While there are a number of changing climatic variables, our analysis focused solely on changes in temperature and precipitation. These variables influence TCB's resource needs, such as suitable roosting habitat (all seasons), foraging habitat, and prey availability (Figure 4.1). Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., average annual temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986–2016 relative to 1901–1960 (Hayhoe et al. 2018, p. 86). Temperatures increased during that time at a regional scale as well, with the largest changes (average increases of more than 1.5 degrees F (0.8 degrees C) in Alaska, the Northwest, the Southwest and the Northern Great Plains and the least change in the Southeast (increase of 0.46 degrees F (0.26 degrees C) (Vose et al. 2017, pp. 186–187; Hayhoe et al. 2018, p. 86). Average annual precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the West, Southwest and Southeast (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events across the U.S. have increased more than the increases in average precipitation (Hayhoe et al. 2018, p. 88).

TCB risk of exposure to changes in the climate is rangewide. However, the magnitude, direction, and seasonality of climate variable changes is not consistent rangewide. In addition, the resiliency of populations and inherent differences (e.g., genetics) among populations may result in differing ability for TCB to respond to the same types of changes across the range. Therefore, the overall impact of climate change for such a wide-ranging species is challenging to describe. Although there may be some benefit to TCB from a changing climate, overall negative impacts are anticipated. Although we lack species-specific observations for TCB, observed impacts to date for other insectivorous bats, such as little brown bat, include reduced reproduction due to drought conditions leading to decreased availability of drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years (drought) (Frick et al. 2010, pp. 131–133). While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During the anticipated heavier precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795), which may be especially important for bats like TCB that roost in foliage rather than inside more protected shelters. Bats are likely to reduce their foraging bouts during heavy rain events and reduced reproduction of insectivorous bats has been observed during cooler, wetter springs (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Responses will vary throughout TCB's range based on the extent of annual temperature rise in the future. For additional information on climate change see Appendix 4C.

Habitat Loss

Roosting/Foraging/Commuting Habitat Loss

As discussed in Chapter 2, TCB require suitable habitat for roosting and foraging, and commuting between those habitats during spring, summer, and fall. Forest is a primary component of roosting, foraging, and commuting habitat. Wetlands and water features are important foraging and drinking water sources. Loss of these habitats influences survival and reproduction of TCB colonies.

We reviewed changes in various NLCD landcover classes within each RPU from 2006–2016 in the continental U.S. Deciduous forest landcover decreased across all RPUs by 768,903 ha (1,900,000 ac) for an average loss of 76,890 ha (190,000 ac) per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 687,966 ha (1,700,000 ac). See Appendix 4D for additional information.

These changes in landcover may be associated with losses in suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colonies, and direct injury or mortality. While temporary or permanent habitat loss may occur throughout the species' range, impacts to TCB and its habitat typically occur at a more local-scale (i.e., individuals and potentially colonies). Impacts to TCB from loss of habitat vary depending on the timing, location, and extent of the removal. Impacts from forest habitat removal may range from minor (e.g., removal of a small portion of foraging habitat in largely forested landscapes with robust TCB populations) to significant (e.g., removal of roosting habitat in highly fragmented landscapes with small, disconnected populations). Adverse impacts are more likely in areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), as there is a higher probability of removing roosts or causing loss of connectivity between roosting and foraging habitat. There are a variety of conservation measures that can either serve to reduce effects from habitat loss or help maintain or enhance habitat. See Appendix 4D for examples.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, TCB require hibernation sites with specific microclimates and TCB exhibit high interannual fidelity to their hibernacula. Therefore, the complete loss of or modification of winter roosts (such that the site is no longer suitable) can result in impacts to individuals or at the population level. In addition, disturbance within hibernacula can render a site unsuitable or can pose harm to individuals using the site.

Modifications to bat hibernacula (e.g., erecting physical barriers to control cave and mine access, intentional or accidental filling or sealing of entries, or creation of new openings) can alter a bat's ability to access hibernacula (Spanjer and Fenton 2005, p. 1110) or can affect the airflow and alter microclimate of the subterranean habitat, and thus the ability of the cave or mine to support hibernating bats, such as TCB. These well-documented effects on cave-hibernating bat species were discussed in the USFWS's *Indiana Bat Draft Recovery Plan* (USFWS 2007, pp. 71–74). In addition to altering the thermal or humidity regime and ability of the site to support hibernating bats, bats present during any excavation or filling can be crushed or suffocated. Sources of these stressors include fill from adjacent activities, mining, and intentional closures of abandoned mines or cave openings to restrict access.

Human entry or other disturbance to hibernating bats results in additional arousals from hibernation which require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable. This is even more important for hibernacula where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Boyles and Willis 2010, p. 96).

There are many conservation efforts and protections (e.g., bat-friendly gates, closure of caves to exclude humans during hibernation, conservation easements) in place that attempt to reduce the risk of modifications to hibernacula and disturbance to overwintering bats. See Appendix 4D for more information.

Conservation Efforts

Conservation efforts associated with reducing the effects of WNS, wind related mortality, and habitat loss are mentioned above and discussed further within associated appendices. In addition to those efforts, below we highlight the regulatory protections afforded to TCB in parts of its range.

Federal, State, Provincial Protection

TCB was listed as endangered on Schedule 1 of Canada's Species at Risk Act in 2014. This provided the TCB protection from being killed, harmed, harassed, captured, or taken in Canada. Environment and Climate Change Canada finalized a recovery strategy for the little brown bat, northern long-eared bat, and TCB in 2018 (Environment and Climate Change Canada 2018, entire).

In addition, TCB receives varying degrees of protection through State laws as it is State-listed endangered in Connecticut, Indiana, Massachusetts, New Hampshire, Ohio, Pennsylvania, Vermont, and Virginia; State-threatened in Tennessee and Wisconsin; and special concern in Alabama, Georgia, Iowa, Maine, Michigan, Minnesota, Missouri, South Carolina, and West Virginia.

Synopsis of Current Threats

To provide a comparative and semi-qualitative assessment of the primary influences, we summarize the scope, severity, and impact of each of the four influences using criteria defined by Master et al. (2012, pp. 28–35; Table 4.2). Currently, WNS is the greatest threat to TCB (*High Impact*), with extreme population level declines (90–100%) over a large (59%) portion of its range (Cheng et al. 2021, p. 7). Wind energy related mortality has the next highest level of impact (*Medium Impact*), with moderate (19–21%, see Table A-3D1) population-level declines over a large (53%) portion of its range. Both habitat loss and climate change are pervasive across TCB's range, while severity of population level declines are considered slight; therefore, we assigned *Low Impact* level for both habitat loss and climate change given current state conditions. While confidence in impact to TCB from WNS and wind were "high" due to availability of quantitative data, our confidence in our impact analysis of habitat loss and climate change are "low" to "moderate" due to limited data. See Appendix 3D for additional details.

Table 4.2 Assessment of current impact to TCB from primary threats (adapted from Master et al. 2012). See Chapter 1 for definitions of the criteria (Figure 1.2).

Criteria	WNS	Wind Mortality	Habitat Loss	Climate Change
Scope	Large	Large	Pervasive	Pervasive
Severity	Extreme	Moderate	Slight	Slight
Impact	**High**	**Medium**	**Low**	**Low**
Confidence Level	High	High	Moderate	Low

Future Scenarios

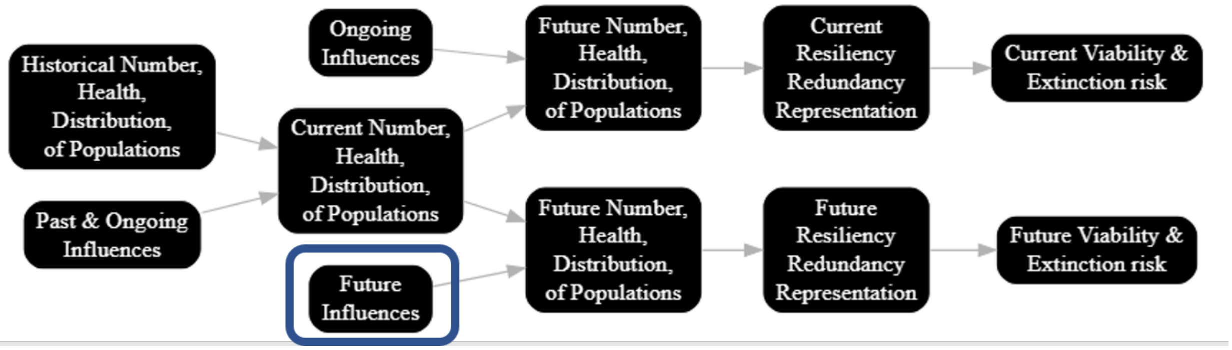

Figure 4.8. Highlighting (blue rectangle) the current step in our analytical framework.

To assess how TCB will respond to foreseeable changes in *Pd* and wind energy capacity, we identified the plausible future state of these influences (Figure 4.8). We developed realistic lower and upper bounds for both and combined them to create composite plausible "high impact" and "low impact" scenarios. The composite future scenarios for WNS and wind mortality are summarized in Table 4.3. These scenarios and their underlying rationales are described below, along with the future projected conditions for habitat loss and climate change. We provide further rationale for our low and high impact scenarios in Appendix 5

Table 4.3. TCB composite plausible future scenarios.

Plausible Scenario	*Pd* Occurrence Model	WNS Impact Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition
Low impact	*Pd* Occurrence Model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	Regional-specific
High impact	*Pd* Occurrence Model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	Regional-specific

White-nose Syndrome

To project future impacts of WNS, we relied on 1) predicted current and future occurrence of *Pd* on the landscape using two different models (hereafter, "*Pd* occurrence models") and 2) the WNS impacts schedule. For the latter, we assumed winter colonies that are exposed to *Pd* in the future will respond similarly to those currently exposed (i.e., colonies exposed in the future will follow the same WNS impacts schedule) (see Chapter 1, *Step 3. Identify the Primary Drivers (Influences)* and Appendix 5 for more detail).

To project future spread of WNS, we relied upon two *Pd* occurrence models, *Pd* occurrence model 1 (derived by Wiens et al. 2022, pp. 226–229) and *Pd* occurrence model 2 (derived by Hefley et al. 2020, entire); both models are briefly described in Appendix 2. For a low impact scenario, we used *Pd* occurrence model 1 for predicted year of arrival (YOA) and assumed that the WNS impacts schedule continues for 15 years after arrival *Pd*, after which the colonies return to pre-WNS survival rates for the remainder of the simulation (i.e., no WNS impacts applied after 15 years since *Pd* arrival). Return to pre-WNS growth rates at YOA 15 is the earliest year we can reasonably assume (given data show impacts continue occurring 14 years since the first detection in New York). For the high impact scenario, we used *Pd* occurrence model 2 for predicted YOA and assumed that WNS impacts continue through 2060 (i.e., after YOA 0 to 6, survival rates remain in the endemic phase).

Wind Related Mortality

To project future installed wind capacity, we relied upon National Renewable Energy Laboratory's (NREL; Cole et al. 2020) and Canadian Energy Regulator's (CER) (CER 2020) projections for the U.S. and Canada, respectively (Figure 4.9). Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario* (Figure 4.10). Our high impact scenario (i.e., higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Figure 4.11). For both scenarios, we calculated TCB fatalities per MW using the species composition approach (see Chapter 1 methods and Appendix 2A for additional detail). We applied the reduced species composition rate observed after *Pd* arrival. The annual mortality associated with the future low and high impact scenarios by Year 2050 is provided in Table 4.4.

We selected NREL's scenarios per consultation with the USDOE's Wind Energy Technology Office (Gilman 2020, pers. comm.). The NREL scenarios model future deployment levels based on projected trends in electricity demand, technology cost trajectories, and existing Federal and state energy policies Cole et al. 2020, p. iii; see Appendix 5 for details). NREL's 2020 (Cole et al. 2020, entire) report presents 45 power sector scenarios that consider present day through 2050. We chose the *High Wind Cost* and *Low Wind Cost* scenarios as reasonable lower and upper bounds of future wind build-out, respectively. NREL agreed that use of the *High Wind Cost* and *Low Wind Cost* scenarios provides a reasonable range of future wind build-out (Cole 2020, pers. comm.).

CER's *Canada's Energy Future* report is published annually and provides up-to-date projections for wind build-out in Canada. CER uses economic and energy models to project future scenarios "*based on assumptions about trends in technology, energy and climate policies, energy markets, human behavior and the structure of the economy*" (CER 2019, p. 1). Annual wind build-out projections are produced at the province/territory level and data are continually refined based on current trends. We chose the *Reference Scenario* as our lower-impact scenario (i.e., lower wind build-out) and the *Evolving Scenario* as our higher-impact scenario (i.e., higher wind build out; see Appendix 5 for details).

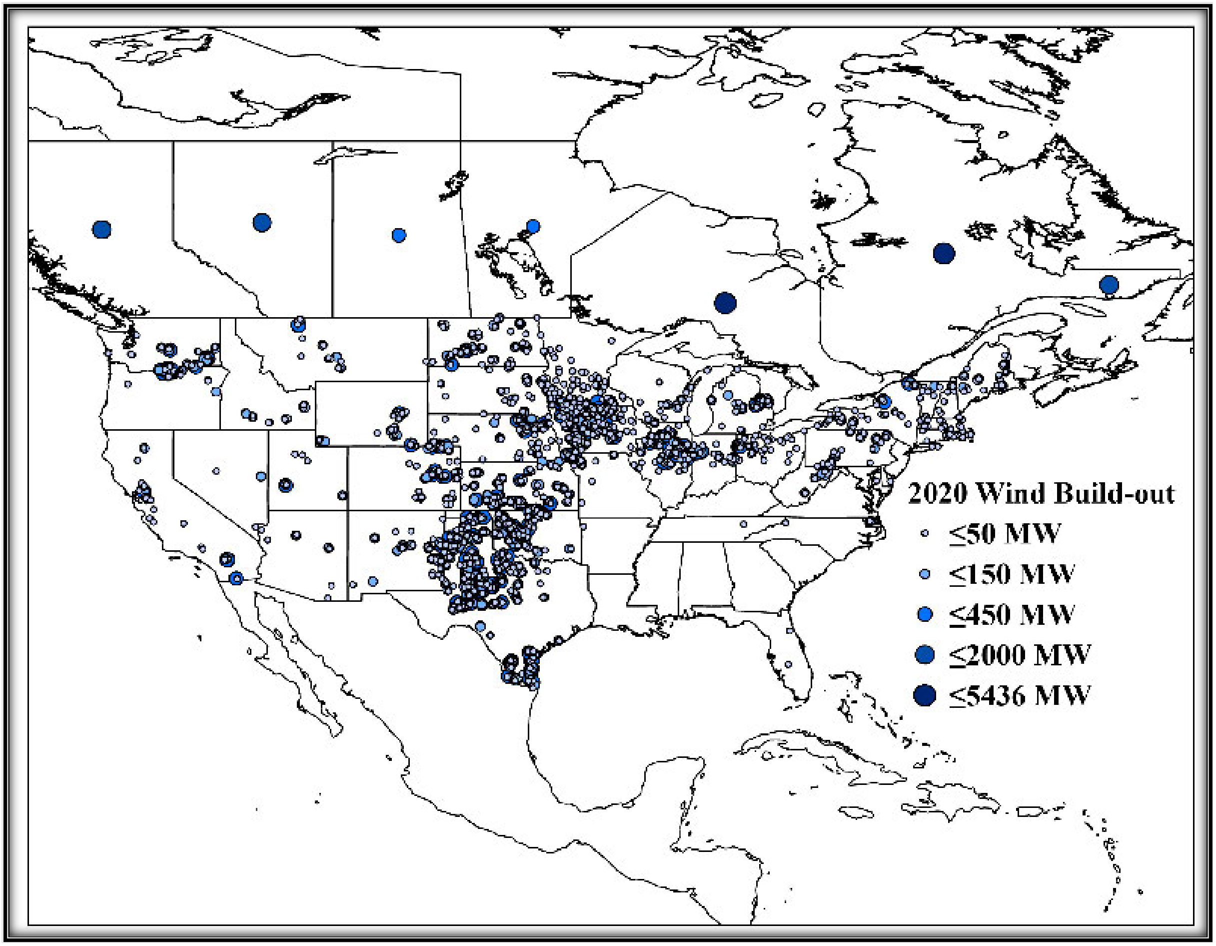

Figure 4.9. Wind build-out as of October 2020 for the U.S. and Canada (Udell et al. 2022, entire). U.S. capacity is summed by 11x11-km NREL grid cell and Canadian capacity by Province.

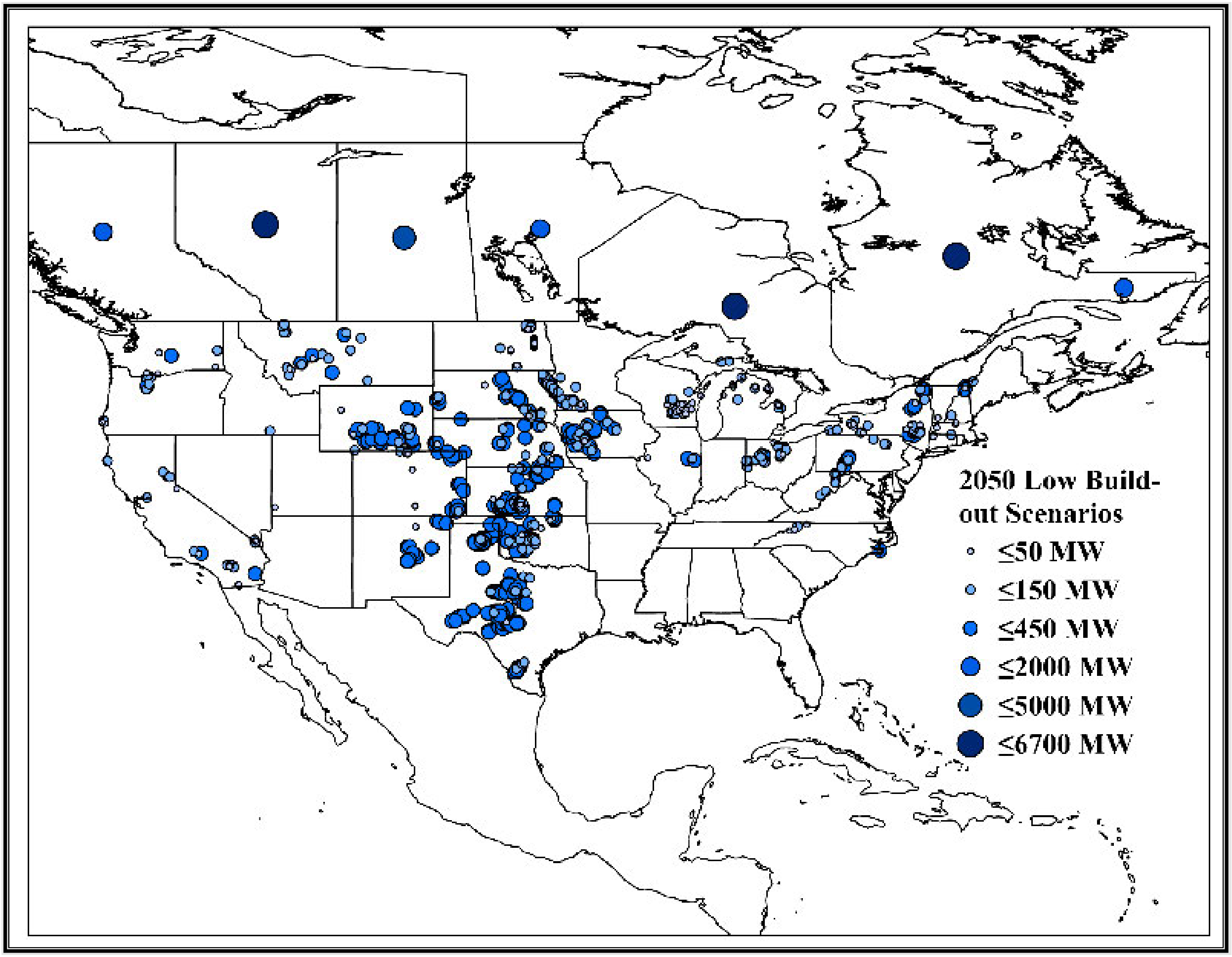

Figure 4.10. Projected wind build-out for the year 2050 per low build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km grid cell and Canadian future capacity by Province.

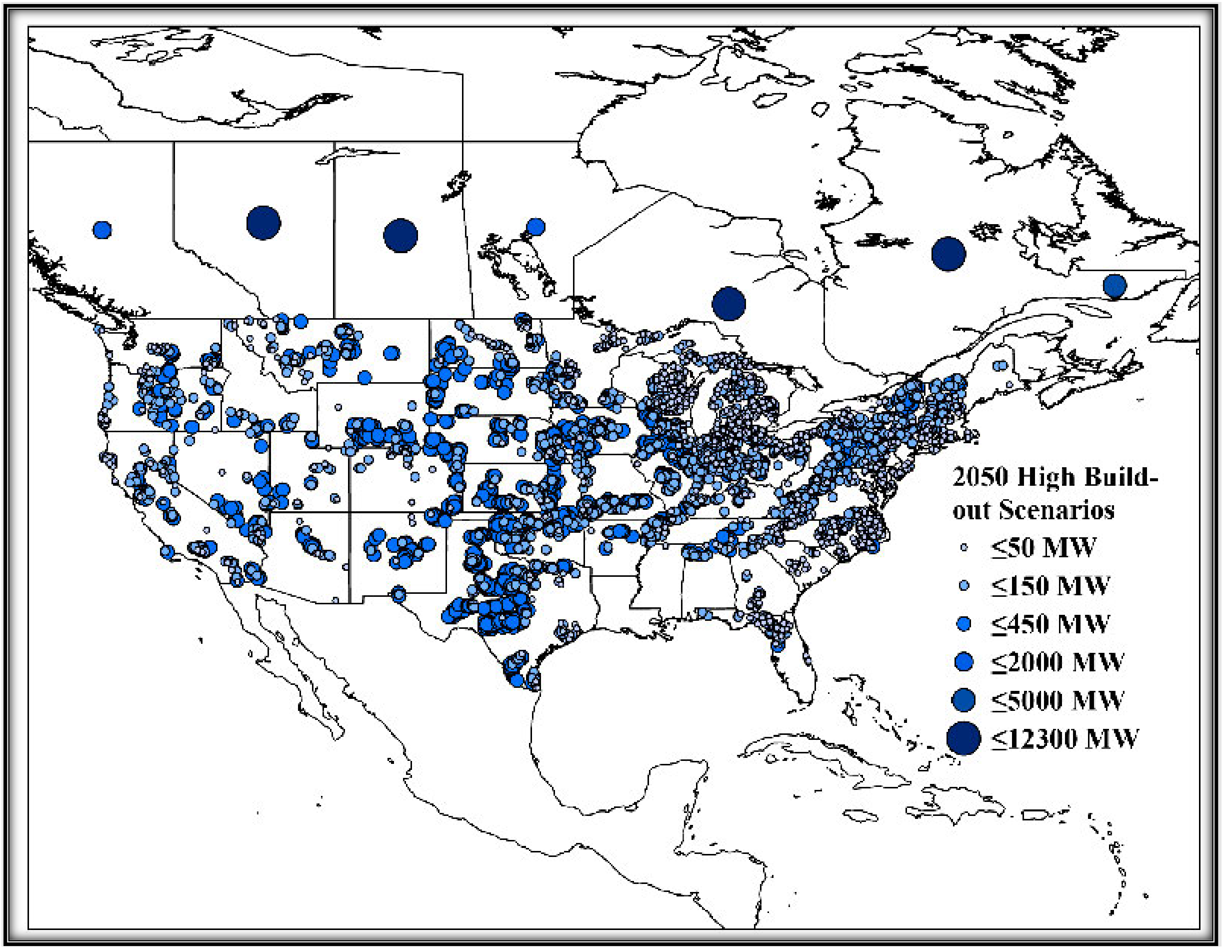

Figure 4.11. Projected wind build-out for the year 2050 per high build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km NREL grid cell and Canadian future capacity by Province.

Table 4.4. Predicted mean annual TCB mortality[6] (25th-75th percentile) by USFWS Region and Canada, based on projected 2050 installed wind capacity under low and high build-out scenarios (Udell et al. 2022, entire).

Location	Low Build-out Mortality	High Build-out Mortality
Region 2	861 (284–877)	2,183 (720–2,233)
Region 3	458 (148–555)	9,546 (2,985–11,743)
Region 4	218 (66–269)	19,590 (5,924–24,332)
Region 5	1,681 (507–2,092)	25,699 (7,750–31,983)
Region 6	90 (37–93)	153 (62–158)
Quebec	4 (1–5)	20 (6–25)
Total	**3,312 (1,043–3,891)**	**57,191 (17,447–70,474)**

[6] Mortality levels are based on pre and post *Pd* arrival % species composition estimates (see Appendix 2). It is likely that % composition will decline as the species declines over time. To capture insights on the sensitivity of the results to wind energy mortality, we ran scenarios with zero and 50% reduction in wind energy mortality (see Appendix 1B).

Climate Change

Over the next few decades, average annual temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C) relative to 1985–2015, regardless of any currently used representative concentration pathway (RCP-2.6 to RCP-8.5) (Hayhoe et al. 2018, p. 86). Larger increases are projected by late century of 2.3–6.7 degrees F (1.3–3.7 degrees C) under RCP4.5 and 5.4–11.0 degrees F (3.0–6.1 degrees C) and 5.4–11.0 degrees F (3.0–6.1 degrees C) under RCP8.5, relative to 1986–2015 (Hayhoe et al. 2018, p. 86).

For the period of 2070–2099 relative to 1986–2015, precipitation increases of up to 20% are projected in winter and spring for northcentral U.S., with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al. 2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

TCB's responses to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as little brown bat (see above and Appendix 4C). This includes reduced reproduction due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years in the Northeast (Frick et al. 2010, pp. 131–133) or reduced reproduction during cooler, wetter spring in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will vary depending throughout the ranges of the species' and on how much the annual temperature actually rises in the future.

Habitat Loss

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources. This assessment was influenced by a set of future scenarios with varying assumptions regarding global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010–2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million ha (16–34 million ac or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million ha (9–21 million ac) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8-7). In addition, TCB is not uniformly distributed across the landscape. While past and projected forest loss and forest regeneration rates can provide a coarse assessment of long-term trends, they are not particularly meaningful for determining the magnitude of impact unless overlaid where the species actually occurs. In addition, forest lands also may remain in that classification (i.e., not converted to other land cover types) while roosts are annually harvested. Loss of essential population needs of roosts and foraging and commuting habitat within TCB's home range where they remain is the issue. Furthermore, loss of summer roosting and foraging habitat and/or winter hibernacula compounds the impacts from WNS (see Appendix 4D).

Synopsis of Future Threats

Using the available data and information summarized above and in Chapters 5 and 6, for each of the primary influences, we assigned the scope, severity, and impact to TCB given the projected future state conditions (Table 4.5). WNS is predicted to continue to be the primary influence (*Very High Impact*), reaching 100% of TCB's range in the U.S. by 2025 (Wiens et al. 2022, pp. 226–229) and causing extreme population declines. Regardless of future low or high-build out, wind energy related mortality maintains the next highest level of impact (*Medium Impact* to *High Impact*) due to its large to pervasive scope (impacting 37–74% of TCB's range) and causing moderate to serious population declines up to 38% by 2060 (Table A-3D2). Both habitat loss and climate change are forecasted to remain pervasive across the species' range, while the severity of population level declines are predicted to increase from current state conditions. Given TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies), the severity of habitat loss at occupied sites will vary between slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between *Low Impact* and *Very High Impact*. Lastly, increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase in the future leading to increased negative effects to TCB (e.g., increased mortality, reduced reproductive success); therefore, our impact analysis predicts *Medium Impact* from climate change under future state conditions. While confidence in the level of impact to TCB from WNS and wind were "high" due to availability of quantitative data, our confidence in our impact analysis of habitat loss and climate change remain "low" to "moderate" due to limited data. See Appendix 3D for additional details.

Table 4.5 Assessment of future impact from primary threats (adapted from Master et al. 2012 and Cheng et al. 2021, p. 5). See criteria definitions in Chapter 1 (Figure 1.2).

Criteria	**WNS**	**Wind Mortality**	**Habitat Loss**	**Climate Change**
Scope	Pervasive	Large-Pervasive	Pervasive	Pervasive
Severity	Extreme	Moderate-Serious	Slight-Extreme	Moderate
Impact	**Very High**	**Medium to High**	**Low to Very High**	**Medium**
Confidence Level	High	High	Moderate	Low

CHAPTER 5 – CURRENT CONDITION

Current viability is the ability of TCB to sustain healthy populations into the future given the current demographic condition of the species and the current state of the influences (Figure 5.1). To assess TCB current viability, we used the BatTool to project future abundance over time, which allows us to assess the future number, health, and distribution of TCB populations given *CURRENT* conditions, and hence, TCB's current viability. In this chapter, we describe the current demographic conditions and the projected number, health, and distribution of TCB populations given these current conditions (i.e., current abundance, growth rate, WNS occurrence, and installed wind energy capacity). We describe the viability implications for TCB in Chapter 7.

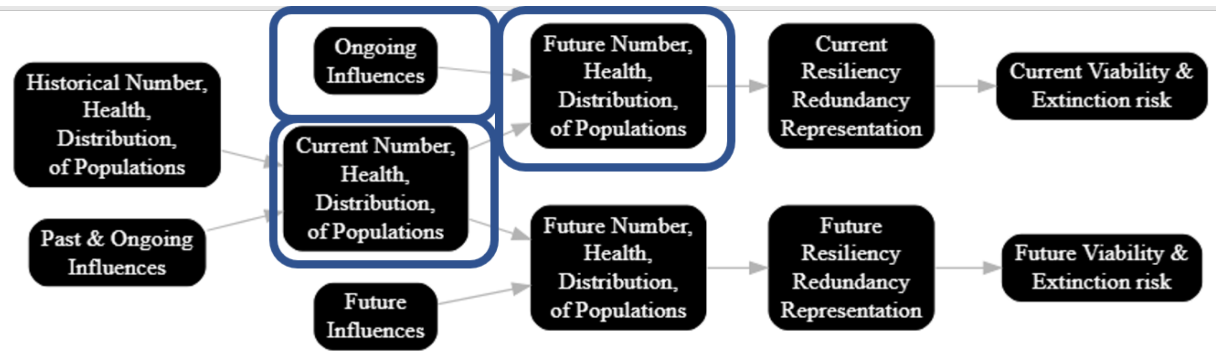

Figure 5.1. Highlighting (blue rectangle) the current step in our analytical framework.

Current demographic conditions based on past declines - Available evidence indicates TCB abundance has and will continue to decline substantially over the next 10 years under current conditions (Figure 5.2). Evidence of the past decline is demonstrated in available data in both winter and summer. For example, rangewide winter abundance has declined by 52% (Figure 5.2) and the number of extant winter colonies (populations) by 29% since 2000 (Table A-3B1). There has also been a noticeable shift towards smaller colony sizes (Figure 5.3). The magnitude of the winter declines, although widespread, varies spatially (Figure 5.4). Abundance has declined 89%, 57%, and 24% in the Eastern RPU, Northern RPU, and Southern RPU, respectively. The number of winter colonies (i.e., occupied hibernacula) have also decreased 46%, 24%, and 34% in the Eastern RPU, Northern RPU, and Southern RPU, respectively. Lastly, across all RPUs, the potential for population growth is currently undetectable, i.e., (λ) >1 is 0% (Table A-3B2).

Declining trends in TCB occurrence and abundance is also evident from summer data. Based on derived rangewide summaries from Stratton and Irvine (2022, pp. 99–108), for example, found rangewide occupancy has declined by 28% from 2010–2019 (Table A-3B4, Figure 5.7). Aggregated metrics of probability of occupancy declined in all RPUs (Table A-3B4). Similarly, Whitby et al. (2022, pp. 162–163) using data collected from mobile acoustic transects found a 53% decline in rangewide relative abundance from 2009–2019. They found measurable declines in the Northern RPU (86%), Southern RPU (65%), and Eastern RPU (38%) (Table A-3B4). Finally, Deeley and Ford (2022, entire) observed a significant decline in mean capture rates from 1999 to 2019 across the range. Estimates derived from their results correspond to a 12% decline in rangewide mist-net capture rates compared to pre-WNS capture rates. Capture rates decreased

19%, 16%, and 12%, in the Eastern RPU, Northern RPU, and Southern RPU, respectively (Table A-3B4).

Future projections based on current conditions - Collectively, these data indicate TCB has declined and given the declining trajectories, will continue to decline. Future projections from the BatTool, assuming no further WNS spread nor increases in wind capacity (current stressor conditions), show continued declines in rangewide abundance, number of hibernacula, and spatial extent in the future.

- By 2030 (~ 1 bat generation), rangewide abundance declines by 89% (CI 81–94%) (Figure 5.2; Table A-3B1).
- The number of winter colonies (i.e., occupied hibernacula) declines by 91%, with only 171 of 1,951 historical hibernacula occupied by 2030 (Figure 5.5) and only 49 extant hibernacula by 2040 (Table A-3B1).
- The colony size also declines, with the number of large hibernacula (≥100 bats) declining from 127 to 21 (83% decline) between 2020 and 2030 (Figure 5.3).
- Subsequent to declines in the number of hibernacula, TCB's range declines by 65% and winter occurrence becomes more concentrated (Table A-3B1), with 53 hibernacula containing 90% of individuals by 2030.

The projected declines are widespread across RPUs. Abundance declines in all RPUs through 2040, though afterward, there is a modest increase in TCB abundance projected in the Northern and Southern RPUs (Figure 5.4). Despite these projected increases in abundance, however, TCB's spatial extent and number of extant hibernacula will continue to decrease across all RPUs under the current scenario (Table A-3B1, Table A-3B2).

- In the Eastern RPU, abundance and the number of extant hibernacula decline by 99% by year 2030. Of the 211 historical sites, TCB will persist in only 3 hibernacula (Table A-3B2) and there is 0 probability that >500 bats will persist (Figure 5.6).
- In the Northern RPU, abundance declines by 94% and the number of extant hibernacula by 91% by year 2030. Of the 1,124 historical sites, TCB will persist in only 97 hibernacula (Table A-3B2) and only 7 will be large (≥100 individuals) (Figure 5.3).
- In the Southern RPU, abundance declines by 66% and the number of extant hibernacula by 88% by year 2030. Of the 616 historical sites, TCB will persist in 71 hibernacula (Table A-3B2) and only 14 will be large (≥100 individuals) (Figure 5.3).

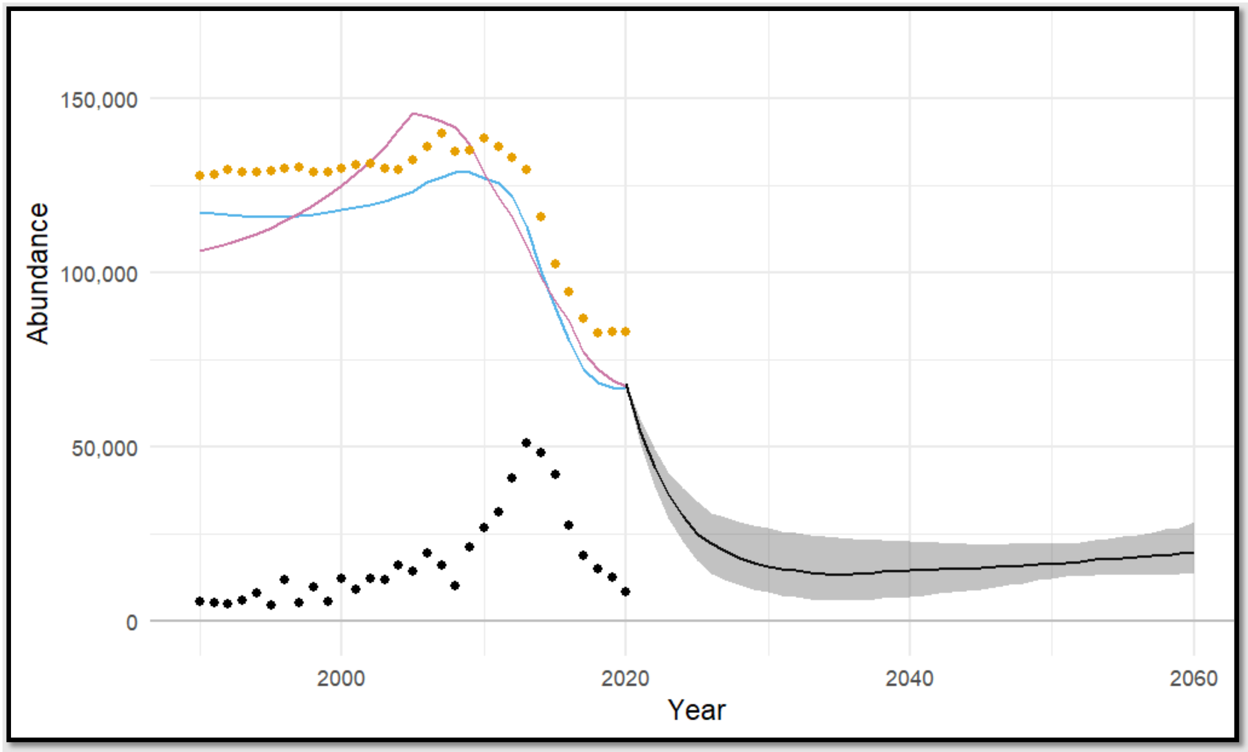

Figure 5.2. Median projected rangewide abundance (black line) and 90% CI (gray shading) given CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity). Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status and trend model informed by Pd occurrence model 2 (pink line).

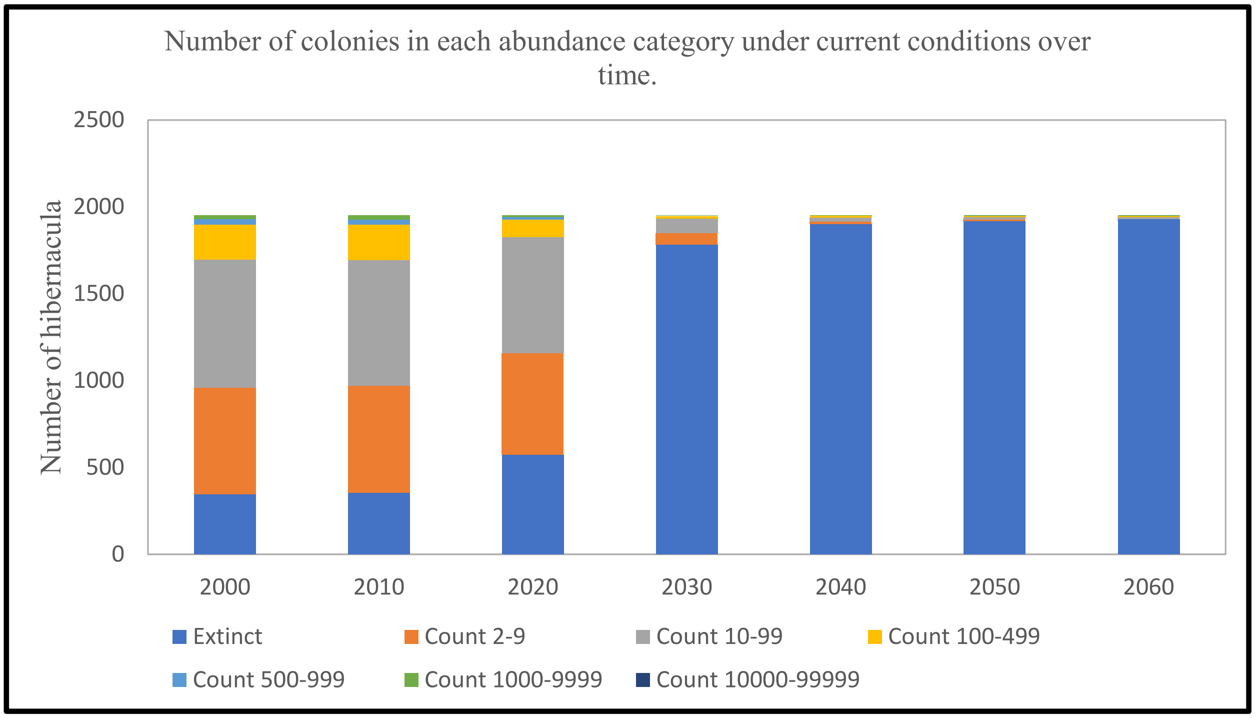

Figure 5.3. The number of hibernacula in each colony abundance category under CURRENT state conditions.

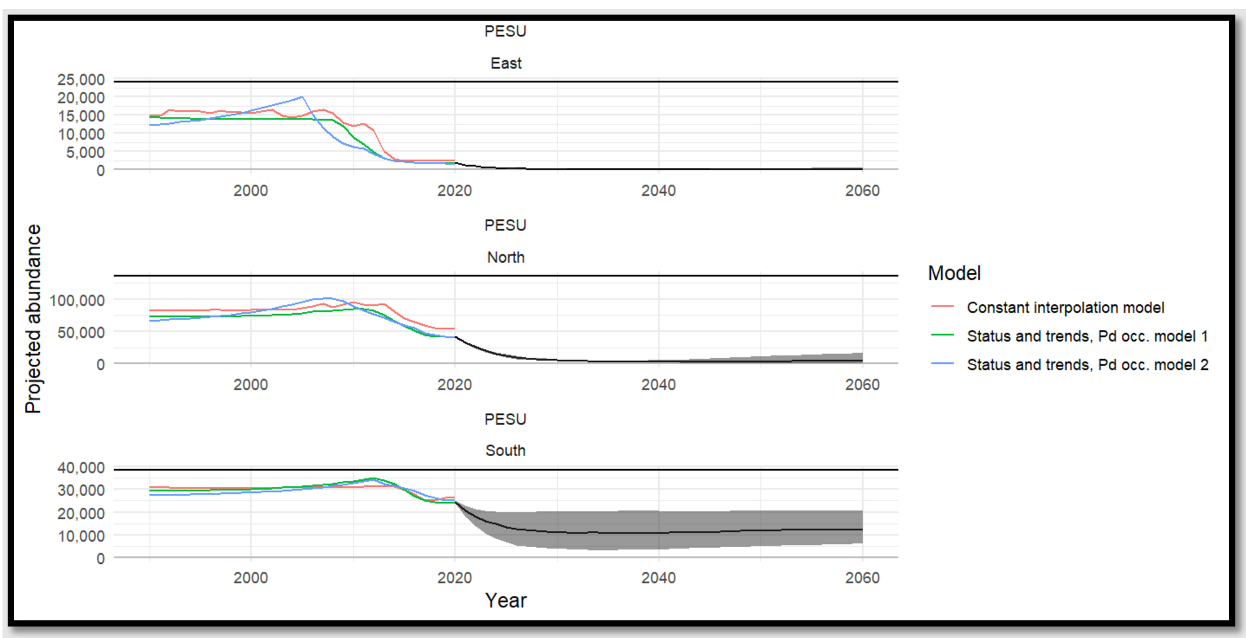

Figure 5.4. Median projected RPU abundance (black line) and 90% CI (gray) under CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity for the three RPUs. Abundance from 1990–2020 derived from winter colony count data using a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).

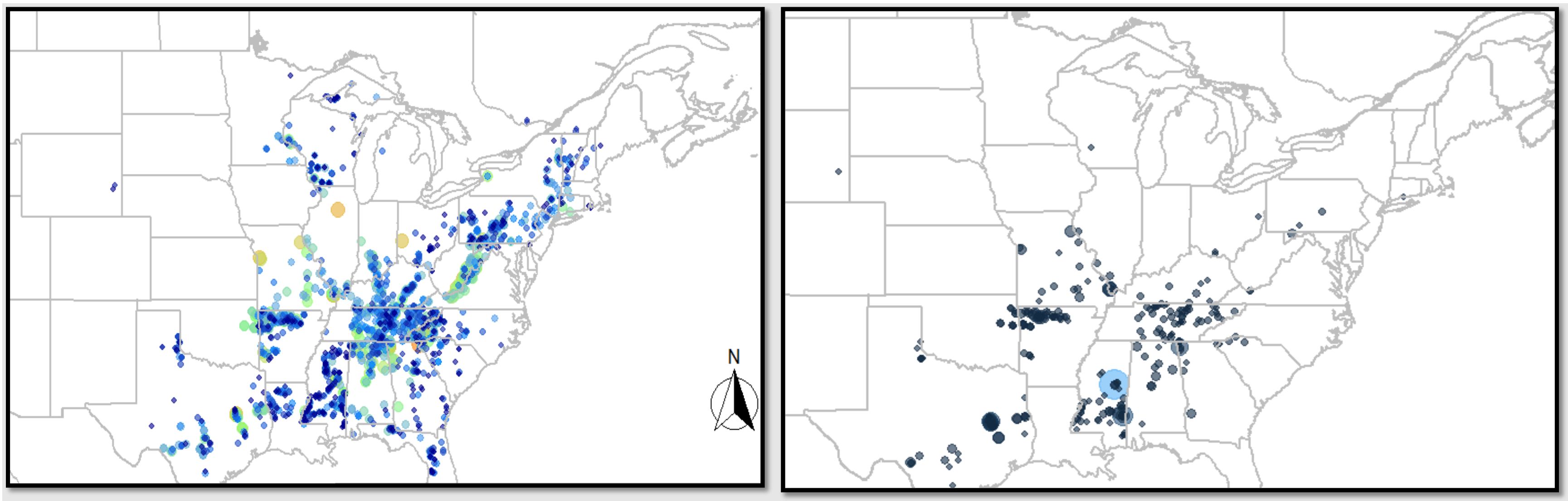

Figure 5.5. TCB extant hibernacula in 2000 (left) and projected 2030 (right) given CURRENT state conditions. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

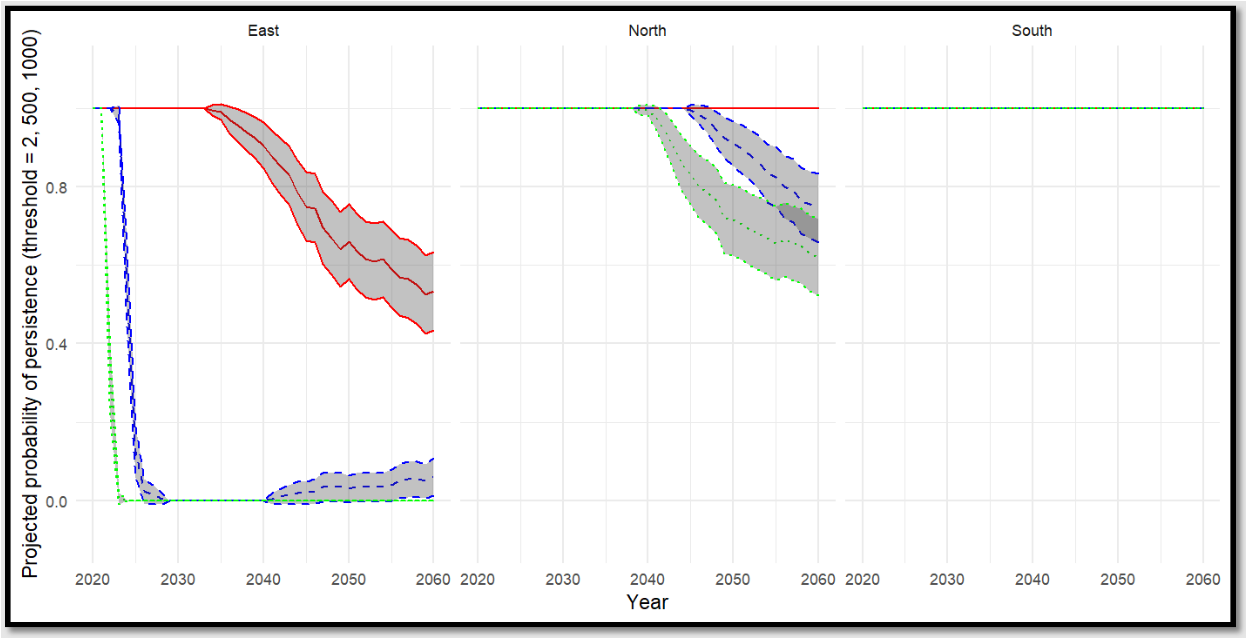

Figure 5.6. Probability of RPU-abundance remaining above X individuals given CURRENT state conditions, x= 2 bats (red), x=500 bats (blue), and b) x=1000 bats (green).

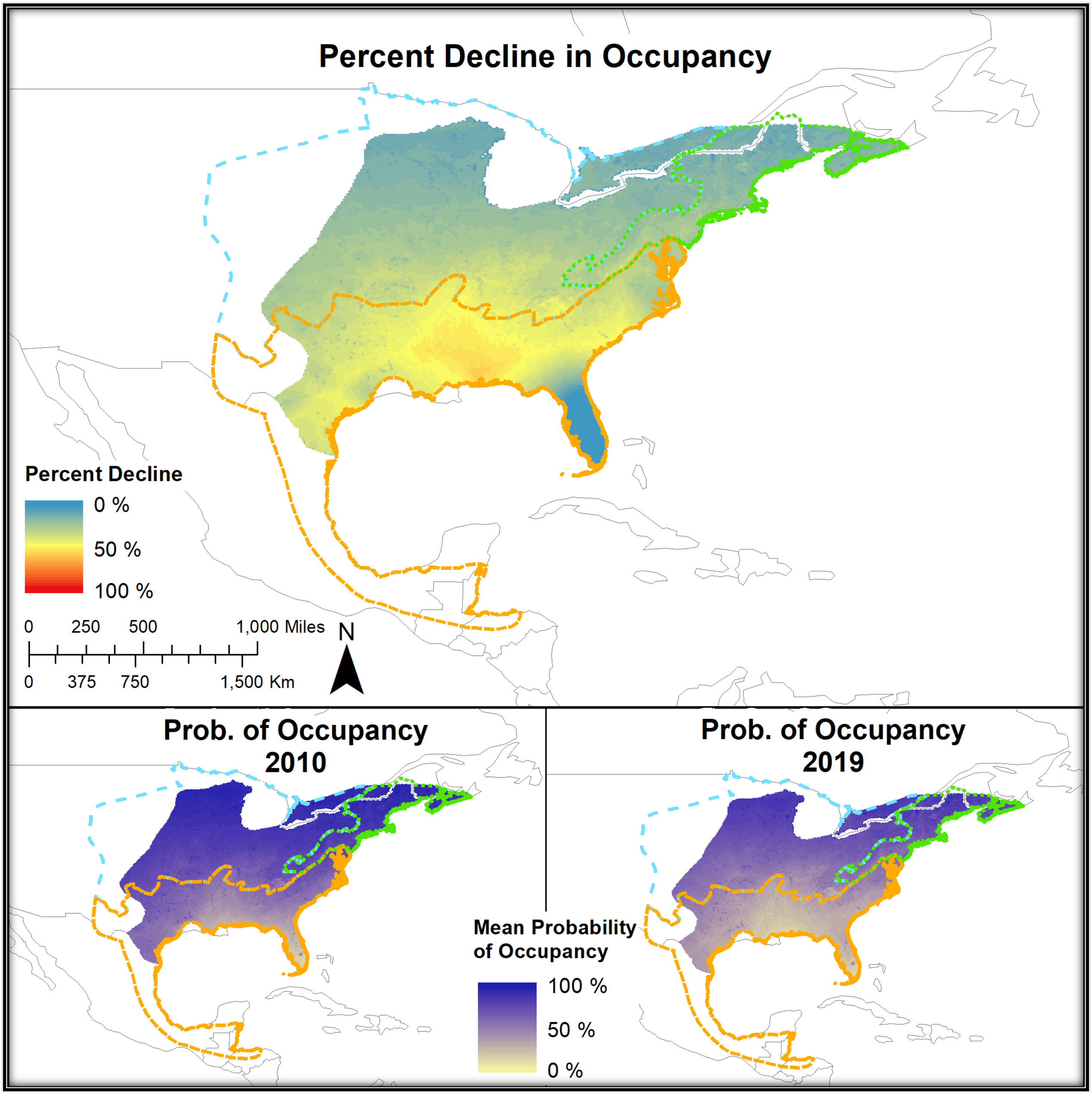

Figure 5.7. Predicted percent decline in probability of occupancy (top) and probability of TCB summer occupancy in 2010 (bottom left) and 2019 (bottom right) based on data collected from stationary and mobile transect acoustic monitoring and capture records summarized at the 10km x 10km NABat grid cell (Stratton and Irvine 2022, entire). Dotted boundaries correspond to representation units. Cooler colors represent lower percent declines (top panel) or higher probability of occupancy (bottom panels).

CHAPTER 6—FUTURE CONDITION

Future viability is the ability of TCB to sustain healthy populations into the future given its current demographic condition and future condition of the influences (Figure 6.1). To assess TCB future viability, we again used the BatTool to project hibernaculum abundance over time given projected *Pd* occurrence and wind energy build-out (see Chapter 4, *Future Scenarios*, for further description). Projection of future number, distribution, and health of populations is needed to understand TCB's future ability to withstand normal stochasticity, stressors, catastrophic events, and novel environmental changes (i.e., its viability under future influences). In this chapter, we describe the projected number, health, and distribution of TCB given *FUTURE* state conditions (i.e., future *Pd* occurrence and future installed wind energy capacity) and describe the viability implications under future influences in Chapter 7.

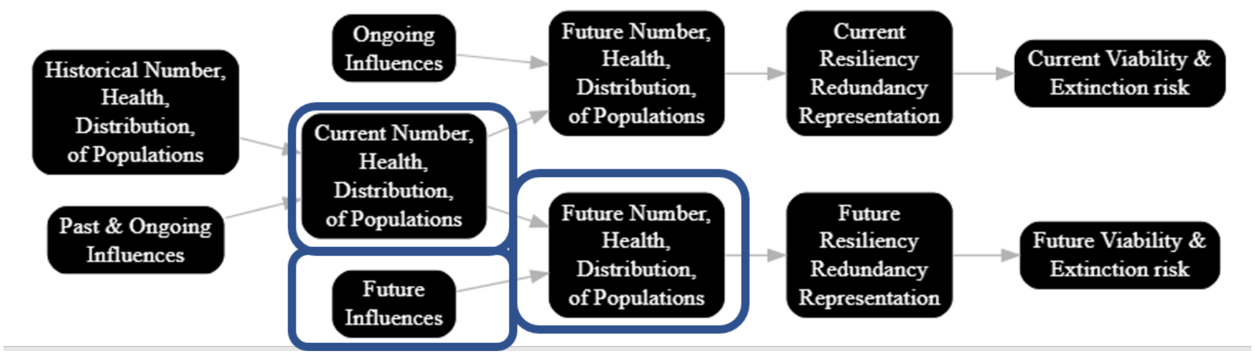

Figure 6.1. Highlighting (blue rectangles) the current step in our analytical framework.

Under the future scenarios, TCB declines worsen precipitously.

- Median rangewide abundance declines 93% (CI 91–94%) by 2030 and 95% by 2040 (Figure 6.2). Under the future scenarios, the decline trajectory halts after year 2040 and slowly grows (Table A-3C1).
- The number of extant hibernacula decline 94% by 2030 and decline 100% (CI 89–100%) by 2060, whereby, only 9 (CI 2–214) hibernacula remain (Figure 6.3, Table A-3C1).
- The colony size also declines, with the number of large hibernacula (≥100 bats) declining from 127 to 11 (91% decline) between 2020 and 2030 (Figure 6.4).
- Subsequent to declines in the number of hibernacula, TCB's range declines by 70% and winter occurrence becomes more concentrated (Table A-3C1), with 42 hibernacula containing 90% of individuals by 2030.

Similar to projections under current conditions, declines under the future scenarios are widespread across RPUs (Figure 6.5, Table A-3C2).

- In the Eastern RPU, median abundance and number of extant hibernacula both decline by 99% by year 2030. Of the 211 historical sites, TCB will persist in only 2 hibernacula (Table A-3C2) and there is 0 probability that >500 bats will persist (Figure 6.6).
- In the Northern RPU, median abundance and number of extant hibernacula decline by 95% and 93%, respectively, by year 2030. Of the remaining 81 extant hibernacula (out of 1,124 historical sites), only 7 will be large (≥100 individuals) by year 2030 (Figure 6.4).

- In the Southern RPU, median abundance and number of extant hibernacula decline by 84% and 93%, respectively, by year 2030. Of the remaining 41 extant hibernacula (out of 616 historical sites), only 14 will be large (≥100 individuals) by 2030 (Figure 6.4).

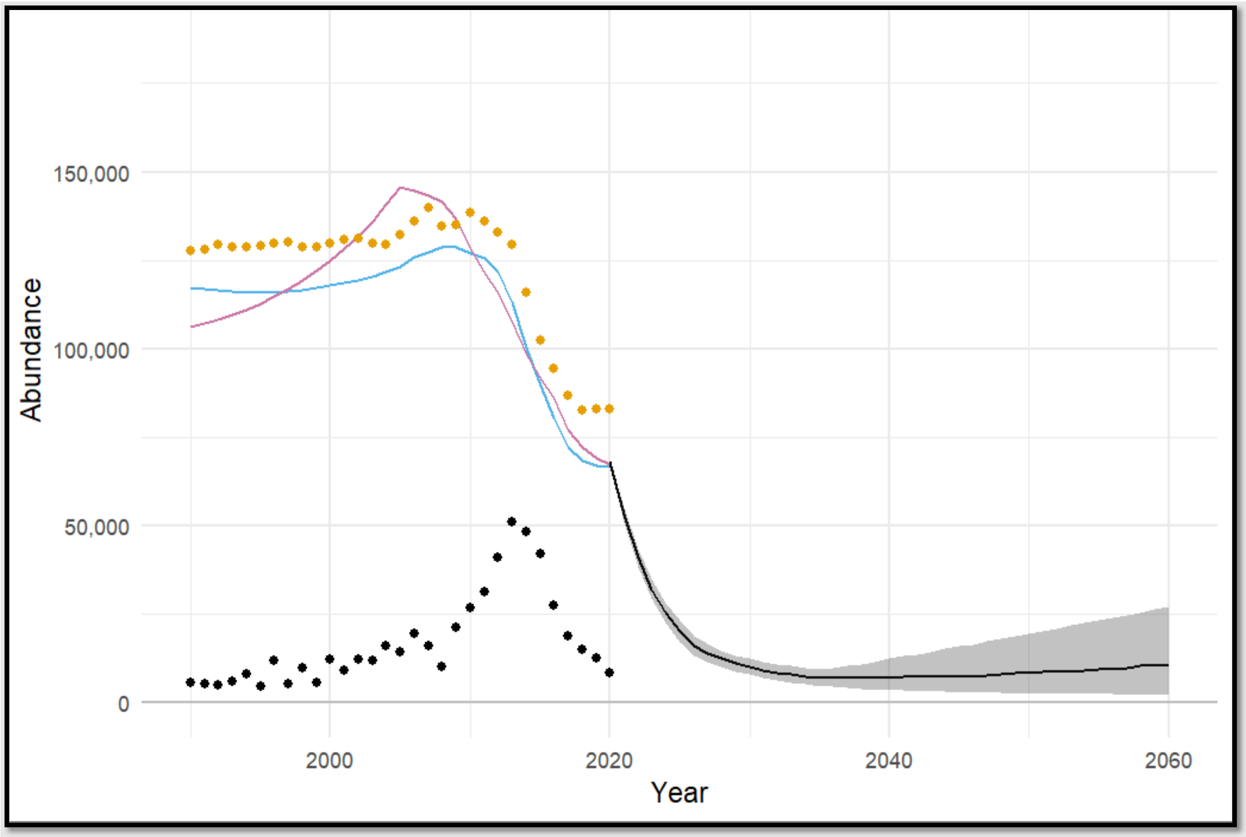

Figure 6.2. Projected median rangewide abundance (black line) and 90% CI (gray shading) under FUTURE state conditions. Abundance from 1990–2020 derived from raw data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status and trend model informed by Pd occurrence model 2 (pink line).

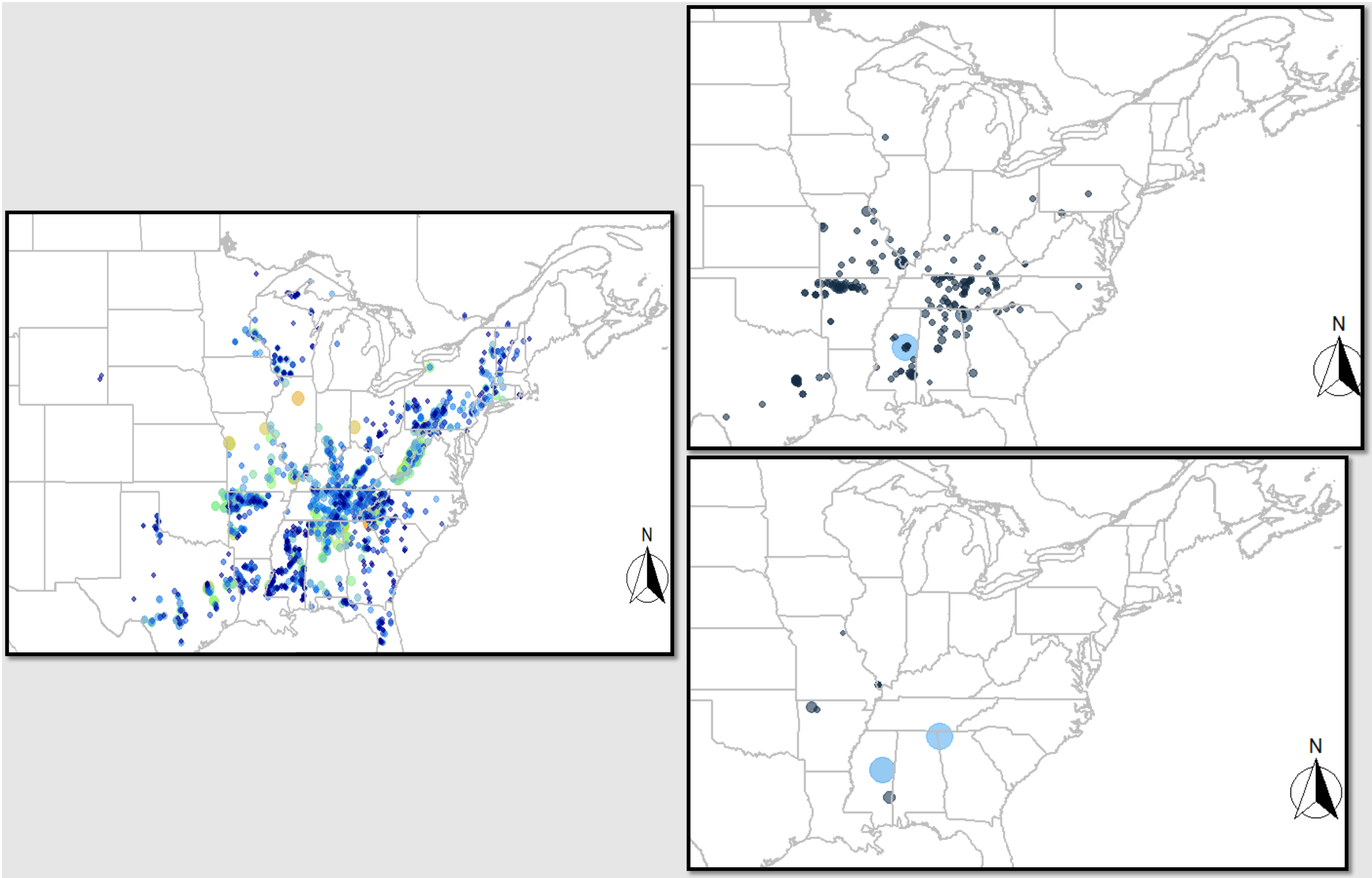

Figure 6.3. TCB extant hibernacula in 2000 (left) and projected 2030 (upper right) and 2060 (bottom right) given FUTURE state conditions. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

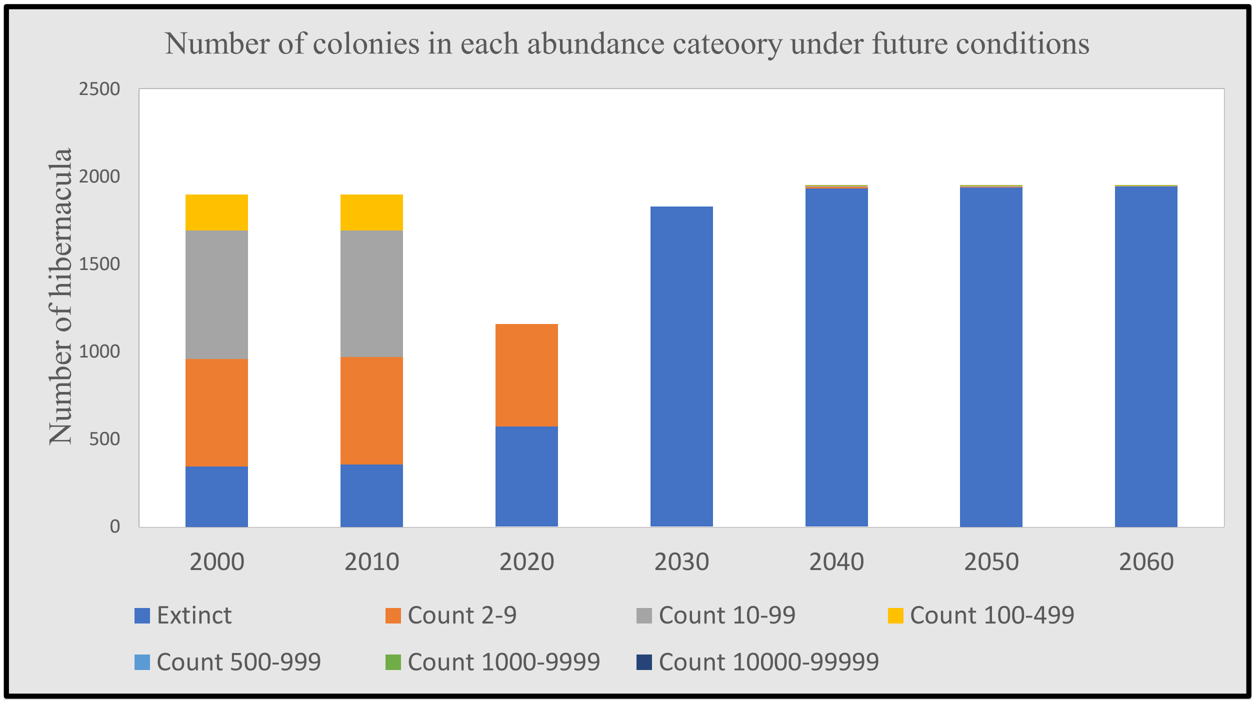

Figure 6.4. The number of hibernacula in each colony abundance category under FUTURE state conditions.

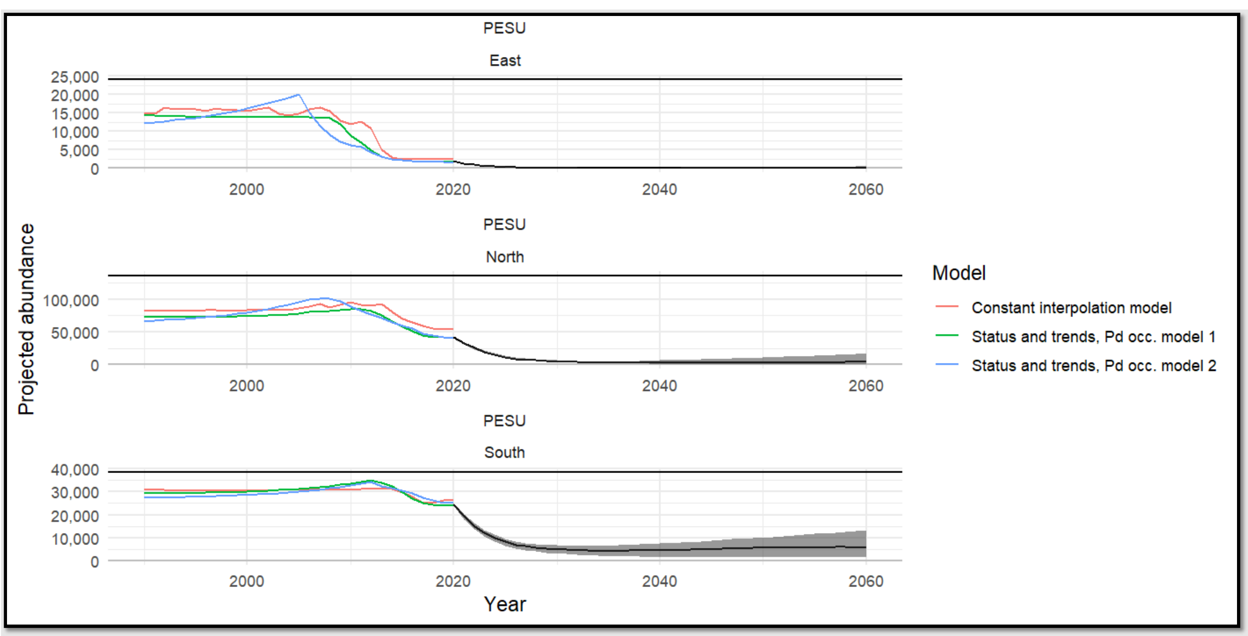

Figure 6.5. Projected median (black line) and 90% CI (gray shading) for RPU abundance under FUTURE state conditions for the three RPUs. Abundance from 1990–2020 derived from raw data using a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).

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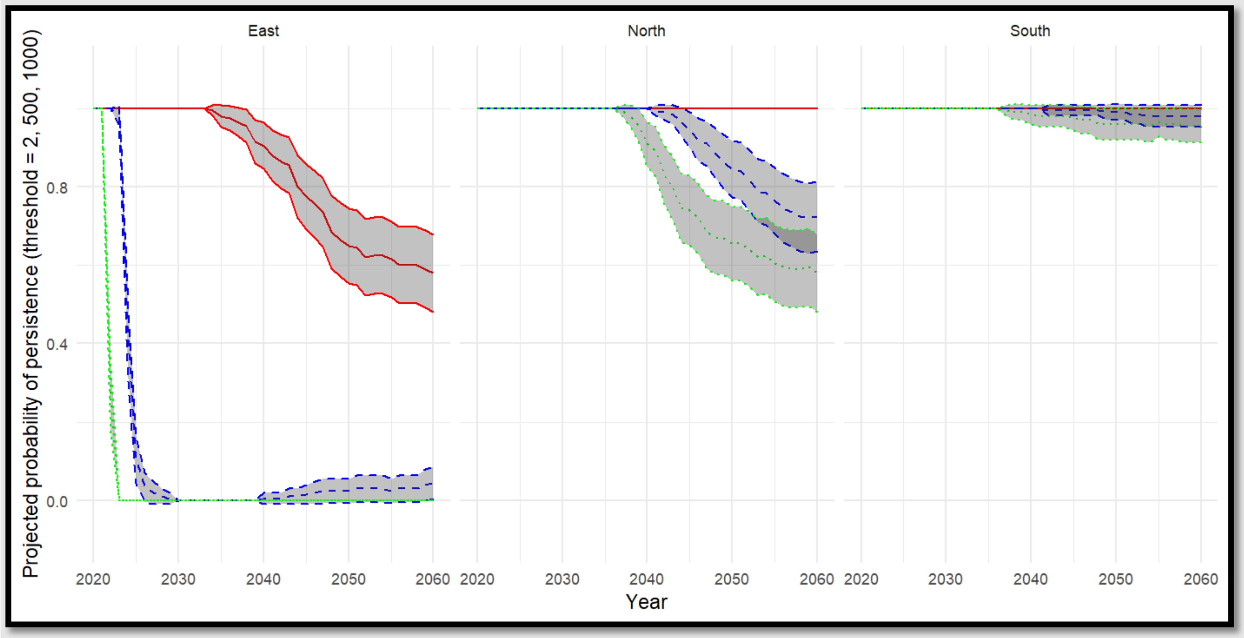

Figure 6.6. Probability of RPU-abundance remaining above X individuals given FUTURE state conditions, x= 2 bats (red), x=500 bats (blue), and b) x=1000 bats (green).

Habitat Loss and Climate Change

As discussed previously, we did not incorporate habitat loss and the effects of climate change into our quantitative modeling efforts (i.e., not included in the projections depicted in Figures 6.2–6.5). Ongoing effects from habitat loss and climate change likely continue into the future and may even be exacerbated based on reduced abundance and distribution anticipated under our current and future scenarios. See Table 4.5 for a description of the scope, severity, and impact of future habitat loss and climate change impacts. Additionally, future impacts from habitat loss and climate change are discussed more thoroughly in the Appendix 4.

CHAPTER 7—SPECIES VIABILITY

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences for TCB viability (Figure 7.1). TCB viability is influenced by the number, health, and distribution of populations. Across the range and within all RPUs, TCB abundance and distribution has decreased. Multiple data types and analyses indicate downward trends in TCB population abundance and distribution (Table 7.1), and we found little evidence to suggest that this downward trend will change in the future (Figure 7.2). Like all species status assessments we do not have perfect information on TCB's occurrence, but the best available data suggest that bats at unknown hibernacula will undergo similar declines observed at known winter colonies. We outline the key uncertainties in our analyses and our resolution of them in Appendix 1.

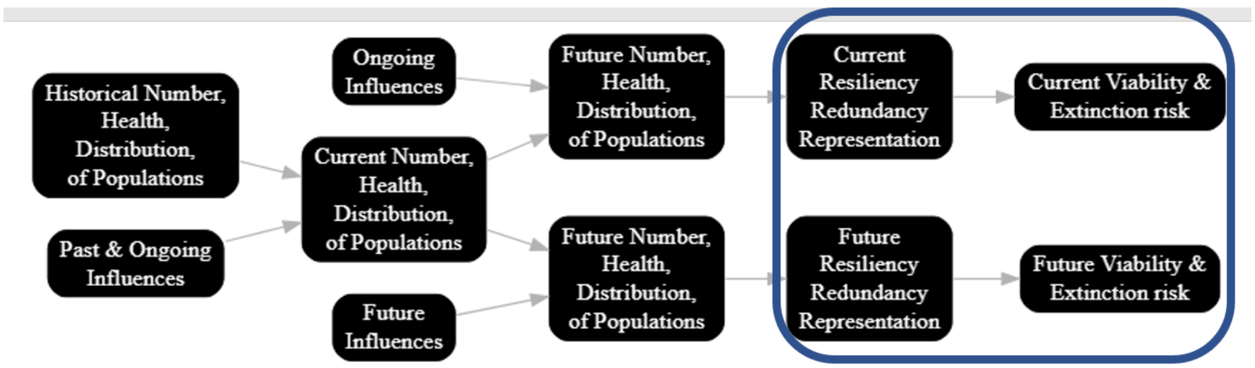

Figure 7.1. Highlighting (blue rectangle) the current step in our analytical framework.

Table 7.1. Summary of recent TCB population trends from multiple data types and analyses. Winter Colony analysis – derived from Wiens et al. (2022, entire) data; Summer Occupancy analysis – Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire).

Scale	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Eastern	-89%	-17%	-19%	-38%
Northern	-57%	-17%	-16%	-86%
Southern	-24%	-37%	-12%	-65%
Rangewide	-52%	-28%	-12% to -19%	-53%

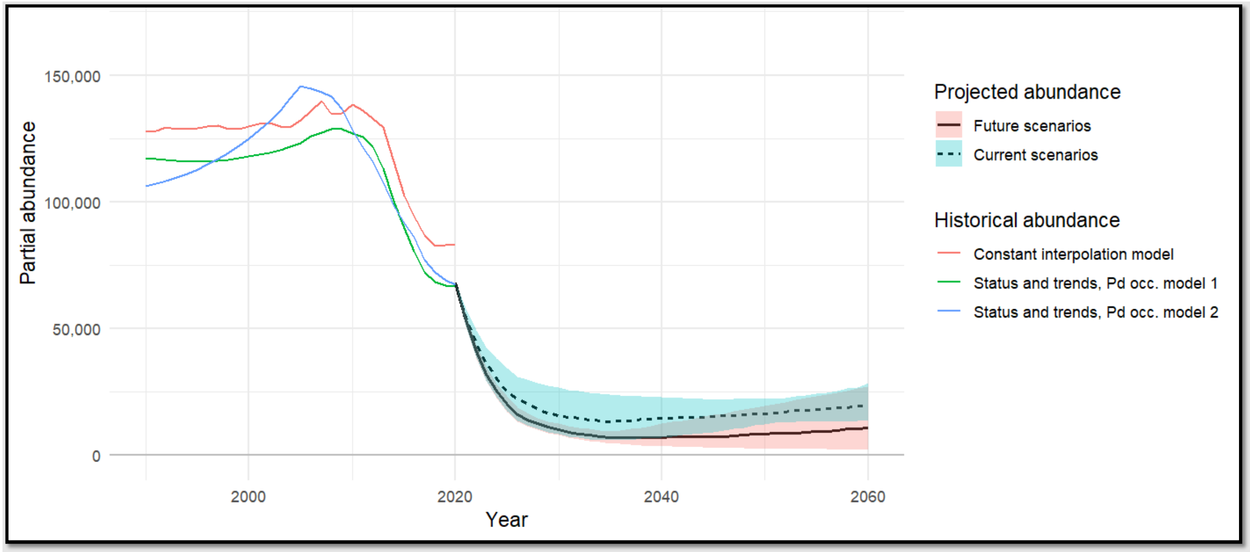

Figure 7.2. The projected TCB abundance over time given current (blue) Pd occurrence and installed wind capacity and plausible future scenarios (pink) for Pd occurrence and increased installed wind energy capacity. The solid, black lines represent the median abundance under current and future scenarios, respectively. Historical abundance from 1990–2020 derived from a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).

The viability of a species depends upon its ability to sustain populations in the face of normal environmental and demographic stochasticity, catastrophes, and novel changes in its environment. For example, demographically and physically healthy populations better withstand and recover from environmental variability and disturbances. Additionally, populations spread across heterogeneous conditions are unlikely to be exposed at the same time to poor environmental conditions, thereby guarding against synchronous population losses. Similarly, species with genetically healthy populations (large *Ne*, which begets genetic diversity) spread across the breadth of genetic and phenotypic diversity preserve a species' adaptive capacity, which is essential for adapting to their continuously changing environment (Nicotra et al. 2015, p. 1269). Without such variation, species are less responsive to change and more prone to extinction (Spielman et al. 2004, p. 15263). Lastly, having multiple healthy populations widely distributed guards against losses of adaptive diversity and RPU-level extirpation in the face of catastrophic events.

We quantitatively assessed TCB's current viability by projecting the species' abundance and distribution given current WNS occurrence (no further spread) and current installed wind energy capacity, and future viability given future plausible scenarios of further WNS spread and increased wind energy capacity. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. All existing data and our qualitative and quantitative analyses suggest that TCB viability has and will continue to steeply decline over time under the current and plausible future conditions.

WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future (Table 7.2). Currently, WNS occurs across 59% of TCB's range (Cheng et al. 2021, p. 7) and is estimated to be impacting 85–100% of hibernacula (Wiens et al. 2022, pp. 231–247). In addition, WNS is predicted to reach 100% of the species' range in the U.S. by 2025 (Wiens et al. 2022, pp. 226–229; Figure A-2B4). Prior to WNS, TCB was abundant and widespread, and abundance and occupancy were generally stable (Cheng et al. 2022, p. 205). WNS impacts have resulted in most winter colonies experiencing a 90–100% decline in abundance compared to historical conditions (Cheng et al. 2021, p. 7).

Wind energy related mortality is proving to be a pervasive and consequential driver to TCB's viability (Table 7.2). Based on 2020 wind build-out, an estimated 1,021 to 3,778 (mean = 3,327) TCB are killed annually at wind facilities and annual mortality is projected to increase to 3,312 to 57,191 individuals by 2050 under the future low and high build-out scenarios, respectively (Figures 4.10 and 4.11). Wind related mortality is discernible, even with ongoing declines from WNS (Figure A-1B2; see also Whitby et al. 2022, pp. 151–153). TCB abundance is projected to decline between 19–21% by 2030 and up to 35% by 2060 from current wind related mortality alone and up to 38% under the future scenarios (Tables A-3D1 and A-3D2). Consequently, mortality from wind turbines will continue to cause detectable declines in TCB abundance.

Although we consider habitat loss pervasive across TCB's range, impacts to TCB and its habitat are often realized at the individual or colony level. Loss of hibernation sites (or modifications such that the site is no longer suitable) can result in impacts to winter colonies. Impacts from forest loss (e.g., roosting or foraging habitat) vary depending on the timing, location, and extent of the removal. Given how common and wide-ranging TCB was prior to the arrival of WNS, we assume the rangewide magnitude of impact from habitat loss was low. However, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies) and remaining populations are anticipated to be less resilient, habitat loss at occupied sites will vary from slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between low to very high (Table 7.2).

Climate change impacts are challenging to describe for wide-ranging species, such as TCB. The changing climate has had and will likely continue to have a multitude of impacts on species throughout North America (Foden et al. 2018, p. 9). Despite being pervasive, however, we believe the rangewide magnitude of impact is currently low (Table 7.2). In addition, there are questions about whether some negative effects are currently offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). Although there may be some offsetting of effects under current climate conditions, increasing negative impacts are anticipated in the future (Table 7.2). Increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase, leading to increased TCB mortality and reduced reproductive success. As mentioned above, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies) and populations are anticipated to be less resilient, effects from climate change may be more impactful.

Table 7.2. Threat (impact) level for the primary influences currently and projected future low and high impact scenarios.

	WNS	Wind Mortality	Habitat Loss	Climate Change
Current	**High**	Medium	Low	Low
Low Impact	**Very High**	Medium	Low	Medium
High Impact	**Very High**	**High**	**Very High**	Medium

While we focused our analyses on ongoing and anticipated effects from WNS, wind, climate change and habitat loss, we also recognize that novel threats (e.g., new disease or invasive species) may emerge for TCB. TCB's mobility and roost-shifting behaviors provide mechanisms for individual bats to respond to changes in temperature, prey availability and roost suitability. However, as discussed in Chapter 2 and Appendix 2B, temperate zone insectivorous bats including TCB have several inherent traits that limit their ability to respond to changes in the environment, especially to rapid changes. These include their high site fidelity (winter and summer), concentration of individuals in both winter and summer, and specialized winter habitat requirements and summer roost needs. We have already observed the extremely limited ability for TCB to respond to the novel threat WNS. Most exposed to WNS have died and many individual bats that survive a year of exposure continue to return to infected hibernacula.

Viability under Current Conditions

Under current conditions, TCB abundance, number of occupied hibernacula, spatial extent, probability of persistence, summer habitat occupancy (measured by bat captures and acoustic recordings) across the range and within all RPUs are decreasing (Chapter 5 and Table 7.1). Since the arrival of WNS, TCB abundance steeply declined, with most (93%) winter colonies having fewer than 100 individuals. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, TCB's ability to recover from these low abundances is limited given their low reproduction output (two pups per year). Therefore, TCB's resiliency is greatly compromised in its current condition. Additionally, TCB's spatial extent is projected to decline, with 65% reduction by 2030. As TCB's abundance and spatial extent decline, TCB will also become more vulnerable to catastrophic events.

In addition to reduced redundancy and resiliency, TCB's representation has also been reduced. As explained above, TCB's capacity to adapt is constrained by its life history and the level of its intraspecific diversity (e.g., genetic, phenotypic, behavioral, ecological variability). The steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reduced TCB adaptive capacity. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit TCB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen TCB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further WNS spread and additional wind energy development, TCB's viability is likely to rapidly decline over the next 10 years (Figures 7.2 and 7.3).

Viability under Future Scenarios

Under the projected range of plausible future scenarios, WNS spread reaches 100% of TCB's range (Wiens et al. 2022, pp. 226–229) and wind energy related mortality increases by 66% to more than 2000% (Udell et al. 2022, entire; see Table 4.4). By 2060, TCB abundance declines by 92% (Figure 7.1) and the number of extant hibernacula declines by 100% (CI 89–100%) (Figure 7.3). Under the future scenario, by 2060, 0 out of 211 hibernacula remain in the Eastern RPU and only 6 out of 1,124 and 3 out of 616 hibernacula remain in the Northern and Southern RPUs, respectively (Figure 7.3). Given the projected low abundance, the few number and restricted distribution of winter colonies, TCB's currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

Uncertainty remains with regards to WNS impacts to hibernating TCB at road-associated culverts in the southern U.S. As discussed in *Individual-level Ecology and Needs*, culverts account for the majority of hibernacula documented in Louisiana and Mississippi. No *Pd* has been detected at culverts in Louisiana (Limon et al. 2018; entire) and although *Pd* has been detected since 2014 at several culverts that house overwintering TCB in Mississippi, no disease, mortality, or population impacts have been documented (Cross 2019, entire). A variety of environmental and biological factors may contribute to the differences observed in culverts (e.g., year-round temperature profiles may affect the environmental reservoir of *Pd* and shorter winters and milder climates may affect TCB hibernating behavior and physiology). TCB in the Southern RPU exhibit shorter torpor bouts and remain active and feed during the winter (see Chapter 2), and TCB winter movements within and among culverts has been documented (Anderson et al. undated). Consequently, there is uncertainty associated with progression of WNS within these TCB winter colonies. Regardless, TCB summer occupancy, summer captures, and summer mobile acoustic detections have declined 37%, 12%, and 65% in the Southern RPU, respectively (Table 7.1). And, TCB winter colonies at caves and cave-like hibernacula in the Southern RPU have declined, regardless of similar winter length and climates shared with bats hibernating at culverts (Chapter 4).

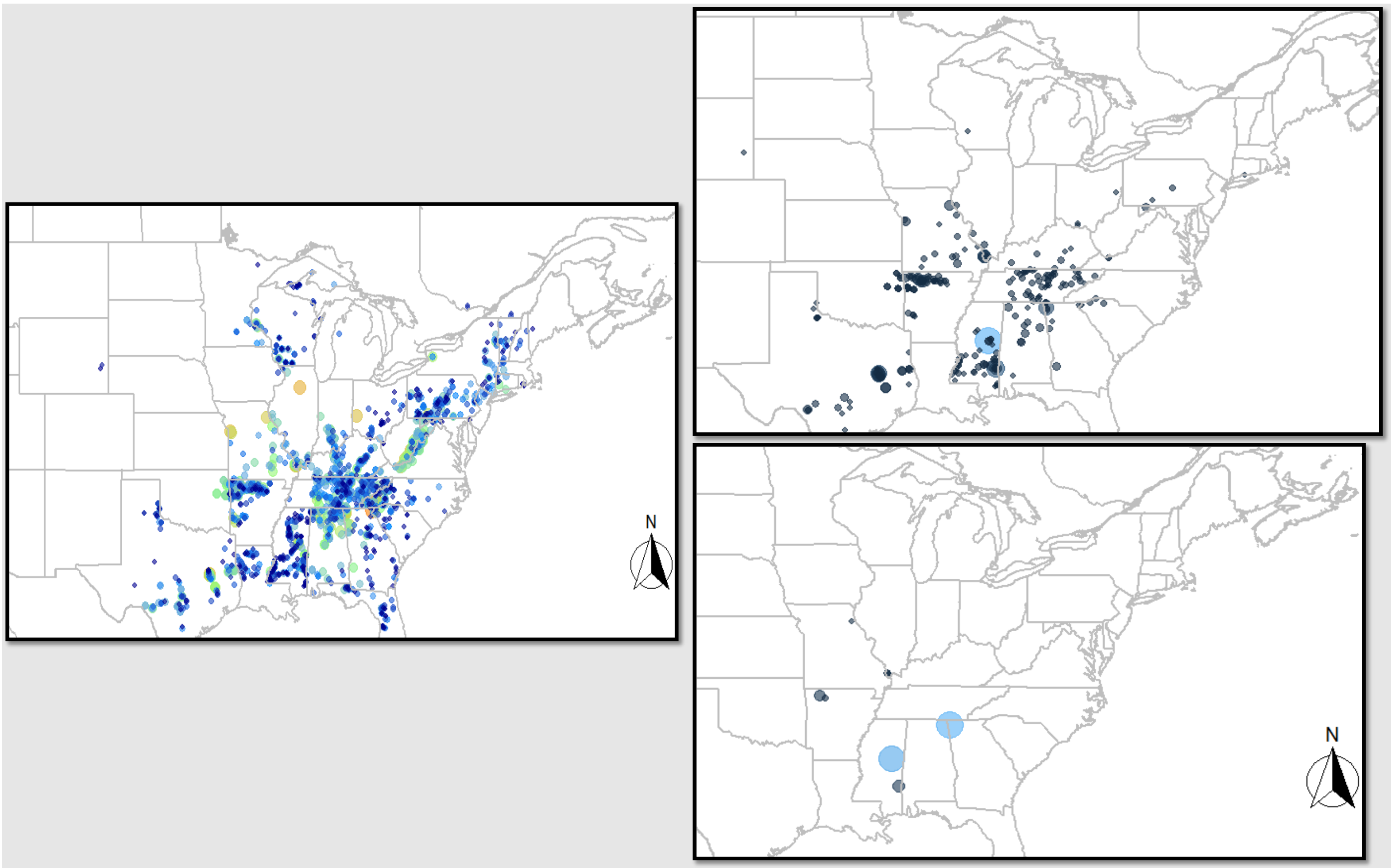

Figure 7.3. Projected change in TCB winter distribution over time: 2000 (far left), 2030 under CURRENT state conditions (top right) and 2060 under FUTURE state conditions (bottom right). Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

REFERENCES CITED

Adams, R.A. 2010. Bat reproduction declines when conditions mimic climate change projections for western North America. Ecology 91(8):2437–2445.

Adams, R.A., B. Stoner, D. Nespoli, and SM. Bexell. 2018. New records of tricolored bats (*Perimyotis subflavus*) in Colorado, with first evidence of reproduction. Western North American Naturalist 78(2):212–215.

Allen, A.A. 1921. Banding bats. Journal of Mammalogy 2(2):53–57.

American Wind Energy Association (AWEA). 2020. Basics of Wind Energy. Available at: https://www.awea.org/wind-101/basics-of-wind-energy

Anderson, A.N., M.C. Arias, M.L. Hoggatt, J.M. LaCour, and A.M. Long. Undated. Noninvasive color marking to examine movement patterns in Louisiana culverts. Louisiana Department of Wildlife and Fisheries, Baton Rouge, Louisiana; Louisiana State University Agricultural Center School of Renewable Natural Resources, Baton Rouge, Louisiana.

Arnett E.B., C.D. Hein, M.R. Schirmacher, M.M.P. Huso, and J.M. Szewczak. 2013. Evaluating the effectiveness of an ultrasonic acoustic deterrent for reducing bat fatalities at wind turbines. PLoS ONE 8(6):e65794.

Bailey, R.G. 2016. Bailey's ecoregions and subregions of the United States, Puerto Rico, and the U.S. Virgin Islands. Fort Collins, CO: Forest Service Research Data Archive. https://doi.org/10.2737/RDS-2016-0003.

Barbour, R.W. and W.H. Davis. 1969. Bats of America. The University Press of Kentucky, Lexington, Kentucky.

Beever, E.A., J. O'Leary, C. Mengelt, J.M. West, S. Julius, N. Green, D. Magness, L. Petes, B. Stein, A.B. Nicotra, and J.J. Hellmann. 2016. Improving conservation outcomes with a new paradigm for understanding species' fundamental and realized adaptive capacity. Conservation Letters 9:131–137.

Benedict, R.A., H.H. Genoways, and P.W. Freeman. 2000. Shifting distributional patterns of mammals in Nebraska. Transactions of the Nebraska Academy of Sciences and Affiliated Societies, 50.

Berkeley Lab. 2020. Electricity Markets and Policy. Wind Technologies Market Report. https://emp.lbl.gov/wind-technologies-market-report. Accessed online: May 21, 2021.

Bernard, R.F., J. Evans, N.W. Fuller, J.D. Reichard, J.T. Coleman, C.J. Kocer, and E.H. Campbell Grant. 2019. Different management strategies are optimal for combating disease in East Texas cave versus culvert hibernating bat populations. Conservation Science and Practice 1(10), e106.

Bernard, R.F., J.D. Reichard, J.T.H. Coleman, J.C. Blackwood, M.L. Verant, J.L. Segers, J.M. Lorch, J. Paul White, M.S. Moore, A.L. Russell, R.A. Katz, D.L. Lindner, R.S. Toomey, G.G. Turner, W.F. Frick, M.J. Vonhof, C.K.R. Willis, and E.H.C. Grant. 2020. Identifying research needs to inform white-nose syndrome management decisions. Conservation Science and Practice 2(8):e220.

Berthinussen, A., O.C. Richardson, and J.D. Altringham. 2021. Bat Conservation: Global Evidence for the Effects of Interventions. Conservation Evidence Series Synopses. University of Cambridge, Cambridge, UK. Available at: www.conservationevidence.com

Blehert, D.S., A.C. Hicks, M.J. Behr, C.U. Meteyer, B.M. Berlowski-Zier, E.L. Buckles, J.T.H. Coleman, S.R. Darling, A. Gargas, R. Niver, J.C. Okoniewski, R.J. Rudd, and W.B. Stone. 2009. Bat White-Nose Syndrome: an emerging fungal pathogen? Science 323:227.

Boyles, J.G. and C.K.R. Willis. 2010. Could localized warm areas inside cold caves reduce mortality of hibernating bats affected white-nose syndrome? Frontiers in Ecology and the Environment 8:92–98.

Brack V. Jr., S.A. Johnson, and R.K. Dunlap. 2003. Wintering populations of bats in Indiana, with emphasis on the endangered Indiana myotis, *Myotis sodalis*. Proceedings of the Indiana Academy of Science 112(1):61–74.

Breed, G., S. Stichter, and E. Crone. 2013. Climate-driven changes in northeastern U.S. butterfly communities. Nature Climate Change 3:142–145.

Briggler, J.T. and J.W. Prather. 2003. Seasonal use and selection of caves by the eastern pipistrelle bat (*Pipistrellus subflavus*). The American Midland Naturalist 149(2):406–412.

Castleberry, S., T. Morris, and E. Ferrall. 2019. Understanding White-nose Syndrome Resilience and Response in Tri-colored bats Using Traditional and Nontraditional Roosts at the Southeastern Disease Frontier. 2019 U.S. Fish and Wildlife Service White-nose Syndrome Grant Proposal.

Carr, J.A., R.F. Bernard, and W.H. Stiver. 2014. Unusual bat behavior during winter in Great Smoky Mountains National Park. Southeastern naturalist 13(2):18–21.

Cheng, T.L., A. Gerson, M.S. Moore, J.D. Reichard., J. DeSimone, C.K.R. Willis, and A.M. Kilpatrick. 2019. Higher fat stores contribute to persistence of little brown bat populations with white-nose syndrome. Journal of Animal Ecology 88:591–600.

Cheng, T.L., J.D. Reichard, J.T.H. Coleman, T.J. Weller, W.E. Thogmartin, B.E. Reichert, A.B. Bennett, H.G. Broders, J. Campbell, K. Etchison, D.J. Feller, R. Geboy, T. Hemberger,

C. Herzog, A.C. Hicks, S. Houghton, J. Humber, J.A. Kath, R.A. King, S.C. Loeb, A. Massé, K.M. Morris, H. Niederriter, G. Nordquist, R.W. Perry, R.J. Reynolds, D.B. Sasse, M.R. Scafini, R.C. Stark, C.W. Stihler, S.C. Thomas, S.C. Thomas, G.G. Turner, S. Webb, B. Westrich, and W.F. Frick. 2021. The scope and severity of white-nose syndrome on hibernating bats in North America. Conservation Biology 2021:1–12.

Cheng, T., B. E. Reichert, W. E. Thogmartin, B. J. Udell, A. M. Wiens, M. Whitby, W. Frick, J.D Reichard, and J. Szymanski. 2022. Winter Colony Count Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter D in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

Chevin, L.M., R. Lande, and G.M. Mace. 2010. Adaptation, plasticity, and extinction in a changing environment: Towards a predictive theory. PLOS Biology 8(4):e1000357.

Cole, W. National Renewable Energy Laboratory (personal communication, August 3, 2020).

Cole, W.S.C., N. Gates, T. Mai, and P. Das. 2020. 2020 Standard Scenarios Report: A U.S. Electricity Sector Outlook. Golden, Colorado: National Renewable Energy Laboratory. Available at: NREL/TP-6A20-77442.

Coleman, J.T.H. and J.D. Reichard. 2014. Bat white-nose syndrome in 2014: A brief assessment seven years after discovery of a virulent fungal pathogen in North America. Outlooks on Pest Management 25(6):374-377.

Cope, J.B. and S.R. Humphrey. 1972. Reproduction of the bats *Myotis keenii* and *Pipistrellus subflavus* in Indiana. Bat Research News 13(9).

Cornelison, C., S. Loeb, T. McElroy, and A. Edelman. 2019. Investigating winter activity and population connectivity of *Perimyotis subflavus* in traditional and nontraditional hibernacula in the Southeastern US as it relates to susceptibility to white-nose syndrome. 2019 U.S. Fish and Wildlife Service White-nose Syndrome Grant Proposal.

Crandall, K.A., O.R. Bininda-Emonds, G.M. Mace, and R.K. Wayne. 2000. Considering evolutionary processes in conservation biology. Trends in Ecology and Evolution, 15:290–295.

Cross, K. 2019. Mississippi white-nose syndrome monitoring. Mississippi Department of Wildlife, Fisheries, and Parks. Jackson, Mississippi. Museum Technical Report No. 206.

Cuvier, F. 1832. Essai de classification naturelle des Vespertilions, et description de plusieurs especes de ce genre. In Nouvelles Annales du Muséum d'Histoire Naturelle, Paris, volume 1:1–20.

Damm, J.P. and K. Geluso. 2008. Use of a mine by eastern pipistrelles (*Perimyotis subflavus*) in east central Nebraska. Western North American Naturalist 68(3):382–389.

Davis, W.H. 1959. Taxonomy of the eastern pipistrelle. Journal of Mammalogy, 40(4):521–531.

Davis, W.H. 1966. Population dynamics of the bat *Pipistrellus subflavus*. Journal of Mammalogy 47(3):383–396.

Davis, W.H. and R.E. Mumford. 1962. Ecological notes on the bat *Pipistrellus subflavus*. American Midland Naturalist, pp. 394–398.

Deeley, S.M., and W.M. Ford. 2022. Summary Assessment of USFWS Capture Data for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter A in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Duchamp, J.E. and R.K. Swihart. 2009. Shifts in bat community structure related to evolved traits and features of human-altered landscapes. Landscape Ecology 23(7):849–860.

Environment and Climate Change Canada. 2018. Recovery Strategy for the Little Brown Myotis (*Myotis lucifugus*), the Northern Myotis (*Myotis septentrionalis*), and the Tri-colored Bat (*Perimyotis subflavus*) in Canada. Species at Risk Act Recovery Strategy Series. Environment and Climate Change Canada, Ottawa. ix + 172 pp.

Erickson, R.A., W.E. Thogmartin, and J.A. Szymanski. 2014. BatTool: an R package with GUI for assessing the effect of White-nose syndrome and other take events on *Myotis* spp. of bats. Source code for biology and medicine 9(1):1–10. https://doi.org/10.1186/1751-0473-9-9

Farrow, L.J. and H.G. Broders. 2011. Loss of forest cover impacts the distribution of the forest-dwelling tri-colored bat (*Perimyotis subflavus*). Mammalian Biology 72(2):172–179. doi: 10.1016/j.mambio.2010.04.004

Feldhamer, G.A., T.C. Carter, A.T. Morzillo, and E.H. Nicholson. 2003. Use of bridges as day roosts by bats in southern Illinois. Transactions of the Illinois State Academy of Science 96(2):107–112.

Ferrara, F.J. and P.L. Leberg. 2005. Characteristics of positions selected by day-roosting bats under bridges in Louisiana. Journal of Mammalogy 86(4):729–735.

Field, K.A., J.S. Johnson, T.M. Lilley, S.M. Reeder, E.J. Rogers, M.J. Behr, and D.M. Reeder. 2015. The white-nose syndrome transcriptome: activation of anti-fungal host responses in wing tissue of hibernating little brown myotis. PLoS Pathog 11(10):e1005168.

Foden, W.B., B.E. Young, H.R. Akcakaya, R.A. Garcia, A.A. Hoffmann, B.A. Stein, C.D. Thomas, C.J. Wheatley, D. Bickfod, J.A. Carr, D.G. Hole, T.G. Martin, M. Pacifici, J.W. Pearce-Higgins, P.J. Platts, P.Cisconti, J.E.M. Watson, and B. Huntley. 2018. Climate change vulnerability assessment of species. WIREs Climate Change 10(e551):1–36. https://doi.org/10.1002/wcc.551.

Forsman A. 2014. Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion, and infection biology. Proceedings of the National Academy of Sciences of the United States of America, 111:302–307.

Francl, K.E., W.M. Ford, D.W. Sparks, and V. Brack. 2012. Capture and reproductive trends in summer bat communities in West Virginia: assessing the impact of white-nose syndrome. Journal of Fish and Wildlife Management 3(1): 33–42. DOI: https://doi.org/10.3996/062011-JFWM-039

Fraser, E.E., L.P. McGuire, J.L. Eger, F.J. Longstaffe, and M.B. Fenton. 2012. Evidence of latitudinal migration in tri-colored bats, *Perimyotis subflavus*. PLoS One 7(2):e31419.

Frick, W.F., S.J. Puechmaille, and C.K.R. Willis. 2016. White-nose syndrome in bats. In C.C. Voigt, and T. Kingston, editors. Bats in the Anthropocene: conservation of bats in a changing world, Springer International Publishing, Cham, Heidelberg, New York, Dordrecht, London.

Frick, W.F., T.L. Cheng, K.E. Langwig, J.R. Hoyt, A.F. Janicki, K.L. Parise, J.T. Foster, and A.M. Kilpatrick. 2017. Pathogen dynamics during invasion and establishment of white-nose syndrome explain mechanisms of host persistence. Ecology 98: 624–631.

Fujita, M.S. and T.H. Kunz. 1984. *Pipistrellus subflavus*. Mammalian species, 228:1-6.

Fuller, N.W., L.P. McGuire, E.L. Pannkuk, T. Blute, C.G. Haase, H.W. Mayberry, T.S. Risch, and C.K.R. Willis. 2020. Disease recovery in bats affected by white-nose syndrome. Journal of Experimental Biology 223: jeb211912 doi:10.1242/jeb.211912.

Funk, W.C., B.R. Forester, S.J. Converse, C. Darst, and S. Morey. 2019. Improving conservation policy with genomics: A guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics, 20:115–134.

Geipel, I., M.J. Smeekes, W. Halfwerk, and R.A. Page. 2019. Noise as an informational cue for decision-making: the sound of rain delays bat emergence. Journal of Experimental Biology:222, jeb192005. doi:10.1242/jeb.192005.

Geluso, K., T.R. Mollhagen, J.M. Tigner, and M.A. Bogan. 2005. Westward expansion of the eastern pipistrelle (*Pipistrellus subflavus*) in the United States, including new records from New Mexico, South Dakota, and Texas. Western North American Naturalist 65(3):12.

Ghalambor C.K., J.K McKay, S.P. Carroll, and D.N. Reznick. 2007. Adaptive versus non-adaptive phenotypic plasticity and the potential for contemporary adaptation in environments. Functional Ecology, 21:394–407.

Gilman, P. Department of Energy (personal communication, July 14, 2020).

Government of Canada. 2020. Canadian Wind Turbine Database. Available at: https://open.canada.ca/data/en/dataset/79fdad93-9025-49ad-ba16-c26d718cc070

Grider, J.F., A.L. Larsen, J.A. Homyack, and M.C. Kalcounis-Rueppell. 2016. Winter activity of coastal plain populations of bat species affected by white-nose syndrome and wind energy facilities. PloS one, 11(11):e0166512.

Griffin, D.R. 1940. Migrations of New England bats. Bulletin of the Museum of Comparative Zoology 86:217–246.

Griffith, L.A. and J.E. Gates. 1985. Food habits of cave-dwelling bats in the central Appalachians. Journal of Mammalogy 66(3):451–460.

Grindal, S., T. Collard, R. Brigham, and R.M.R. Barclay. 1992. The influence of precipitation on reproduction by Myotis bats in British Columbia. The American Midland Naturalist 128(2):339–344. doi:10.2307/2426468.

Groves, C.R., D.B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, and M.G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. BioScience 526:499–512.

Guthrie, M.J. 1933. The reproductive cycles of some cave bats. Journal of Mammalogy 14(3):199–-216.

Haavik, L. 2019. Northeastern U.S. Forest Pests. 2019. U.S. Department of Agriculture, Forest Service, Climate Change Resource Center. http://www.fs.fed.us/ccrc/topics/northeastern-us-forest-pests. Accessed online: March 21, 2021.

Hall, J.S. 1962. A life history and taxonomic study of the Indiana bat, *Myotis sodalis*. Gallery Publication 12, Reading Public Museum, Reading, Pennsylvania.

Hamilton, W.J. 1943. The Mammals of Eastern United States. Comstock Publishing Company, Inc., Binghamton, New York.

Hamilton, W.J. and J.O. Whitaker. 1979. Mammals of the Eastern United States. Cornell University Press, Ithaca, New York.

Hammerson G.A., M. Kling, M. Harkness, M. Ormes, and B.E. Young. 2017. Strong geographic and temporal patterns in conservation status of North American bats. Biological Conservation 212:144–152.

Hanski, I. and M. Gilpin, editors. 1997. Metapopulation Biology: Ecology, Genetics, and Evolution. London, UK: Academic Press. 512 pp.

Hanttula, M.K. and E.W. Valdez. 2021. First record and diet of the tri-colored bat (*Perimyotis subflavus*) from Guadalupe Mountains National Park and Culberson County, Texas. Western North American Naturalist 81(1): article 11. Available at: https://scholarsarchive.byu.edu/wnan/vol81/iss1/11

Hayhoe, K., D.J. Wuebbles, D.R. Easterling, D.W. Fahey, S. Doherty, J. Kossin, W. Sweet, R. Vose, and M. Wehner. 2018: Our changing climate. *In* Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (editors)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 72–144. doi: 10.7930/NCA4.2018.CH2.

Hayman, D.T.S., J.R.C. Pulliam, J.C. Marshall, P.M. Cryan, and C.T. Webb. 2016. Environment, host, and fungal traits predict continental-scale white-nose syndrome in bats. Science Advances 2(1):e1500831.

Hefley, T.J., R.E. Russell, A.E. Ballmann, and H. Zhang. 2020. When and where: estimating the date and location of introduction for exotic pests and pathogens. Available at: https://arxiv.org/abs/2006.16982

Hein, C.D., S.B. Castleberry, and K.V. Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257(4):1200–1207.

Hein, C. and B. Straw. 2021. Proceedings from the State of the Science and Technology for Minimizing Impacts to Bats from Wind Energy. 80 pp. Available at: https://tethys.pnnl.gov/publications/proceedings-state-science-technology-minimizing-impacts-bats-wind-energy

Hendry A.P., M.T. Kinnison., M. Heino, T. Day, T.B. Smith, G. Fitt, C.T. Bergstrom, J. Oakeshott, P.S. Jørgensen, M.P. Zalucki, and G. Gilchrist. 2011. Evolutionary principles and their practical application. Evolutionary Applications, 4:159–183.

Hoen, B.D., J.E. Diffendorfer, J.T. Rand, L.A. Kramer, C.P. Garrity, and H.E. Hunt. 2018. United States Wind Turbine Database (V3.2, October 14, 2020): U.S. Geological Survey, American Clean Power Association, and Lawrence Berkeley National Laboratory data release, https://doi.org/10.5066/F7TX3DN0. Data downloaded 3 November 2020.

Hoffmann A.A. and C.M. Sgro. 2011. Climate change and evolutionary adaptation. Nature 470:479–485.

Holderegger R., U. Kamm, and F. Gugerli. 2006. Adaptive vs neutral genetic diversity: Implications for landscape genetics. Landscape Ecology 21:797–807.

Hoofer, S.R. and R.A. Van Den Bussche. 2003. Molecular phylogenetics of the Chiropteran family Vespertilionidae. Acta Chiropterologica 5(1):1–63.

Hoofer, S.R., R.A. Van Den Bussche, and I. Horáček. 2006. Generic status of the American pipistrelles (Vespertilionidae) with description of a new genus. Journal of Mammalogy 87(5):981–992.

Hoying, K.M. and T.H. Kunz. 1998. Variation in size at birth and post-natal growth in the insectivorous bat *Pipistrellus subflavus* (Chiroptera: Vespertilionidae). Journal of Zoology 245(1):15–27.

Hoyt, J.R., K.E. Langwig, K. Sun, K.L. Parise, A. Li, Y. Wang, X. Huang, L. Worledge, H. Miller, J.P. White, H.M. Kaarakka, J.A. Redell, T. Görföl, S.A. Boldogh, D. Fukui, M. Sakuyama, S. Yachimori, A. Sato, M. Dalannast, A. Jargalsaikhan, N. Batbayar, Y. Yovel, E. Amichai, I. Natradze, W.F. Frick, J.T. Foster, J. Feng, and A.M. Kilpatrick. 2020. Environmental reservoir dynamics predict global infection patterns and population impacts for the fungal disease white-nose syndrome. Proceedings of the National Academy of Sciences 117:7255–7262.

Hoyt, J.R., A.M. Kilpatrick, and K.E. Langwig. 2021. Ecology and impacts of white-nose syndrome on bats. Nature Reviews Microbiology 19(3):196–210.

Humphrey, S.R., R.K. LaVal, and R.L. Clawson. 1976. Nursery populations of *Pipistrellus subflavus* (Chiroptera, Vespertilionidae) in Missouri. Transactions of the Illinois State Academy or Science 69:367.

Johnson, J.S. Ohio University (personal communication, May 5, 2021).

Jones, C. and J. Pagels. 1968. Notes on a population of *Pipistrellus subflavus* in southern Louisiana. Journal of Mammalogy 49(1):134–139.

Jones, C. and R.D. Suttkus. 1973. Colony structure and organization of *Pipistrellus subflavus* in southern Louisiana. Journal of Mammalogy 54(4):962–968.

Jones, G., D.S. Jacobs, T.H. Kunz, M.R. Willig, and P.A. Racey. 2009. Carpe noctem: the importance of bats as bioindicators. Endangered Species Research 8:93–115.

Jones G. and H. Rebelo. 2013. Responses of bats to climate change: learning from the past and predicting the future. In: Adams, R., S. Pedersen. (editors) Bat Evolution, Ecology, and Conservation. Springer, New York, NY. https://doi.org/10.1007/978-1-4614-7397-8_22

Katzenmeyer, J.B. 2016. Use of highway culverts, box bridges, and caves by winter-roosting bats in Mississippi. Master of Science in Wildlife and Fisheries Science, Mississippi State University, Mississippi. 70 pp.

Kerns, J. and P. Kerlinger. 2004. A study of bird and bat collision fatalities at the Mountaineer Wind Energy Center, Tucker County, West Virginia Annual Report for 2003. Unpublished report. 39 pp.

Kindvall O. 1996. Habitat heterogeneity and survival in a bush cricket metapopulation. Ecology 77(1):207–214.

Kurta, A., L. Winhold, J.O. Whitaker, and R. Foster. 2007. Range expansion and changing abundance of the eastern pipistrelle (Chiroptera: Vespertilionidae) in the central Great Lakes region. The American Midland Naturalist 157(2):404–411.

Lane, H.K. 1946. Notes on *Pipistrellus subflavus subflavus* (F. Cuvier) during the season of parturition. Proceedings of the Pennsylvania Academy of Science, 20:57–61.

Langwig, K.E., W.F. Frick, J.T. Bried, A.C. Hicks, T.H. Kunz, and A.M. Kilpatrick. 2012. Sociality, density-dependence and microclimates determine the persistence of populations suffering from a novel fungal disease, white-nose syndrome. Ecology Letters 15(9):1050–1057.

Langwig, K.E., J. Voyles, M.Q. Wilber, W.F. Frick, K.A. Murray, B.M. Bolker, J.P. Collins,T.L. Cheng, M.C. Fisher, J.R. Hoyt, D.L. Lindner, H.I. McCallum, R. Puschendorf, E.B. Rosenblum, M. Toothman, C.K.R. Willis, C.J. Briggs, and A.M. Kilpatrick. 2015a. Context-dependent conservation responses to emerging wildlife diseases. Frontiers in Ecology and the Environment 13(4):195–202.

Lankau R., P.S. Jørgensen, D.J. Harris, and A. Sih. 2011. Incorporating evolutionary principles into environmental management and policy. Evolutionary Applications 4:315–325.

LaVal, R.K. and M.L. LaVal. 1980. Ecological studies and management of Missouri bats, with emphasis on cave-dwelling species. Missouri Department of Conservation, Conservation Commission of the State of Missouri, Terrestrial Series No. 8.

Layne, J.N. 1992. Status of the Eastern Pipistrelle, *Pipistrellus subflavus*, at its southern range limit in eastern United States. Bat Research News 33:43–46.

Limon, D.A., C.J. Garcia, B.B. Gregory, R.D. Stevens, and M.A. Barnes. 2018. The status of *Pseudogymnoascus destructans* in Louisiana. The Southwestern Naturalist 63(4):216–219.

Lindner, D.L., A. Gargas, J.M. Lorch, M.T. Banik, J. Glaeser, T.H. Kunz, and D.S. Blehert. 2011. DNA-based detection of the fungal pathogen *Geomyces destructans* in soils from bat hibernacula. Mycologia 103(2):241–246.

Lorch, J.M., C.U. Meteyer, M.J. Behr, J.G. Boyles, P.M. Cryan, A.C. Hicks, A.E. Ballmann, J.T.H. Coleman, D.N. Redell, D.M. Reeder, and D.S. Blehert. 2011. Experimental infection of bats with *Geomyces destructans* causes white-nose syndrome. Nature 480:376–379.

Lutsch, K. 2019. Assessment of culverts and bridges as roosting habitat for *Perimyotis subflavus* (tri-colored bat) and disease transmission corridors for *Pseudogymnoascus destructans.* Master of Science in Integrative Biology, Kennesaw State University, Georgia. 44 pp.

Mangel, M and C. Tier. 1993. A simple direct method for finding persistence times of populations and application to conservation problems. Proceedings of the National Academy of Sciences of the United States of America, 90:1083–1086.

Martin, A.M. 2014. Historical demography and dispersal patterns in the eastern pipistrelle bat (*Perimyotis subflavus*). Master of Science in Biology, Grand Valley State University, Michigan. 65 pp.

Master, L.L., D. Faber-Langendoen, R. Bittman, G.A. Hammerson, B. Heidel, L. Ramsay, K. Snow, A. Teucher, and A. Tomaino. 2012. NatureServe Conservation Status Assessments: Factors for Evaluating Species and Ecosystem Risk. NatureServe, Arlington, VA. 64 pp.

McClure, M.L., D. Crowley, C.G. Haase, L.P. McGuire, N.W. Fuller, D.T.S. Hayman, C.L. Lausen, R.K. Plowright, B.G. Dickson, and S.H. Olson. 2020. Linking surface and subterranean climate: implications for the study of hibernating bats and other cave dwellers. Ecosphere, 11:e03274. 10.1002/ecs2.3274.

McGuire, L.P., H.W. Mayberry, and C.K.R. Willis. 2017. White-nose syndrome increases torpid metabolic rate and evaporative water loss in hibernating bats. American Journal of Physiology; Regulatory, Integrative and Comparative Physiology 313: R680-R686.

Medina-Fitoria, A., O. Saldaña, J.G. Martínez, Y. Aguirre, W. Silva, M. Chávez, M. Salazar, N. Carballo, O. Jarquín, R.A. González, L. Díaz, C. Chambers, F. Reid, R. Mies, K. Williams, J.M. Zolotoff, C. Molina, T. Pérez, J. Rodríguez, L. Gutiérrez, M. Fernández, R. Mendieta, and J. Pérez. 2015. Nuevos reportes sobre los murcielagos (Mammalia: Chiroptera) de Nicaragua, America Central, con la adicion de siete nuevos registros de especies. Mastozoología Neotropical 22(1):43–54.

Meierhofer, M.B., S.J. Leivers, R.R. Fern, L.K. Wolf, J.H. Young, Jr., B.L. Pierce, J.W. Evans, and M.L. Morrison. 2019. Structural and environmental predictors of presence and abundance of tri-colored bats in Texas culverts. Journal of Mammalogy 100(4):1274–1281.

Merritt, J.F. 1987. Guide to the Mammals of Pennsylvania. University of Pittsburgh Press, Pittsburgh, Pennsylvania.

Meteyer, C.U., E.L. Buckles, D.S. Blehert, A.C. Hicks, D.E. Green, V. Shearn-Bochsler, N.J. Thomas, A. Gargas, and M.J. Behr. 2009. Histopathologic criteria to confirm white-nose syndrome in bats (cited as: Pathology criteria for confirming white-nose syndrome in bats). Journal of Veterinary Diagnostic Investigation 21:411–414.

Meteyer, C.U., D. Barber, and J.N. Mandl. 2012. Pathology in euthermic bats with white-nose syndrome suggests a natural manifestation of immune reconstitution inflammatory syndrome. Virulence 3:583–588.

Miller, G.S., Jr. 1897. Revision of the North American bats of the family Vespertilionidae. North American Fauna 13:1–40.

Miller-Butterworth, C.M., M.J. Vonhof, J. Rosenstern, G.G. Turner, and A.L. Russell. 2014. Genetic structure of little brown bats (*Myotis lucifugus*) corresponds with spread of white-nose syndrome among hibernacula. Journal of Heredity 105(3):354–364.

Mohr, C.E. 1976. The World of the Bat. J.B. Lippincott Company, United States of America.

Moritz, C. 2002. Strategies to protect biological diversity and the evolutionary processes that sustain it. Systematic Biology 51:238–254.

Mumford, R.E. and J.O. Whitaker. 1982. Mammals of Indiana. Indiana University Press, Bloomington, Indiana.

National Conference of State Legislatures (NCSL). 2021. State Renewable Portfolio Standards and Goals. Available here: https://www.ncsl.org/research/energy/renewable-portfolio-standards.aspx. Accessed online: May 12, 2021.

Neubaum, D.J. and J.L. Siemers. 2021. Bat swarming behavior among sites and its potential for spreading white-nose syndrome. Ecology 00(00):e03325. 10.1002/ecy.3325

Newman, B.A. 2020. Winter torpor and roosting ecology of tri-colored bats (*Perimyotis subflavus*) in trees and bridges. Master of Science in Wildlife and Fisheries Biology, Clemson University, Clemson, South Carolina. 90 pp.

Nicholson, C.P., R.D. Tankersley Jr., J.K. Fiedler, and N.S. Nicholas. 2005. Assessment and Prediction of bird and bat mortality at wind energy facilities in the southeastern United States. Tennessee Valley Authority, Knoxville, Tennessee. 65 pp.

Nicotra, A.B., E.A. Beever, A.L. Robertson, G.E. Hofmann, and J. O'Leary. 2015. Assessing the components of adaptive capacity to improve conservation and management efforts under global change. Conservation Biology 29:1268–1278.

North American Bat Monitoring Program (NABat 2020) Database v5.3.11 (Provisional Release): U.S. Geological Survey. Accessed 2020-11-18. NABat Request Number 10. https://doi.org/10.5066/P9UXA6CF.

North American Bat Monitoring Program (NABat 2021) Database v6.0.12 (Provisional Release): U.S. Geological Survey. Accessed 2021-02-10. NABat Request Number 12. https://doi.org/10.5066/P9UXA6CF.

Ofori, B., A. Stow, J. Baumgartner, and L. Beaumont. 2017. Influence of adaptive capacity on the outcome of climate change vulnerability assessment. Scientific Reports 7:1–12. [12979]. DOI: 10.1038/s41598-017-13245-y.

Oliver T., D.B. Roy, J.K. Hill, T. Brereton, and C.D. Thomas. 2010. Heterogeneous landscapes promote population stability. Ecology Letters 13:473–484.

O'Shea, T.J., P.M. Cryan, D.T.S. Hayman, R.K. Plowright, and D.G. Streicker. 2016. Mammal Review. 13 pp.

Paradiso, J.L. and A.M. Greenhall. 1967. Longevity records for American bats. American Midland Naturalist 78(1):251–252.

Parmesan, C., N. Ryrholm, C. Stefanescu, J.K. Hillk, C.D. Thomas, H. Descimon, B. Huntley, L. Kaila, J. Kullberg, T. Tammaru, W.J. Tennent, J.A. Thomas, and M. Warren. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature 399:579–583.

Perry, R.W. and R.E. Thill. 2007. Tree roosting by male and female eastern pipistrelles in a forested landscape. Journal of Mammalogy 88(4):974–981.

Perry, R. U.S. Forest Service (personal communication, April 29, 2021).

Ploskey, G.R. and J.A. Sealander. 1979. Lipid deposition and withdrawal before and during hibernation in *Pipistrellus subflavus* (Chiroptera: Vespertilionidae). The Southwestern Naturalist, pp.71–78.

Poissant, J.A. 2009. Roosting and social ecology of the tricolored bat, *Perimyotis subflavus*, in Nova Scotia. Master of Science in Applied Science, Saint Mary's University, Halifax, Nova Scotia. 76 pp.

Poissant, J.A., H.G. Broders, and G.M. Quinn. 2010. Use of lichen as a roosting substrate by *Perimyotis subflavus*, the tricolored bat, in Nova Scotia. Ecoscience 17(4):372–378.

Quinn, G.M. and H.G. Broders. 2007. Roosting and foraging ecology of eastern pipistrelle (*Perimyotis subflavus*) bats in SW Nova Scotia. Department of Biology, Saint Mary's University, Halifax, Nova Scotia. 34 pp.

Raesly, R.L. and J.E. Gates. 1987. Winter habitat selection by north temperate cave bats. American Midland Naturalist 1987:15–31.

Redford, K.H., G. Amato, J. Baillie, P. Beldomenico, E.L. Bennett, N. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G.M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to conserve a (vertebrate) species? BioScience 61:39–48.

Reeder, D.M., C.L. Frank, G.G. Turner, C.U. Meteyer, A. Kurta, E.R. Britzke, M.E. Vodzak, S.R. Darling, C.W. Stihler, A.C. Hicks, R. Jacob, L.E. Grieneisen, S.A. Brownlee, L.K. Muller, and D.S. Blehert. 2012. Frequent arousal from hibernation linked to severity of infection and mortality in bats with white-nose syndrome. Plos One 7:e38920. http://dx.doi.org/10.1371%2Fjournal.pone.0038920

Reichard, J. D. and T.H. Kunz. 2009. White-nose syndrome inflicts lasting injuries to the wings of little brown myotis (*Myotis lucifugus*). Acta Chiropterologica 11:457–464. https://doi.org/10.3161/150811009X485684

Reid, F.A. 1997. A Field Guide to the Mammals of Central America and Southeast Mexico. Oxford University Press, Inc., New York, New York.

Reynolds, R.J., K.E. Powers, W. Orndorff, W.M. Ford, and C.S. Hobson. 2016. Changes in rates of capture and demographics of *Myotis septentrionalis* (northern long-eared bat) in western Virginia before and after onset of white-nose syndrome. Northeastern Naturalist 23(2):195–204.

Ruckelhaus M., P. McElhany, and M.J. Ford. 2002. Recovering species of conservation concern: Are populations expendable? Pp. 305–329 in P. Kareiva and S. Levin, editors. The Importance of Species: Perspectives on Expendability and Triage. Princeton, NJ: Princeton University Press.

Samoray, S.T., S.N. Cotham, and M.W. Gumbert. 2019. Spring migration behavior of a *Perimyotis subflavus* (tri-colored bat) from Tennessee. Southeastern Naturalist 18(3):16–20.

Sandel, J.K., G.R. Benatar, K.M. Burke, C.W. Walker, T.E. Lacher, and R.L. Honeycutt. 2001. Use and selection of winter hibernacula by the eastern pipistrelle (*Pipistrellus subflavus*) in Texas. Journal of Mammalogy 82(1):173–178.

Sasse, D.B., D.A. Saugey, and D.R. England. 2011. Winter roosting behavior of Rafinesque's big-eared bat in southwestern Arkansas. Pp. 123–128 in Loeb, S.C.; Lacki, M.J.; Miller, D.A. (editors). Conservation and management of eastern big-eared bats: a symposium. Gen. Tech. Rep. SRS-145. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 157 pp.

Schirmacher, M.S., A. Prichard, T. Mabee, and C.D. Hein. 2018. Evaluating a Novel Approach to Optimize Operational Minimization to Reduce Bat Fatalities at the Pinnacle Wind Farm, Mineral County, West Virginia, 2015. An annual report submitted to NRG Energy and the Bats and Wind Energy Cooperative. Bat Conservation International. Austin, Texas, USA.

Sealander, J.A. and H. Young. 1955. Preliminary Observations on the Cave Bats of Arkansas. Journal of the Arkansas Academy of Science 7(10).

Sgro C.M., A.J. Lowe, and A.A. Hoffmann. 2011. Building evolutionary resilience for conserving biodiversity under climate change. Evolutionary Applications 4:326–337.

Shaffer, M.L. and B.A. Stein. 2000. Safeguarding our precious heritage. Pp. 301–321 in Stein B.A., L.S. Kutner, and J.S. Adams, editors. Precious Heritage: The Status of Biodiversity in the United States. New York: Oxford University Press.

Sherwin, H.A., W.I. Montgomery, and M.G. Lundy. 2013. The impact and implications of climate change for bats. Mammal Review. 18 pp.

Silvis, A., R.W. Perry, and W.M. Ford. 2016. Relationships of three species of bats impacted by White-Nose Syndrome to forest condition and management. Gen. Tech. Rep. SRS–214. Asheville, North Carolina: U.S. Department of Agriculture Forest Service, Southern Research Station. 48 p.

Slider, R.M. and A. Kurta. 2011. Surge tunnels in quarries as potential hibernacula for bats. Northeastern Naturalist 18(3):378–381.

Smith, C. Minnesota Department of Transportation (personal communication, September 8, 2020).

Smith, L.M., T.J. Doonan, and J.A. Gore. 2021. Declines of tri-colored bats wintering in Florida caves; p. 21. In *MEETING PROGRAM 26th Annual Meeting of the Southeastern Bat Diversity Network and the 31st Annual Colloquium on the Conservation of Mammals in the Southeastern U.S.* February 25, 2021.

Spanjer, G.R. and M.B. Fenton. 2005. Behavioral responses of bats to gates at caves and mines. Wildlife Society Bulletin 33:1101–1112.

Spielman, D., B.W. Brook, and R. Frankham. 2004. Most species are not driven to extinction before genetic factors impact them. Proceedings of the National Academy of Sciences 101(42):15261–15264.

Stevens, R.D., C.J. Garcia, M.A. Madden, B.B. Gregory, and R.W. Perry. 2020. Seasonal Changes in the Active Bat Community of the Kisatchie National Forest, Louisiana. Southeastern Naturalist 19(3):524–536.

Stratton, C., and K.M. Irvine. 2022. Summertime Analysis Statistical Report for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter B in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Teplitsky C., M.R. Robinson, J. Merilä. 2014. Evolutionary potential and constraints in wild populations. Pp. 190–208 in Charmantier A, D. Garant, and L.E.B. Kruuk, editors. Quantitative Genetics in the Wild. Oxford, UK: Oxford University Press.

Thames, D.B. 2020. Summer Foraging Range and Diurnal Roost Selection of Tri-colored bats, *Perimyotis subflavus*. Master's Thesis University of Tennessee Knoxville.

Thurman, L, B. Stein, E. Beever, W. Foden, S. Geange, N. Green, J. Gross, D. Lawrence, O. LeDee, J. O’Leary, J. Olden, L. Thompson, and B. Young. 2020. Persist in place or shift in space? Evaluating the adaptive capacity of species to climate change. Frontiers in Ecology and the Environment 18(9):520–528.

Turcios-Casco, M.A., C.J Mejía-Suazo, D.J. Ordoñez Bautista, and H.D. Ávila-Palma. 2020. Noteworthy records of the Geoffroy’s tailless bat and the Eastern Pipistrelle in Copán, western Honduras (Mammalia Chiroptera). Biodiversity Journal 11(2):527–534.

Turcios-Casco, M.A., R.K. LaVal, D.E. Wilson, and H.D. Ávila-Palma. 2021. Bats in time: Historical and Geographic Distribution in Honduras. Occasional Papers, Museum of Texas Tech University, No. 375:1–22.

Turner, G.G. and D.M. Reeder. 2009. Update of white-nose syndrome in bats, September 2009. Bat Research News 50(3):47–53.

Turner, G.G., D.M. Reeder, and J.T.H. Coleman. 2011. A five-year assessment of mortality and geographic spread of white-nose syndrome in North American bats and a look to the future. Bat Research News 52(2):13–27.

Udell, B.J., B.R. Straw, J. Szymanski, M. Seymour, J. Wong, A.M. Wiens, B.E. Reichert, J.E. Diffendorfer, L. Kramer, and Z. Ancona. 2022. Spatially explicit method of predicting bat wind take at wind facilities and allocating it among winter hibernacula populations. Appendix E-1 in Straw, B.R., J.A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological

Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

U.S. Fish and Wildlife Service (USFWS). 2007. Indiana Bat (*Myotis sodalis*) Draft Recovery Plan: First Revision. U.S. Fish and Wildlife Service, Fort Snelling, Minnesota, 258 pp.

USFWS. 2019. Final Environmental Impact Statement for Proposed Habitat Conservation Plan and Incidental Take Permit. MidAmerican Energy Company Wind Energy Facility, Iowa. September 6, 2019. 253 pp. Available at: https://fws.gov/midwest/rockisland/te/pdf/Final%20EIS%20for%20the%20MidAmerican%20Energy%20Company%20HCP%20and%20ITP.pdf

U.S. Forest Service (USFS). 2012. Future of America's forest and rangelands: Forest Service 2010 resources planning act assessment. Gen. Tech. Rep. WO-87. Washington, DC. 198 pp.

USFS. 2016. Future of America's Forests and Rangelands: Update to the 2010 Resources Planning Act Assessment. Gen. Tech. Report WO-GTR-94. Washington, DC. 250 p.

Veilleux, J.P., J.O Whitaker, and S.L. Veilleux. 2003. Tree-roosting ecology of reproductive female eastern pipistrelles, *Pipistrellus subflavus*, in Indiana. Journal of Mammalogy 84(3):1068–1075.

Veilleux, J.P. and S.L. Veilleux. 2004a. Intra-annual and interannual fidelity to summer roost areas by female eastern pipistrelles, *Pipistrellus subflavus*. The American Midland Naturalist 152(1):196–200.

Veilleux, J.P. and S.L. Veilleux. 2004b. Colonies and reproductive patterns of tree-roosting female eastern pipistrelle bats in Indiana. Proceedings of the Indiana Academy of Science 113(1):60–65.

Verant, M., C. Meteyer, J. Speakman, P. Cryan, J. Lorch, and D.S. Blehert. 2014. White-nose syndrome initiates a cascade of physiologic disturbances in the hibernating bat host. BMC Physiology, 14(1):10.

Vincent, E.A. and J.O. Whitaker. 2007. Hibernation of the eastern pipistrelle *Perimyotis subflavus*, in an abandoned mine in Vermillion County, Indiana, with some information on *Myotis lucifugus*. Proceedings of the Indiana Academy of Science 116(1):58–65.

Voigt, C.C., K. Schneeberger, S.L. Voigt-Heucke, and D. Lewanzik. 2011. Rain increases the energy cost of bat flight. Biological Letters 7:793–795. doi:10.1098/rsbl.2011.0313

Vose, R.S., D.R. Easterling, K.E. Kunkel, A.N. LeGrande, and M.F. Wehner. 2017. Temperature changes in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J.

Dokken, B.C. Stewart, and T.K. Maycock (editors)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 185–206, doi: 10.7930/J0N29V45.

Walley, H.D. and W.L. Jarvis. 1972. Longevity record for *Pipistrellus subflavus*. Transactions of the Illinois State Academy of Science 64:305.

Warnecke, L., J.M. Turner, T.K. Bollinger, V. Misra, P.M. Cryan, D.S. Blehert, G. Wibbelt, and C.K.R. Willis. 2013. Pathophysiology of white-nose syndrome in bats: a mechanistic model linking wing damage to mortality. Biology Letters 9:20130177.

Webb, D.R. and J.R. King. 1984. Effects of wetting on insulation of bird and mammal coats. Journal of Thermal Biology 9(3)189–191.

Whitaker J.O. 1972. Food habits of bats from Indiana. Canadian Journal of Zoology 50(6):877–883.

Whitaker J.O. 1998. Life history and roost switching in six summer colonies of eastern pipistrelles in buildings. Journal of Mammalogy 79(2):651–659.

Whitby, M., B.J. Udell, A.M. Wiens, T. Cheng, W. Frick, B.E. Reichert, and J.D. Reichard. 2022. Summer Mobile Acoustic Transect Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter C in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

Wiens, A.M., J. Szymanski, B.J. Udell, and W. E. Thogmartin. 2022. Winter Colony Count Data Assessment and Future Scenarios for the Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter E in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

Wilson, R.J., D. Gutierrez, J. Guitierrez, and V.J. Monserrat. 2007. An elevational shift in butterfly species richness and composition accompanying recent climate change. Global Change Biology 13:1873–1887. doi: 10.1111/j.1365-2486.2007.01418.x

Wiser, R., J. Rand, J. Seel, P. Beiter, E. Baker, E. Lantz, and P. Gilman. 2021. Expert elicitation survey predicts 37% to 49% declines in wind energy costs by 2050. Nat. Energy. Available at: https://doi.org/10.1038/s41560-021-00810-z

Zackay, A. 2007. Random genetic drift and gene fixation. Available here: https://www.metabolic-economics.de/pages/seminar_theoretische_biologie_2007/ausarbeitungen/zackay.pdf. Accessed online: June 21, 2018.

APPENDICES

1. **Key Uncertainties, Wind Energy Mortality Sensitivity Analyses, and State-of-the-Knowledge**
    - A. Key Uncertainties
    - B. Wind Energy Mortality Sensitivity Analysis
    - C. State-of-the-Knowledge
2. **Supplemental Methodology**
    - A. Analytical Framework
    - B. Adaptive Capacity Analysis
3. **Supplemental Results**
    - A. Historical Condition
    - B. Current Condition
    - C. Future Condition
    - D. Qualitative/Comparative Threat Analysis
4. **Supplemental Threat and Future Scenario Information**
    - A. WNS
    - B. Wind
    - C. Climate Change
    - D. Habitat Loss
5. **Supplemental Future Scenarios Descriptions**

Appendix 1: Key Uncertainties, Wind Energy Mortality Sensitivity Analyses, and State-of-the-Knowledge

A. Key Uncertainties

Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios, adding environmental stochasticity to our model, and making reasonable assumptions. The key uncertainties are listed in Table A-1.1 and are described below.

Table A-1.1. A list of key uncertainties addressed in the analysis.

Current Abundance and Trend	White-nose Syndrome Impacts	Wind Energy Related Mortality	Climate Change and Habitat Loss
Imperfect abundance data over time and space	*Pd* rate of spread*	Future wind energy capacity*	Response to climate change
	WNS impact schedule	Fatality rates	Response to habitat loss
	Duration of WNS impact*	Fatality risk over time and space	
	Bat response where WNS not yet arrived		
	Unknown hibernacula		

*Uncertainties are addressed directly in our high and low impact future scenarios (see Appendix 5).

Abundance and Population Trend

We do not have **perfect knowledge of current colony abundance and population trend** because hibernacula are not surveyed every year nor concurrently, and there are likely many undocumented hibernacula. Furthermore, TCB can be hidden in inaccessible locations within surveyed hibernacula and may be difficult to identify accurately. We address this uncertainty by using predictive models developed by Cheng et al. (2021, entire) and Wiens et al. (2022, pp. 231–247a) to predict current abundance and population growth rate (trend) for each known hibernaculum. Cheng et al. (2021, entire) explained that using a statistical model rather than inferring from data summaries is preferred because it can account for site-to-site variation, year-to-year variation, and survey effort, thereby allowing evaluation of the main effects of counts over time and the impacts of WNS on counts. Further, statistical methods allow for objectively quantifying the relationships between variables while also quantifying the amount of uncertainty around those results. We summarized the state-of-the-knowledge (raw data summaries) that inform these statistical methods in Appendix 1-B.

The statistical models are constructed from the raw data available (i.e., 6,341 TCB winter observations). These data represent the core of the species' known historical and current abundances, and thus are representative of the species' overall condition. Further, while the imminent threats (i.e., WNS, wind, habitat loss, and climate change) may vary temporally, the spatial distribution and overall severity of these threats are not likely to differ markedly (see WNS impacts assumptions below). Coupling this assumption with information concerning the narrow range of optimal conditions for hibernation, we believe these data provide the best available and reliable dataset to assess the current and future viability of the species.

Regarding bats in general, estimating population abundance and trends is challenging due to their cryptic nature, wide ranging habits, and variable detectability. A variety of methods have been developed and continue to be improved to fulfil this important information need, including winter and summer colony counts, mist-netting, acoustic monitoring, and mark-recapture studies. However, these efforts are often limited in scope or have been inconsistently applied across species' ranges. For several federally protected hibernating bats (e.g., Indiana bat, Virginia big-eared bat (*Corynorhinus townsendii virginianus*), and gray bat (*Myotis grisescens*), successful population monitoring has been achieved through coordinated survey efforts at winter and summer roosts in caves. Fortunately, non-listed species have benefitted from these coordinated survey efforts and monitoring expertise where they overlap with either state or federally listed species. For this reason, estimates of overwintering colony abundance of TCB are available through a substantial portion of their range over recent decades. Winter survey efforts for TCB and other hibernating species also increased when concerns about WNS were first raised in North America over 10 years ago. Other sources of data, to date, are more sporadic spatially and temporally but are still useful to inform population status.

We also do not have perfect knowledge of every hibernacula throughout the range of TCB (**unknown hibernacula**).TCB hibernate in more caves (or cave-like subterranean habitats) than any other cave-hibernating bat species in eastern North America (Sealander and Young 1955, pp. 23–24; Barbour and Davis 1969, p. 117; Brack et al. 2003, p. 65). Almost every cave in Indiana has contained at least one TCB (Mumford and Whitaker 1982, pp. 167–168); and small numbers of TCB have likely occupied most of Missouri's 6,400 caves (Perry 2021, pers. comm.). Hibernating TCB do not typically form large clusters and most commonly roost singly (see *Individual-level Ecology and Needs*); therefore, many TCB (if not the majority) may be distributed in numerous small (and often unidentified) hibernacula during winter (Johnson 2021, pers. comm.; Perry 2021, pers. comm.).

Despite the expectation that many hibernating TCB remain unobserved during winter, abundance estimates based on winter counts represent a sound estimate of the site-specific abundances, relative abundances, or at least population trends. Importantly, although these surveys do not produce a true census of the populations, they provide an estimate (or index) of abundance during winter when both sexes are roosting together. Summer roost counts are possible but much less feasible for TCB due to their roost preferences and frequent roost switching. Mist-netting efforts to estimate capture per unit effort is another method for assessing trends, but these efforts are labor intensive and not commonly available. Finally, acoustic monitoring can be used to estimate occupancy or indices of abundance that are useful to estimate relative changes in populations but are very difficult to interpret as estimates of abundance. For these reasons, winter colony counts produce the most direct, representative, and feasible method for estimating TCB abundance, even if these data only represent minimum estimates of abundance.

Furthermore, WNS is typically detected and causes mortality either during winter or in spring after sick bats emerge from hibernation. Thus, estimating the impacts of this disease is best achieved by evaluating changes in winter colonies, where possible, in response to the arrival of the fungal pathogen. This approach allows for analyses that specify the year of arrival of the fungal pathogen and subsequent changes in population sizes. While winter counts provide the most direct method for estimating the impacts of WNS, additional data streams are used to verify the patterns observed in winter. Analyses of mobile acoustic monitoring and capture efforts provide estimates of changes in relative abundance, while stationary acoustic monitoring produces indices of bat activity. All of these together are also used in occupancy modelling to determine changes in occurrence on the landscape over time. While none of these methods provides a perfect estimate of population abundance, together they improve our understanding of TCB status.

White-nose Syndrome Impacts

To capture the uncertainty in the **rate of spread** of *Pd* we used two different *Pd* occurrence models, a faster spread rate (*Pd* occurrence model 1, Wiens et al. 2022, pp. 226–229) based on spread rates observed and annual changes in the occurrence of *Pd* and a slower spread rate (*Pd* occurrence model 2, Hefley et al. 2020, entire) that incorporates historic occurrence and multiple habitat covariates (Appendices 2A and 5). Both models rely on the same WNS surveillance dataset but each model performs differently in different geographic regions of North America based on the models' parameters. Thus, these two predictions provide a plausible range of the timing of *Pd* spread into the future.

Although we have empirical information on population-level impacts associated with WNS disease progression (on average, 97% decline by the endemic stage, Cheng et al. 2021, entire), there is variability among sites. We identified sites that trended differently (i.e., better) than most and assumed they do not experience further WNS impacts. Wiens et al. (2022, pp. 231–235) used random draws from the impact distribution for each year (Appendix 2A). For all remaining sites, we assumed they would follow the empirically derived yearly impacts schedule.

Another source of uncertainty is the **duration of WNS impacts**. We captured the full breadth of uncertainty in our future scenarios. For all scenarios, WNS impacts ameliorate 6 years after the arrival of *Pd*, forming an endemic stage (see Appendix 2A). Under the low impact scenario, we assumed a 9-year endemic stage and thus yielding a 15-year WNS impacts duration in total. This is the shortest conceivable timeframe based on our analysis of the data available. Under the high impact scenario, we assumed a 34-year endemic stage, thus yielding a 40-year WNS impacts duration in total (Appendix 5). Figure A-1A1 shows results assuming no further WNS impacts beginning in 2020, a 25-year impacts duration, and a 40-year impacts duration.

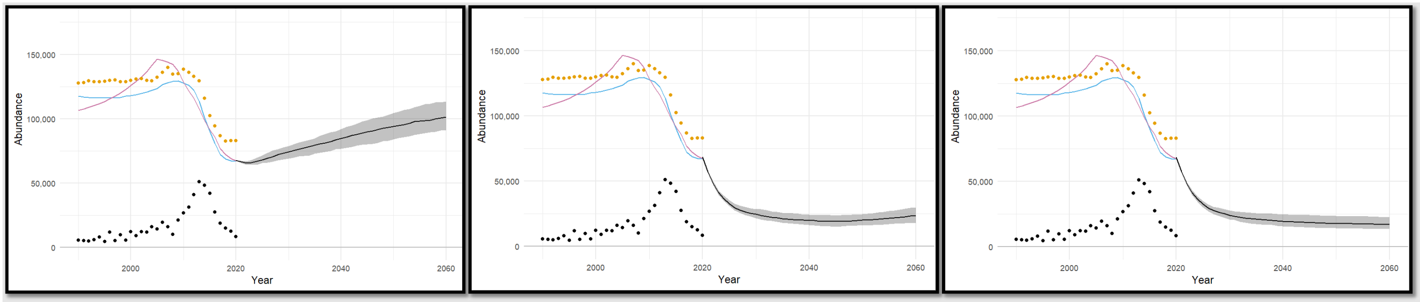

Figure A-1A1. Projected median rangewide abundance (median [black line], 90% CI [gray shading]) over time under no future WNS impacts (left), a 25-year impacts duration (middle), and a 40-year impacts duration (right). Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status and trend model informed by Pd occurrence model 2 (pink line).

Other sources of uncertainty include the **species' response** to WNS in areas where *WNS* has not yet been detected and our imperfect knowledge of every hibernacula throughout the range of the species (**unknown hibernacula**). This is particularly important in the southern portions of the range where TCB commonly overwinter in small colonies or individually, roost in numerous road-associated culverts, and may be active through much of the winter.

Where disease dynamics of WNS have been observed (primarily, but not solely in the eastern half of North America and in cave-like hibernacula), very few TCB colonies have avoided severe impacts of the disease. A variety of site characteristics including colony size, temperature, and humidity may explain some of the variability that is observed in the degree of impact caused by WNS. Wilder et al. (2011) predicted that larger colonies will experience impacts of WNS sooner than smaller colonies. Further, Langwig et al. (2012, p. 6) determined that smaller colonies of TCB may experience less severe impacts than larger colonies during the initial stages of the disease. TCB colonies approached stabilization at low numbers (~ 6 bats) only after significant declines in larger colonies and not as a result of initial small colonies avoiding impacts (Langwig et al. 2012, p. 4). Similarly, Frick et al. (2015, p. 6) found that TCB colonies with smaller pre-WNS counts had a higher risk of local extinction due to WNS than larger ones.

Environmental conditions may also influence impacts of disease. While it has been determined that colder roosts may reduce WNS infections, mortality from WNS has been documented at a wide range of temperatures, including sites with winter temperature approaching 0°C (Langwig et al. 2012, p. 6). Low humidity conditions may also lessen the severity of infection, at least for some species. For example, Indiana bats in drier hibernacula have shown to have less severe impacts from WNS, but this pattern was not observed in TCB (Langwig et al. 2012, p. 6).

Physiological demands of hibernation limit the ranges of temperature and humidity in which bats can hibernate successfully, although these limits or preferences differ among species. Hibernacula temperatures that are too low present a risk of freezing or raise the energetic cost of torpor. Similarly, hibernacula that are too dry lead to dehydration or frequent arousal from torpor that will consume limited fat reserves. Thus, although these factors may delay or reduce the impacts of WNS, none of them would prevent the arrival of *Pd* or avoid impacts of WNS altogether. Because their winter roosts must be cold and humid to allow for successful hibernation and these conditions are also conducive to growth of *Pd*, it is valid to presume WNS impacts will be similar throughout the portions of the species' range where bats hibernate for extended periods, regardless of whether these hibernacula are unknown or human inaccessible.

Wind Energy Related Mortality

We do not know the **future build-out of wind energy capacity** in the U.S. and Canada. We relied on the National Renewable Energy Laboratory's (NREL) (Cole et al. 2020, entire) and Canadian Energy Regulator's (CER) (CER 2020, entire) projections for the U.S. and Canada, respectively. To capture the uncertainty associated with these projections, we incorporated lower and upper bound capacity projections into our future scenarios. Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario*. Our high impact scenario (*i.e.*, higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Chapter 4 and Appendix 5). These build-out scenarios provide reasonable bounds for future expectation of wind capacity in both the U.S. and Canada.

Fatality Rates vary across species, range, and seasons. We used regional specific data garnered from postconstruction monitoring efforts. We obtained nearly 300 reports spanning 20 states and 4 USFWS Regions. We calculated the mean fatality rate for the species within each USFWS Region using currently accepted methods to account for spatial variability (see Appendix 2). We also are uncertain about how **fatality risk varies over time and space**. Although it is logical to assume fatality risk declines with decreasing abundance, the functional relationship is unknown. We evaluated fatality rates pre- and post-WNS arrival to discern a relationship between abundance and fatality risk. Where applicable, we applied pre- and post *Pd* fatality rates to account for the uncertainty in fatality risk as abundance changes over time (see Appendix 2). Additionally, we are uncertain of where bats killed at wind facilities originate. To address this uncertainty, we relied on the analysis completed by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) created a distance decay function to allocate total wind mortality per 11x11-km NREL grid cell among hibernacula within the known average maximum migration distance, relative to the size of the hibernating populations as well as the distance from the grid cell centroids (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). However, the analysis did not account for the possibility that some bats may originate from additional unknown hibernacula within the maximum recorded migration distance, or that bats may be migrating farther than previously documented. To look at how this latter uncertainty may affect the results, we ran a scenario in which wind mortality is 50% of what is projected under the high capacity scenario. The additive effect of wind energy mortality is discernible as seen when comparing a no wind to a wind scenario (Figure A-1B2); although from a viability perspective, the results do not appear sensitive to the range of uncertainty in future mortality levels (i.e., no marked changes in the overall trend in abundance).

Climate change

As we detail further in Chapter 4 and Appendix 4, both habitat loss and climate change are pervasive across the species' range and severity of population level declines are assumed to be slight (recognizing varying impacts by population). Thus, we believe overall climate change impacts are currently low. While there is uncertainty about the magnitude of future temperature increases and any associated changes in precipitation (e.g., regional changes, rate and intensity of extreme weather events), we have high confidence in the precipitation and temperature changes observed to date, and that minimal projected temperature increases (2.2 degrees F, relative to baseline) will occur. Similarly, we have high certainty in observed species responses to changes in temperature and precipitation (which vary geographically). However, we have less certainty about species responses that have not been observed, such as: death of individuals or alteration of hibernacula use due to increased risk of flooding from sea level rise or extreme weather events; reduced reproduction or survival due to increased habitat loss in wildfire prone areas; changes in phenology of bats and their prey; and changes in bat distribution. Lastly, we have uncertainty about possible beneficial impacts from climate change in portions of TCB's range. While possible, beneficial impacts (e.g., warmer temperatures may lead to shorter hibernation periods, which in turn may decrease the *Pd* exposure duration and thus reduce impacts) are more speculative, at least relative to the observed negative impacts reported in the literature. For this reason, our assessment of effects from climate change likely underestimates risk to the species.

Habitat Loss

We have high confidence of prior impacts to winter hibernacula and hibernating bats. We have high confidence that changes in vegetation cover types occur throughout TCB's range. We also have high confidence that these changes in landcover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality (during active season tree removal). Despite this knowledge, we have uncertainty about how much forest removal must occur within a home range before impacts associated with winter tree removal are realized. We also have imperfect knowledge of where roosts (summer and winter) for TCB occur. Therefore, we have uncertainty about which colonies (summer and winter) are at greatest risk of impacts associated with habitat loss.

B. Wind Energy Mortality Sensitivity Analysis

To discern the sensitivity of our results to uncertainty regarding wind energy related mortality, we ran various mortality scenarios. We compared four scenarios: 1) no wind energy related mortality, 2) current predicted mortality, 3) 50% of mortality corresponding to the future high impact scenario, and 4) full projected level of mortality corresponding to the high impact scenario. Clearly, WNS is the driving force in the future trajectory of the species (see Figure A-1A1, comparing no WNS impacts to WNS impact scenarios), thus it is not surprising that the general trend in abundance is unaffected by wind energy mortality (Figure A-1B1). The additive effect of wind energy mortality is discernible as seen when controlling for WNS impacts and comparing no wind to wind scenarios under current wind conditions but not under future wind conditions (Figure A-1B2, see bar 1 vs 2 for current conditions and bar 3 vs 4 and 5 for future).

A likely explanation for this is as *Pd* spreads across the range over time, the results are less sensitive among the wind mortality scenarios because WNS impacts are severe and dominate the dynamics (Figure A-1B2, bars 3–5).

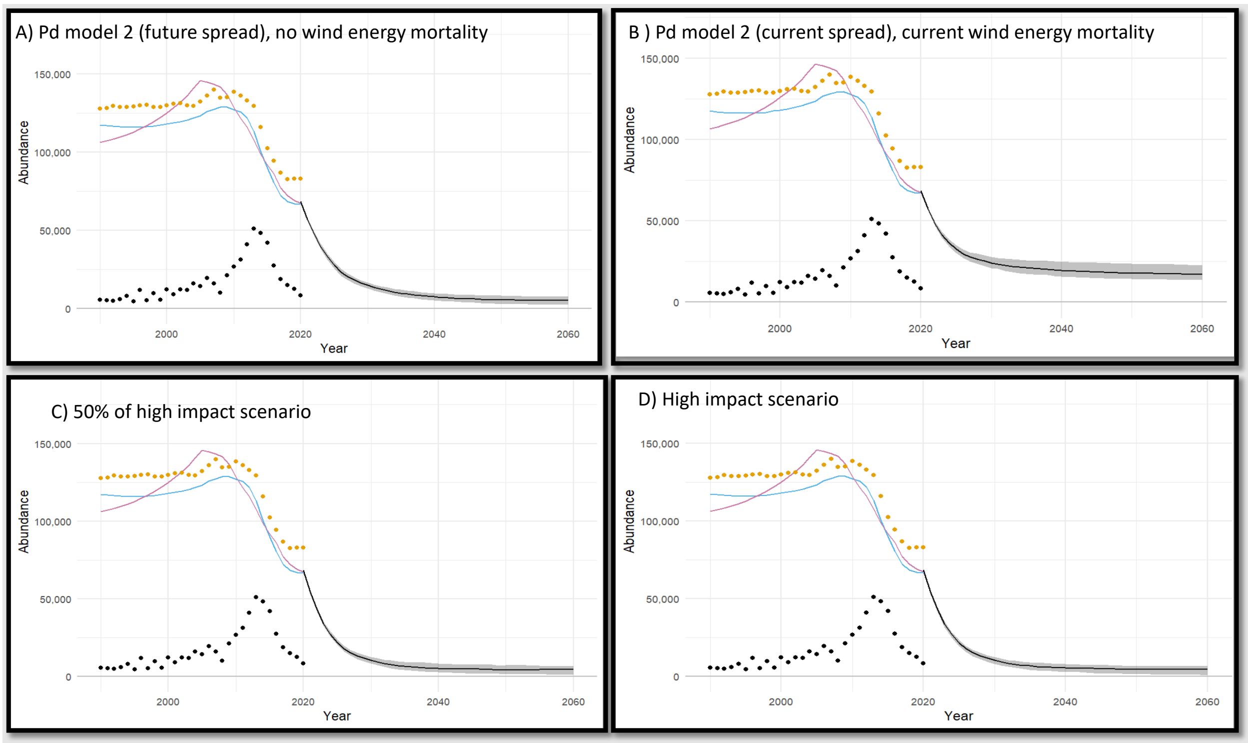

Figure A-1B1. TCB projected abundance under various wind mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) 50% of the future wind energy mortality under the high impact scenario, and (D) high impact scenario mortality. Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status and trend model informed by Pd occurrence model 2 (pink line).

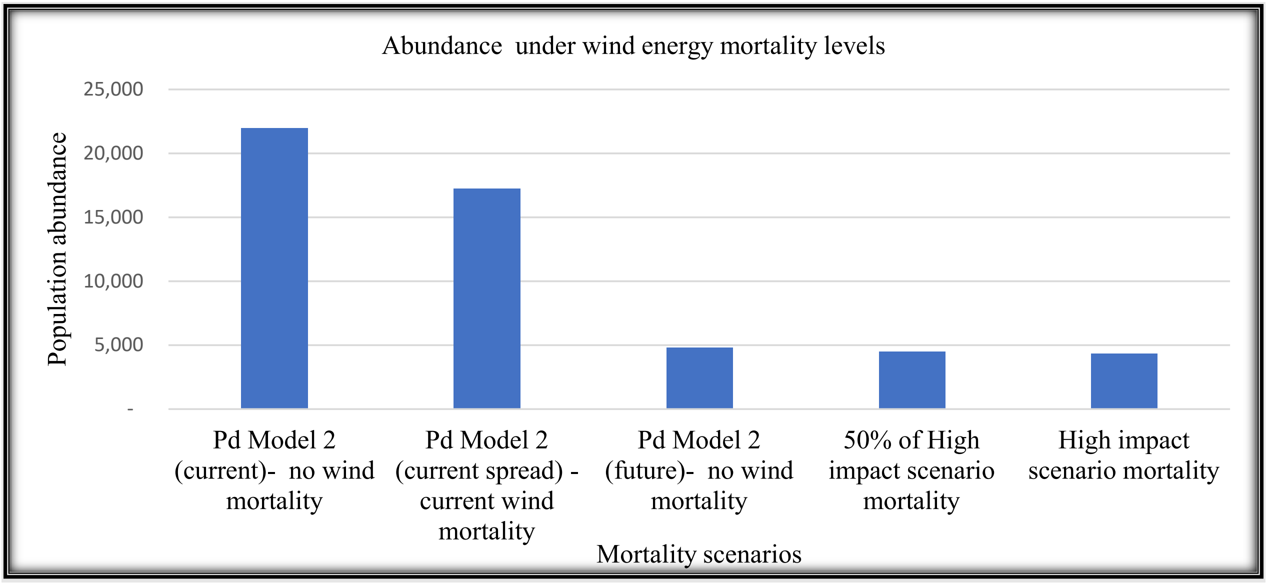

Figure A-1B2. TCB projected median 2060 abundance under five wind energy related mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) future mortality under low impact scenario, (D) 50% of the future mortality under the high impact scenario, and (E) future mortality under the high impact scenario

C. State-of-the-Knowledge

For reasons articulated in subsection A above, we relied upon statistical methods rather than raw data alone to assess the species' current status. We summarize the data underlying these methods here.

- We have 6,341 records from 1,951 hibernacula (58% of the sites are from the Northern RPU).
- Based on these raw data:
  - Number of hibernacula with "Last observed = 0": 165 (1990–2020), 3 (2006–2009), 51 (2010–2015), 108 (2016–2020); the ratio (proportion) of extirpated to extant sites increased since WNS discovered in 2006 (Figure A-1C1)
  - Of the 1,786 potentially extant sites, 43 to 71% have uncertain status (768–1,267 sites do not have ≥ 1 record from 2017–2020 and 2019–2020, respectively)

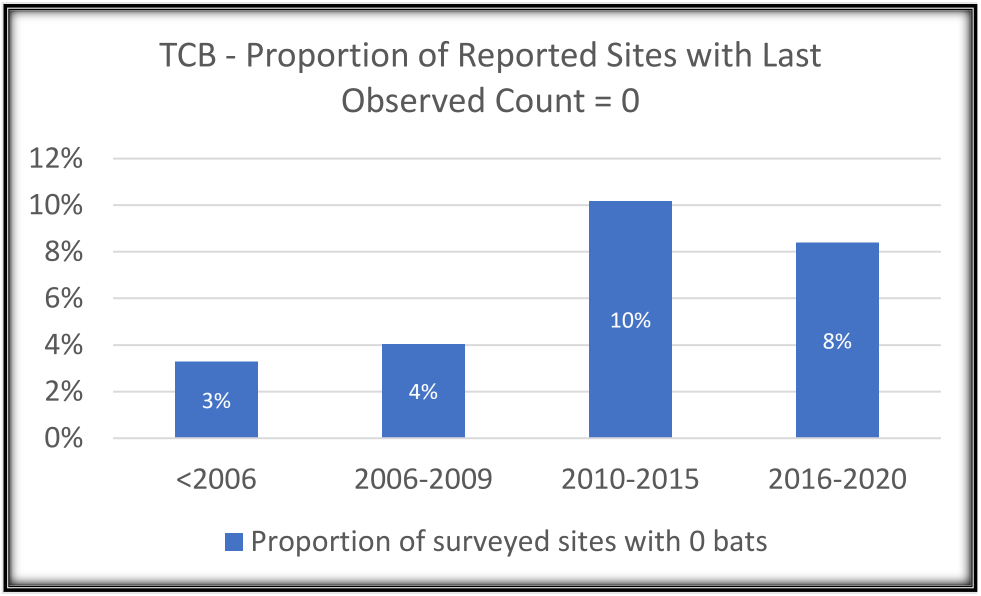

Figure A-1C1. The proportion of sites reported to NABat with 0 as the "last observed count." The proportion is number of hibernacula with 0 counts divided by the total number of hibernacula surveyed.

- As of May 2021, 580 counties across 40 states and 7 Canadian provinces have presumed or confirmed *Pd*/WNS (485 are confirmed WNS/*Pd*) (www.whitenosesyndrome.org, accessed May 13, 2021). WNS/*Pd* suspected/confirmed from Nova Scotia southward to South Carolina, westward to Texas, New Mexico, Wyoming, Montana, and Washington (Figure A-1C2).

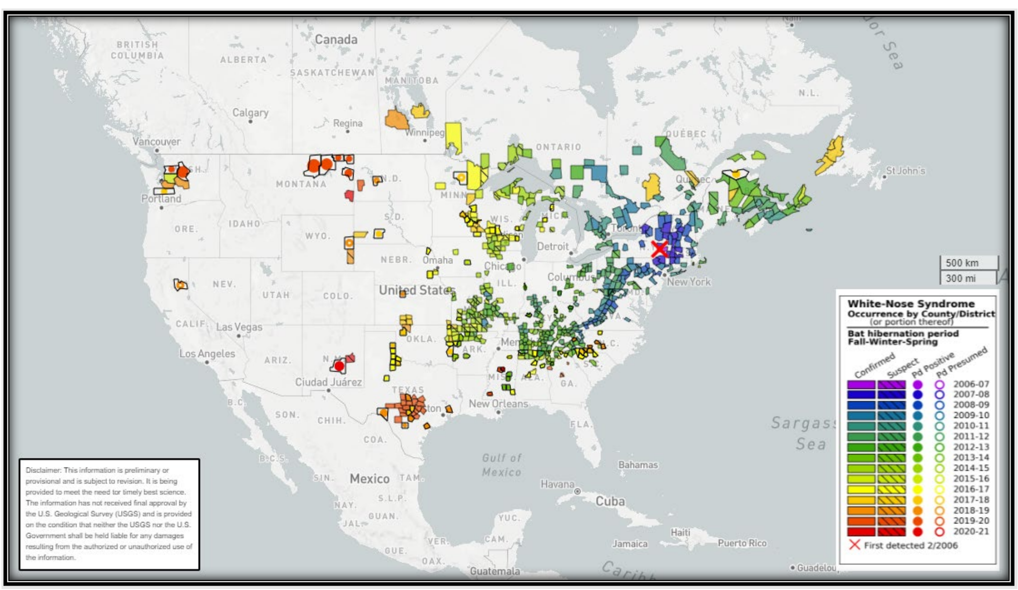

Figure A-1C2. WNS occurrence as of 5/12/2021 (www.whitenosesyndrome.org, accessed May 13, 2021).

- The number of TCB hibernacula with suspected or confirmed WNS is not available; WNS has been confirmed in every RPU. Most of these are from northern portion of the range, and data are scant in non-cave hibernacula in the southern portion of the range.
    - As of May 2021, there are 148 TCB events. Events are winter or summer sites with suspected/confirmed WNS/*Pd* reported on the species of interest (i.e., a species event is recorded only when the species has *Pd*/WNS, even if the WNS/*Pd* confirmed/suspected on other species or the site, www.whitenosesyndrome.org, accessed May 13, 2021).
- Where WNS is present, severe declines have occurred, except in a few (2%) hibernacula. On average, TCB colonies declined by mean 93% (95% CI 90-100%) by the endemic stage of WNS progression (Cheng et al. 2021, p. 7).
- Declines are discernible in summer data as well. Data availability vary among the data type (mobile acoustic, stationary acoustic, and mist-net capture data), however, we incorporated all available data into the analyses.
    - Using mobile transect acoustic data, Whitby et al. (2022, entire) found that relative abundance declined 38% (Eastern RPU) to 86% (Northern RPU) from 2009 to 2019.
    - Using mist-net capture data, Deeley and Ford (2022, entire) found significant decreases in mean capture rate from 1999 to 2019. Estimates derived from their data found 12% (Southern RPU) to 19% (Eastern RPU) declines in mean capture rates post-WNS arrival.
    - Using all three data types (mobile transect acoustic, stationary acoustic, and mist-net capture), Stratton and Irvine (2022, entire) looked at changes in probability of occupancy across the range of the species. Their results showed a decline in TCB occupancy across all RPUs (Stratton and Irvine 2022, entire). Estimates derived from their results found declines in the probability of occupancy ranging from 17% (Eastern and Northern RPUs) and 37% (Southern RPU) from 2010 to 2019.

Appendix 2: Supplemental Methodology

A: Analytical Framework

Below we describe our methods for assessing a species viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and 5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability (Figure A-2A1). Because of the difficulty of delineating individual populations for bat species, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. The terms populations, winter colonies, and hibernacula are used interchangeably.

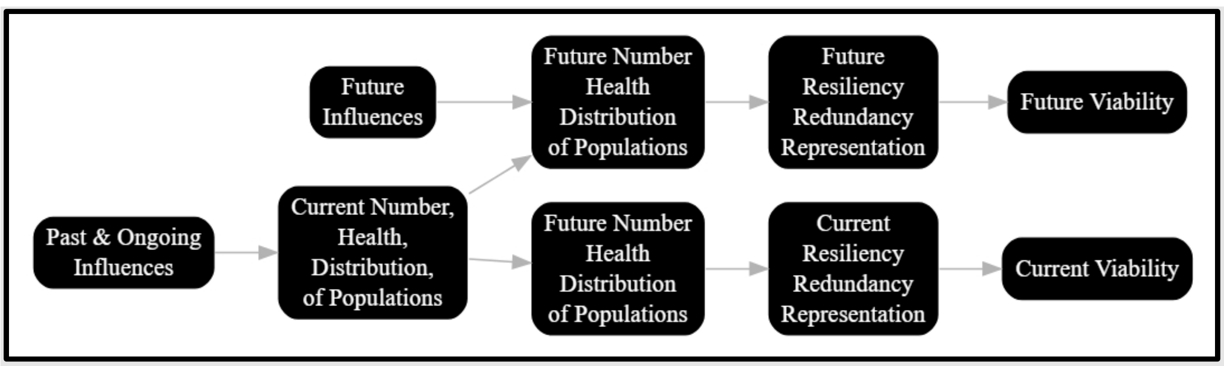

Figure A-2A1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, state and other) across the range to garner summer (capture data and stationary and mobile acoustic) and winter occurrence (hibernacula counts) data. Most of these data are maintained in the North American Bat Monitoring Program (NABat) database[7], unless otherwise requested by the data contributor or the data was not provided in a format that could be accepted by the database. These efforts yielded thousands of records across the range (Figure A-2A2) and one of the largest bat data repositories we are aware of. Hibernacula counts were available for much of the range of TCB, although occurrence information is extremely scarce for the species in Mexico and Central America (Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10). Consistent with the species' biology, we assumed that TCB employs hibernation in cold, humid roosts even when these roosting locations are not observed by data collectors. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (winter hibernacula surveys; NABat 2021).

[7] Colony count data from North American Bat Monitoring Program Database v5.4.3: U.S. Geological Survey. Accessed 2021-02-10. NABat Request Number 12. batmonitoring.org / https://sciencebase.usgs.gov/nabat/#/home

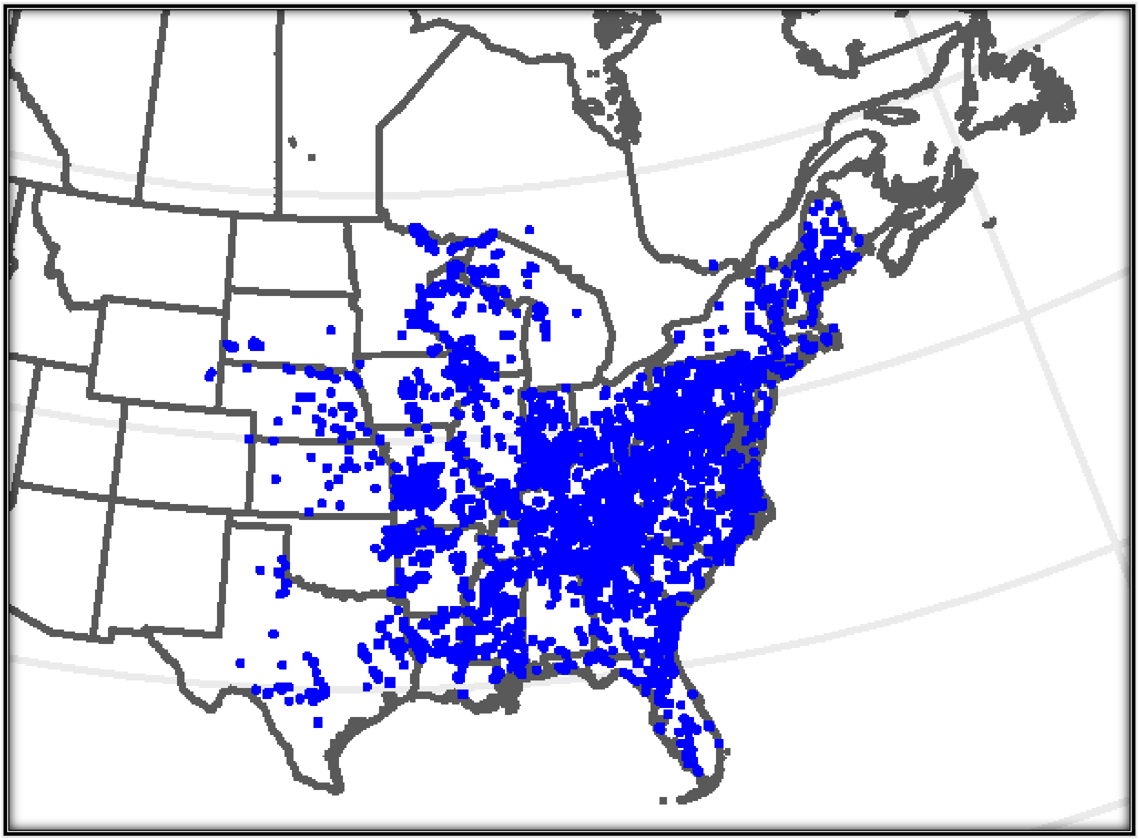

Figure A-2A2. Documented range of TCB (blue dots), as known from available records (acoustic calls, mist-net captures, and hibernacula records) in the U.S. and Canada (Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center). Disclaimer: Provisional information is subject to revision. This map shows data provided to the SSA for TCB and does not replace the species range (Figure. 2.3).

One way we measure population health was hibernacula abundance (*N*) and population trend (λ). Despite the thousands of winter counts, data are not available for all years and not necessarily both pre and post WNS arrival. Thus, to estimate historical TCB *N* and *λ*, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233). For TCB sites with more than 5 data-points (n=462; Table A-2A1), they fit the data using a statistical linear mixed effects model (henceforth referred to as Status/Trends model) to estimate the yearly abundance for each hibernaculum from 1990 through 2020. For sites with fewer than 5 colony counts (n=1,489), they used last observed count and used the λ from closest hibernaculum or complex of hibernacula. The Status/Trends model relies upon WNS year of arrival, thus, *N* and *λ* estimates vary with the occurrence of *Pd*. Wiens et al. (2022, pp. 231–233) used two projections of *Pd* occurrence (referred to as *Pd* occurrence Model 1 and 2) to identify year of arrival for hibernacula lacking data (see *Current and Future Primary Drivers* subsection below) to capture uncertainty in the presence and spread rate of *Pd* at unknown and uncontaminated sites. Both models use available disease surveillance data documenting past detection of *Pd* and surveillance effort but use different parameters to estimate occurrence of *Pd* beyond those detections. Hence, we have two estimates for yearly historical colony *N* and λ. See Appendix 5 for further details on the Status/Trends model.

Table A-2A1. Number of hibernacula by State/Province used to estimate historical TCB N and λ.

State/Province	# of hibernacula
Alabama	5
Arkansas	39
Connecticut	3
Delaware	1
Florida	9
Georgia	8
Illinois	13
Indiana	18
Iowa	1
Kentucky	40
Maryland	6
Massachusetts	4
Michigan	12
Minnesota	5
Mississippi	37
Missouri	2
New Hampshire	5
New Jersey	1
New York	16
North Carolina	27
Ohio	2
Oklahoma	11
Pennsylvania	41
Quebec	1
Rhode Island	1
South Carolina	3
Tennessee	44
Vermont	7
Virginia	13
West Virginia	33
Wisconsin	54
TOTAL	**462**

Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233) as described above. Additionally, because colony estimates are not available for all hibernacula and because bats occupying a given hibernaculum disperse to many different locations on the summer landscape, we also relied upon the results from USGS-led summer capture records and acoustic records analyses to garner insights on population trends at regional scales (see *Summer Data Analyses* subsection below).

Step 3. Identify Current and Future Primary Influences

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) influences of population numbers. We identified WNS, wind related mortality, habitat loss, and climate change as the primary influences on the species' abundance. We also identified several other potential influences but based on available information were either too local in scale or lacking data to assess species response.

Qualitative/Comparative Threat Analysis - We assessed the impact of the four influences using an approach adapted from Master et al. (2012, entire) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. Briefly, scope is the proportion of the populations that can be reasonably expected to be affected by the threat within 10 years (current). Severity is the level of damage to the species from the threat. Impact is the degree to which the species is directly or indirectly threatened based on the interaction between the scope and severity values. The criteria used to assign levels are shown in Figure A-2B3.

SCOPE (% of range)	SEVERITY (% of population decline) Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure A-2A3. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012, entire). Impact level (Low to Very High) is based upon the scope and severity assigned.

Quantitative Threat Analysis – We sought to model the impact of the four primary drivers, however, we did not have the time to rigorously determine the species response to changes in climate change and habitat loss. Although we have information on ongoing effects to North American insectivorous bats associated with climate change in specific geographic areas, given the differences in types and magnitude of climate change, the large range of TCB, and the fact that we had finite time and resources, we were unable to reliably quantify TCB's response in a manner that could be included in the population model (e.g., what specific changes to which specific demographic parameters should we include in response to projected changes in temperature or precipitation). Similarly, habitat loss or alteration can lead to locally consequential effects, especially with the compounding effects of WNS. We considered information on loss or alteration of hibernacula as well as information on changes in landcover types across TCB's range; however, given our finite time and resources we were unable to project rangewide future landcover changes or TCB's associated response in a manner that could be included in the BatTool (e.g., what specific landcover changes would result in changes to demographic parameters). Instead we provided a narrative on the spatial extent and magnitude of impact from these two stressors.

To assess the current and plausible future state conditions (magnitude and severity) for WNS and wind related mortality, we used published data, expert knowledge, and professional judgment. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible "low" and "high" impact scenarios. These scenarios were used as inputs to a population-specific demographic model (BatTool, Erickson et al. 2014, entire; explained Step 4 below) to project abundance given specified WNS and wind mortality scenarios.

WNS – To assess the current and future severity of WNS, we calculated disease-induced fatality rates from data gathered from winter colonies following *Pd* arrival (referred to as "WNS impacts schedule", see below). We assumed that the WNS impacts schedule (severity) will not change into the future, and hence, the only difference between the current and future WNS scenarios is the rate of spread (scope) of WNS. To estimate the current and future occurrence of WNS, we relied on two models (several others are available with similar predictions), Wiens et al. (2022, pp. 226–229) and Hefley et al. (2020, entire). We refer to these projections as "*Pd* occurrence model 1 and 2." Both models rely on the same WNS surveillance dataset but allowed us to capture uncertainty in spread rates. Additionally, each model performs differently in different geographic regions of the country, making one model better than the other in a certain area of the country and vice-versa.

Since 2007, collection and management of surveillance data for WNS and *Pd* on bats or in the environment has been coordinated by the National Response to WNS, led by USFWS. State agencies or other appropriate land-management entities conduct most sample collection for disease surveillance and are responsible for reporting county level-determinations of *Pd* status. WNS is confirmed by histopathological observation of lesions characteristic of the disease (Meteyer et al. 2009, entire), molecular detection of the fungus (Muller et al. 2013, entire), or characteristic field signs associated with WNS Case Definitions determined by USGS, National Wildlife Health Center. Year of arrival of WNS or *Pd* at a location is documented at a county-level resolution (available at www.whitenosesyndrome.org).

Wiens et al. (2022, pp. 226–229) used a Gaussian interpolation and projection using linear movement estimates based on observed rates of spread of *Pd* (see Appendix 5). Hefley et al. (2020, entire) used a diffusion and growth model, which estimates the prevalence (similar to abundance) of *Pd* at a location. In their model, prevalence is influenced by proximity to known occurrences and environmental covariates of percent canopy cover, terrain ruggedness index, waterways, locations of mines, and karst geology. Year of arrival of *Pd* at a location is assigned to the year in which prevalence exceeds 0.25 (this level was chosen by the SSA Core Team based on the prevalence value observed at a subset of sites where *Pd* has already been detected). Separate parameters were calculated to estimate current and future distribution of *Pd* in the Pacific Northwest, where the fungus is expected to have initiated a second epicenter after "jumping" from the nearest known previous occurrence (Lorch et al. 2016, p. 4). Using their estimates of spread rates, future distribution of *Pd* was projected on an annual scale for every 10 km x 10 km grid cell until *Pd* was predicted to be present throughout the entirety of the species' range (Wiens et al. 2022, pp. 226–229) or until statistical confidence interval in the model projection was too great for the value to be reliable (Hefley et al. 2020, entire). The projected *Pd* spread under the two models is shown in Figure A-2B4.

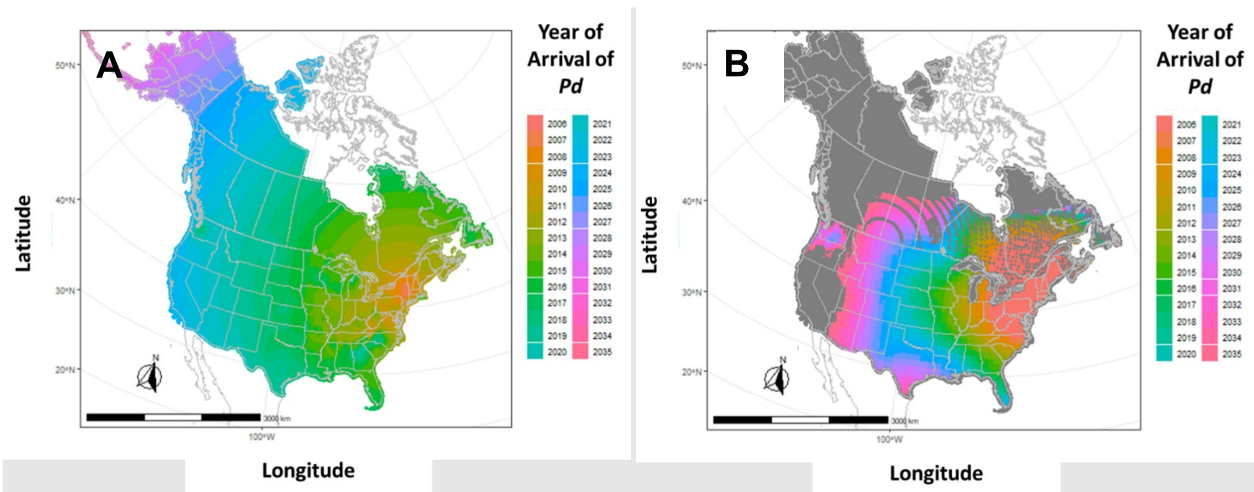

Figure A-2A4. Two models of Pd occurrence in North America since 2007 and into the future. A) A Gaussian interpolation map using spatial relationships and direct observations of Pd occurrence (Wiens et al. 2022, pp. 226–229). B) A diffusion and growth model using observed Pd prevalence in diagnostic samples to predict environmental prevalence of Pd based on spatial and environmental covariates (Hefley et al. 2020, entire).

To estimate current and future WNS impact (fatality rates), we relied on Wiens et al. (2022, pp. 233–235) derived "WNS impacts schedule"; a distribution of annual-specific changes to survival rates. They used data collected during winter hibernacula surveys from 1990–2020 and calculated the proportional change in size of the colony between calendar years and between years since arrival of *Pd*. Assuming that change in the estimated colony size was the result of WNS-induced mortality, these estimates of percent change in colony size were translated into changes in adult over-winter survival rate (a parameter in the BatTool). Lastly, they collated these site-specific over-winter survival rates to create annual distributions, i.e., WNS impacts schedule (Figure A-2A5.). This WNS impacts schedule was used in the BatTool to apply WNS impacts to hibernacula over time. For a few sites, the severity of WNS impact has deviated from the norm; for these exceptions, a colony-specific WNS impacts schedule was derived (see Wiens et al. 2022, pp. 231–247). See Appendix 5 for additional information and further description of future scenarios.

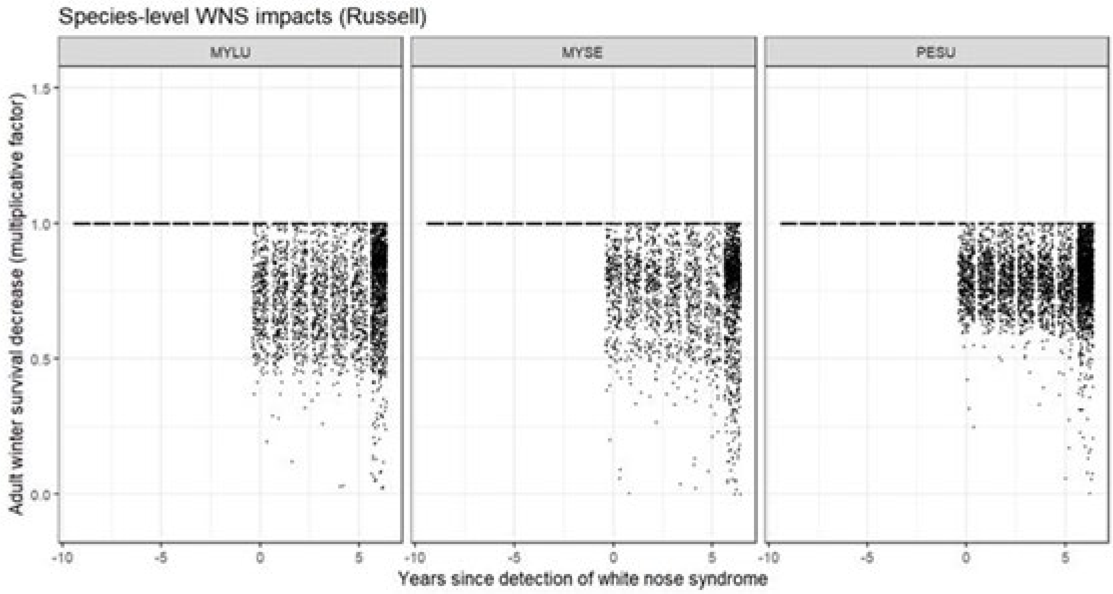

Figure A-2A5. Adult winter survival decreases annually after Pd detection for little brown bat (MYLU), northern long-eared bay (MYSE), and tricolored bat (PESU). These data were used to create the WNS impacts schedule. The data depicted for 6 years since detection of Pd include all years since detection ≥ 6.

Wind - To assess the current and future magnitude and severity of current and future wind energy development, we 1) estimated species-specific wind fatality rates (bats per megawatt (MW) per year), 2) applied current and projected future wind capacity within the species' range, and 3) applied species-specific fatality rates to current and future wind capacity to estimate wind related mortality for known hibernating populations. We assumed the only difference between the current and future wind scenarios is the amount of installed wind capacity and the proportion of curtailed turbines. We did, however, use different fatality rates for pre- and post-WNS as the data indicated different percent species compositions before and after WNS arrival.

To estimate wind fatality rates (severity), we reached out to the public, states, USFWS Ecological Services field offices, and other partners to request data from wind post-construction bat fatality monitoring at wind projects within the ranges of TCB, little brown bat, and northern long-eared bat. We obtained 287 reports for wind projects in 20 states within USFWS Legacy Regions (Regions) 3, 4, 5, and 6 (Figure A-2A6).

Figure A-2A6. U.S. Fish and Wildlife Service Regions.

For a subset (n = 155) of these reports (those that met our inclusion criteria, described below) we calculated species-specific per MW fatality rate using the following equation:

TCB *per MW fat rate = Bfat * %Sp*

Where *Bfat* is the all-bat fatality rate per MW and *%Sp* is the species-specific percent composition of fatalities reported. *Bfat* was calculated for each Region by deriving annual all-bat per MW fatality rates for each study in our subset, applying corrections for unsearched areas and portions of the year as needed, and then averaging the corrected all-bat fatality rates across the studies in each Region. *%Sp* was calculated by dividing the total number of each species' carcasses reported in our subset of studies by the total number of bat carcasses.

To maximize consistency and comparability across studies in our database, we applied the following inclusion criteria:

1. Study must report a bats/megawatts (MW) or bats/turbine fatality rate, corrected for searcher efficiency (SE) and carcass persistence (CP). If bats/turbines is the only reported fatality rate, the report must also include the number of turbines and MW at the site in order to calculate bats/MW.
2. Turbines were operated without curtailment (i.e., no feathering below manufacturer's or other cut-in speeds) during the study period. In a few instances where studies tested certain cut-in speeds in a subset of turbines and reported separate fatality rates for curtailed versus control (uncurtailed) turbines, the control turbine fatality rate was used.
3. The study search interval was seven days or less.
4. The study provided the range of dates when carcass searches were performed.
5. The study provided the search area (i.e., plot) dimensions.

For the U.S., we assessed our species composition rates by USFWS Region. We had insufficient data to generate TCB percent composition rates for Regions 2, 4, and 6. We used American Wind Wildlife Institute's (AWWI) (2020, p. 19) TCB composition rates for Region 2 (AWWI 2020, p. 19), but they did not report values for Regions 4 and 6. For Region 4, along with the southern portion of Region 5, we observed significantly higher TCB fatalities compared to TCB fatality rates in the northern portion of Region 5, likely due to higher TCB abundance in the southern extent of its range. Therefore, we calculated separate TCB composition rates for the northern and southern portions of Region 5, incorporating Region 4 data with Region 5 South. We combined Region 3 and Region 6 data to calculate a Region 3/6 percent species composition rate. For Canada, we used species composition rates (%*Sp*) reported in Bird Studies Canada et al. (2018, pp. 17–18). Additionally, we found differences in *%Sp* following *Pd* arrival, and thus, applied post-*Pd* arrival *%Sp* rates as suggested by the data.

It should be noted that reported fatality rates in our USFWS database were derived using a variety of estimators with differing, imperfect assumptions and biases toward underestimating or overestimating mortality (i.e., see Rabie et al. 2021, entire). Additionally, a recent study by Huso et al. (2021, entire) found that bird and bat fatality rates were relatively constant per unit energy produced by turbines under similar environmental conditions regardless of their size, suggesting that the relative amount of energy produced, rather than simply the size, spacing, or nameplate capacity of turbines, determines the relative all-bat fatality rate. However, bat fatalities per turbine generally increased with turbine size or MW capacity (Huso et al. 2021, p. 4). Lacking information about the capacity factor (total energy produced relative to the theoretical maximum, or nameplate capacity), for all the turbines in our database, we relied on reported bats/MW fatality rates. As such, our averaged fatality rates may overestimate mortality for facilities with high capacity but low energy production (low capacity factor) or vice versa, but are more robust than bats/turbine fatality rates. Moreover, because they are averages across many facilities and states, they should capture the general capacity factor trends across regions, at least for built facilities as of October 2020.

To determine current and future wind capacity (magnitude), we obtained current wind capacity data from the U.S. Wind Turbine Database (USWTDB version 3.2; Hoen et al. 2018, entire) and corrected/incorporated facility-specific curtailment information (USFWS, unpublished data). For future projections, we used—at the counsel of experts at USDOE and NREL—the 2020 NREL High and Low Onshore Wind Cost Scenarios data (Cole et al. 2020, p. 26) as reasonable lower and upper bounds of future U.S. wind capacity by state. For Canada, we used Canada Energy Regulator's (CER) (CER 2020, pp. 5, 22–23, 56–57) Evolving and Reference (baseline) scenarios as our upper and lower bounds, respectively (see Appendix 4 for further description of future scenarios).

Lastly, to calculate hibernacula-specific wind mortality, we relied upon the analysis by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) summed wind capacity under the lower and upper bound scenarios for each 11x11 km NREL grid cell centroid and calculated a grid cell-specific mortality estimate. They then created a distance decay function to allocate the total mortality per 11x11 km grid cell among hibernacula, relative to the size of the hibernating populations and distance of hibernacula (within the known average maximum migration distance) from the grid cell centroid (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). To account for mortality reductions associated with feathering below the manufacturer's cut-in speed or higher, we applied a 50 percent mortality reduction to turbines implementing any level of curtailment during the fall or summer seasons, per our 2020 data (USFWS unpublished data). We then multiplied this 50% mortality reduction by the relative proportion of all-bat mortality reported by season in our post-construction mortality database (USFWS, unpublished data; Table A-2A2). Based on these proportions, we applied an overall mortality reduction of 50 percent to turbines curtailing in both summer and fall and a 34 percent reduction to turbines curtailing in fall only (Table A-2A3).

Table A-2A2. Proportion of all-bat mortality by season (USFWS, unpublished data).

Season	Date Range	Proportion of All-bat Mortality
Spring	March – May 31	0.065
Summer	June 1 – July 30	0.252
Fall	August 1 – November 30	0.68
Summer + Fall	June 1 – November 30	1.0

Table A-2A3. Curtailment categories by season and associated fatality reductions applied to turbine MW.

Category	Curtailment Season	Total Mortality Reduction Applied*
No Curtailment	None	N/A
Fall Only	Fall, Fall + Spring	0.34
Summer + Fall	Summer + Fall, Summer + Fall + Spring	0.50

**Reflects 50% mortality reduction for curtailment multiplied by seasonal proportion of all-bat fatality (Table A-2A2).*

Step 4. Project Future Number, Health, Distribution of Populations Under Current and Future Influences.

To project future abundance and trend given current and future state conditions for WNS and wind, we used an existing bat population tool, updated with TCB-specific demographics (BatTool, Erickson et al. 2014). The BatTool is a demographic model that projects hibernaculum abundance over time given starting abundance (N), trend (λ), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind related mortality as specified by the wind capacity scenarios. Starting abundance (N) and trend (λ) were derived from the Status/Trends model described in Step 1 above. For each hibernaculum, the model was run for 100 simulations projecting 40 years into the future.

Using these projected abundance estimates, we calculated various hibernaculum-level and representation unit-level (RPU, described in Chapter 2) metrics to describe the species' historical, current, and future number, health, and distribution of populations given current and future influences. Figure A-2A7 shows a simplified schematic of the purpose of the various models used and Figure A-2A8 provides the conceptual framework for the BatTool.

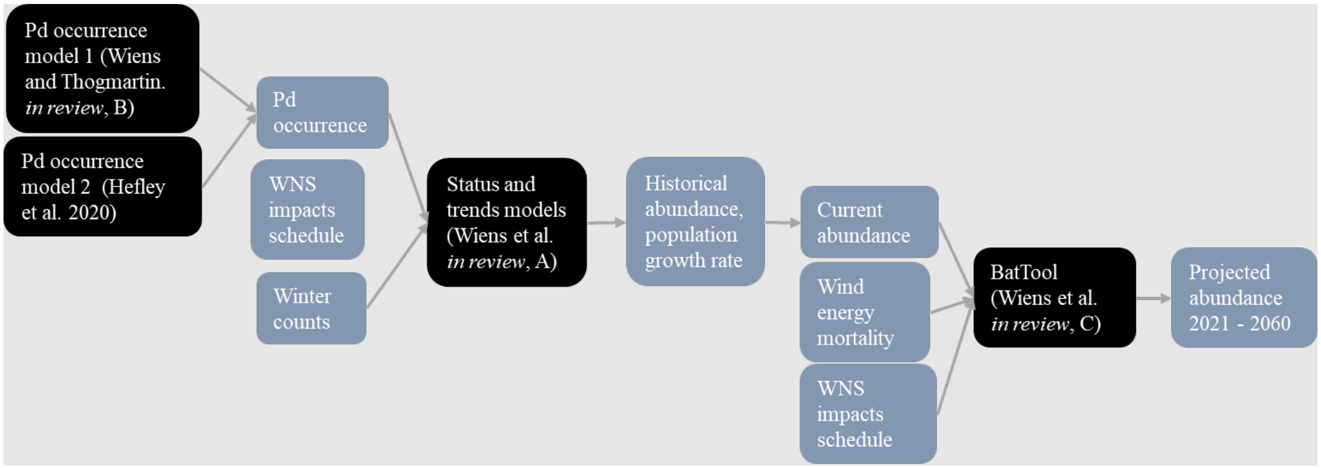

Figure A-2A7. Simplified schematic showing the role for each of the four mathematical models: two Pd prevalence models, Status and Trends model, and BatTool.

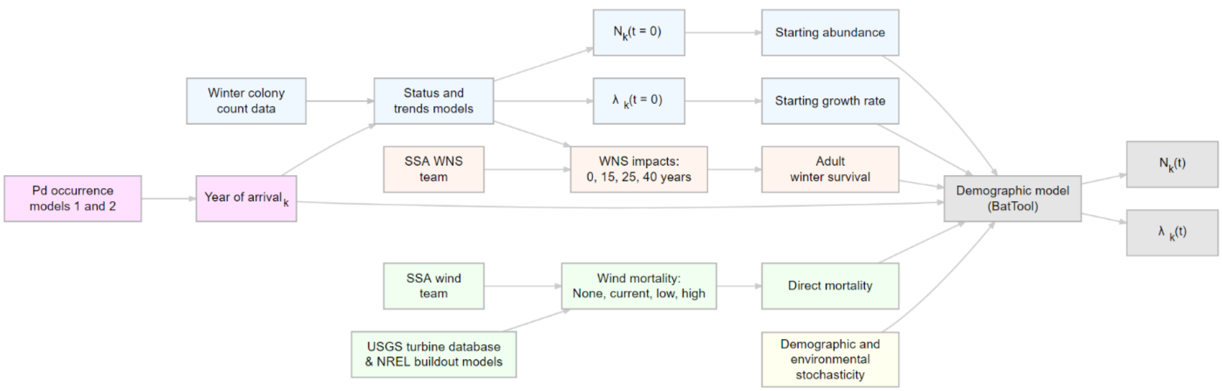

Figure A-2A8. A schematic of the BatTool, including origins of model inputs.

Summer Data Analyses

Because the population of bats monitored at a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led summer capture records and acoustic records analyses. These studies assessed the changes in occupancy (λ) and capture rates over time. We briefly describe their methodologies here; refer to Appendix 5 for further details.

Deeley and Ford (2022, entire) assessed the change in capture rates during summer surveys to garner insights on change in capture rates over time and to assess reproductive conditions of female bats, age structure, and body condition indices of male bats. Between 1999 and 2019, they analyzed 10,489 TCB in 3,290 sampling events in which 736 (7.0%) of records had sufficient information. Rates of capture per unit effort or per sampling event were calculated for each species on an annual timescale by year and by year since arrival of *Pd* based on Wiens et al. (2022, pp. 226–229) *Pd* occurrence estimates (model 1). Stratton and Irvine (2022, entire) assessed recent change in predicted summer occupancy using stationary and mobile acoustical detector records and capture records across TCB's range. They developed a false-positive occupancy model to estimate probability of occurrence, annual rate of change in summertime occupancy (λ_{avg}), and total change in occupancy (λ_{tot}) from 2010 to 2019. Predicted occupancy was calculated for each 10km by 10km grid cell in TCB's range. The occupancy prediction used covariates of mean elevation, terrain ruggedness index, annual mean precipitation, annual mean temperature, distance to nearest wind farm, percent forest cover, and percent water cover to provide estimates in locations that were not sampled directly. Metrics of change were based on aggregating predicted occupancy between 2010 to 2019 at the RPU and rangewide scale. Whitby et al. (2022, entire) analyzed relative abundance of TCB annually using acoustical data collected during mobile transect surveys. They analyzed the number of calls detected along driving routes and estimated changes in abundance over the past decade relative to the arrival of WNS and changes in installed wind energy facilities. These analyses were used to estimate rate of change in populations at state and RPU scales.

B: Adaptive Capacity Analysis

To garner additional insights into the intrinsic (and historical) ability of these species to withstand stressors and adapt to novel changes in the environment, we used the framework put-forth by Thurman et al. (2020, entire). Specifically, Thurman et al. (2020, entire) developed an attribute-based framework for evaluating the adaptive capacity of a given species. Although the basis for the framework is climate change based, the attributes apply to other stressors and changes a species may be exposed to. They identified 12 "core" attributes out of their 36 potential attributes (Figure A-2B1), which collectively provide a comprehensive means of assessing adaptive capacity and are generally available for many species. For each attribute, a species is evaluated on a 5-level "low–moderate–high" scale, with criteria specified for each adaptive capacity level. They do not advise a composite level as many of the attributes interact and some may be "so important that they may overwhelm other considerations (i.e., "deal makers" or "deal breakers"). Using the criteria defined in Thurman et al. (2020, supporting information), we categorized each species' level of adaptive capacity for each of the 12 core attributes (Table A-2B1)

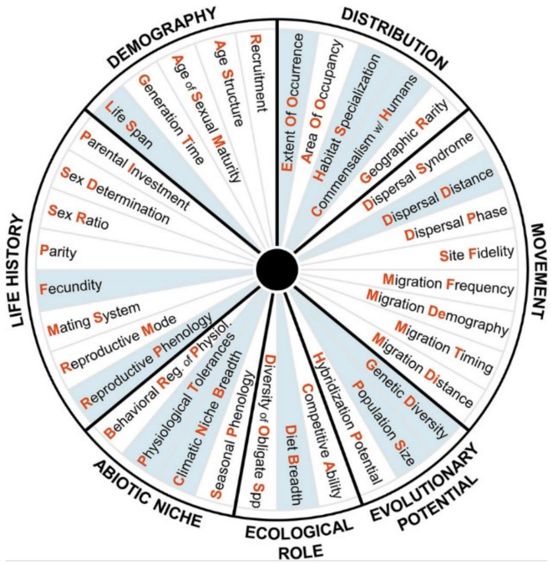

Figure A-2B1. The adaptive capacity "wheel", depicting 36 individual attributes organized by ecological complexes (or themes). Twelve core attributes, representing attributes of particular importance and for which data are widely available, are highlighted in light blue (from Thurman et al. 2020, Figure 1).

Table A-2B1. Assessm*ent of 12 core attributes of TCB adaptive capacity (from Thurman et al. 2020, Supporting Information).*

Core Attribute	**Relative Level**	**Evidence and Relevance**
Extent of Occurrence	High	Broadly distributed (Davis 1959, entire; Geluso et al. 2005, p. 406; Kurta et al. 2007, p. 405; Slider and Kurta 2011, p. 380; Adams et al. 2018, entire; Hanttula and Valdez 2021, p. 132); typically, a broader distribution is expected to confer higher adaptive capacity.
Habitat Specialization	Low/ Moderate	Summer habitat: generalist; suitable roosting habitat includes live and dead leaf clusters of live or recently dead deciduous hardwood trees (Veilleux et al. 2003, p. 1071; Perry and Thill 2007, pp. 976–977; Thames 2020, p. 32); Spanish moss and *Usnea trichodea* lichen (Davis and Mumford 1962, p. 395; Poissant 2009, p. 36; Poissant et al. 2010, p. 374); pine needles (Perry and Thill 2007, p. 977); eastern red cedar (Thames 2020, p. 32); artificial roosts (e.g., barns, beneath porch roofs, bridges, concrete bunkers) (Jones and Pagels 1968, entire; Barbour and Davis 1969, p. 116; Jones and Suttkus 1973, entire; Hamilton and Whitaker 1979, p. 87; Mumford and Whitaker 1982, p. 169; Whitaker 1998, p. 652; Feldhamer et al. 2003, p. 109; Ferrara and Leberg 2005, p. 731; Smith 2020); and rarely within caves (Humphrey et al. 1976, p. 367; Briggler and Prather 2003 p. 408; Damm and Geluso 2008, p. 384). Specific roost requirements needed for successful pregnancy and recruitment likely include narrow temperature ranges. Exhibit high site fidelity.

Core Attribute	Relative Level	Evidence and Relevance
		Winter habitat: specialist; suitable hibernacula conducive to longer torpor bouts; hibernacula include caves and mines (Barbour and Davis 1969, p. 116); road-associated culverts (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Limon et al. 2018, entire; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276); tree cavities (Newman 2020, p. 14); abandoned water wells (Sasse et al. 2011, p. 126); rock shelters (e.g., fissures in sandstone and sedimentary rock) (Johnson 2021, pers. comm.). Exhibit high site fidelity.
Commensalism with Humans	Moderate	Broadly distributed across human-modified landscapes, but less tolerant when suitable roosting sites have been eliminated (e.g., urban and agricultural dominated landscapes) (Duchamp and Swihart 2009, p. 855; Farrow and Broders 2011, p. 177). Conversely, will utilize man-made infrastructure as hibernation sites (e.g., abandoned mines, tunnels, road-associated culverts) (see references above).
Genetic Diversity	Moderate	Martin (2014, entire) observed significantly distinct structure in maternally inherited mitochondrial DNA across the sampled range. Large portions of the range have not been sampled and we are unaware of additional genetic information.
Population Size	Low	Once common, populations have decreased significantly (Cheng et al. 2021, entire); adaptive capacity may decrease with smaller populations.
Dispersal Distance	Moderate/ High	Females migrate up to 243 km (151 miles) from winter to summer habitat (Samoray et al. 2019, entire); individuals have high site fidelity.
Climatic Niche Breadth	High	Broad climatic niche breadth across range (e.g., occur from Canada to Central America) (see references above); may indicate a broader tolerance to climate change because they currently encompass a broader array of climate conditions.
Physiological Tolerances	Moderate	If physiological tolerance reflects the degree to which a species is restricted to a narrow range of abiotic conditions, we assume TCB have at least a moderate level of physiological tolerance during the summer given they are found as far north as Canada and as far south as Central America. Physiological tolerance during hibernation, however, is narrower given requirement for suitable hibernacula conducive to longer torpor bouts.
Diet Breadth	High	Opportunistic feeders; small insects ranging from 4–10 mm in length (primarily Coleoptera, Diptera, Lepidoptera, and Trichoptera) (Ross 1967, p. 223; Whitaker 1972, p. 879; LaVal and LaVal 1980, p. 24; Griffith and Gates 1985, p. 453; Hanttula and Valdez 2021, p. 132). Hibernation period may decrease with warming temperatures, but insect hatches may occur earlier.
Reproductive Phenology	Low	TCB mate in the fall. Females store sperm in their uterus during the winter and fertilization occurs soon after spring emergence from hibernation (Guthrie 1933, p. 209). Females typically give birth to two young, rarely one or three between May and July (Allen 1921, p. 55; Barbour and Davis 1969, p. 117; Cope and Humphrey 1972, p. 9).

Core Attribute	**Relative Level**	**Evidence and Relevance**
Life Span	Moderate/ Low	The greatest longevity records are 14.8 years and 11.2 years for a male and female, respectively (Paradiso and Greenhall 1967, pp. 251–252; Walley and Jarvis 1972, p. 305).
Fecundity	Low	Litter size is usually two, rarely one or three, annually (see references above).

Appendix 3: Supplemental Results

A: Historical Condition

*Table A-3A1. The **historical** number of states/provinces, spatial extent (Extent of Occurrence: EOO), winter abundance and documented hibernacula rangewide.*

# of States / Provinces	EOO (acres)	# of known hibernacula	Abundance (max)
34/1	1.1 billion	1,951	140,547

*Table A-3A2. The **historical** number of documented hibernacula and winter abundance by RPU.*

RPU	# of known hibernacula	Abundance (max)
Eastern	211	16,576
Northern	1,124	95,906
Southern	616	32,433

B: Current Condition

*Table A-3B1. Projected yearly rangewide number of states/provinces with 2 or more bats persisting, spatial extent (EOO in acres), number of hibernacula (90% CI), and median abundance (90% CI) under **current** conditions.*

Year	# of States / Provinces	EOO (ac)	# of hibernacula	Abundance
2020	29/1	929 million	1,378 (CI 1,317–1,378)	67,898 (CI 67,444–68,352)
2030	15/0	383 million	171 (CI 22–734)	15,661 (CI 8,312–26,690)
2040	8/0	262 million	49 (CI 4–464)	14,611 (CI 7,181–23,056)
2050	6/0	205 million	30 (CI 3–379)	16,557 (CI 12,368–22,444)
2060	5/0	136 million	23 (CI 3–340)	19,506 (CI 13,619–28,429)

*Table A-3B2. Projected RPU-level number of hibernacula and probability of population growth (λ)>1 (pPg) under **current** conditions.*

RPU	Year	# of hibernacula	pPg
Eastern	2020	114	0%
	2030	3	10%
	2040	0	54%
	2050	0	45%
	2060	0	41%
Northern	2020	856	0%
	2030	97	0%
	2040	13	52%
	2050	7	66%

RPU	Year	# of hibernacula	pPg
	2060	5	98%
Southern	2020	408	0%
	2030	71	21%
	2040	36	63%
	2050	23	64%
	2060	18	58%

*Table A-3B3. Projected RPU median abundance (90% CI) under **current** conditions.*

RPU	2020	2030	2040	2050	2060
Eastern	1,891 (CI 1,786–1,996)	103 (CI 32–257)	26 (CI 0–177)	14 (CI 0–249)	5 (CI 0–509)
Northern	41,448 (CI 41,428–41,468)	5,374 (CI 3,667–6,989)	2,733 (CI 1,208–5,437)	3,535 (CI 403–11,141)	3,864 (CI 202–17,433)
Southern	24,559 (CI 23,980–25,138)	11,042 (CI 3,860–20,086)	10,853 (CI 3,565–20,544)	11,810 (CI 5,065–20,665)	12,395 (CI 5,999–20,594)

Table A-3B4. Summary of recent TCB population trends from multiple data types and analyses. Winter Colony analysis – Wiens et al. (2022, entire); Summer Occupancy analysis – Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire).

Scale	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Eastern	-89%	-17%	-19%	-38%
Northern	-57%	-17%	-16%	-86%
Southern	-24%	-37%	-12%	-65%
Rangewide	-52%	-28%	-12% to -19%	-53%

C: Future Condition

*Table A-3C1. Projected rangewide number of states/provinces and hibernacula with 1 or more bats persisting, spatial extent (EOO), number of hibernacula (90% CI) and median abundance (90% CI) under **future** scenarios.*

Year	# of States / Provinces	EOO (ac)	# of hibernacula	Abundance
2020	29/1	929 million	1,378 (CI 1,317–1,378)	67,898 (CI 67,444–68,352)
2030	14/0	329 million	124 (CI 18–603)	10,138 (CI 8,053–12,519)
2040	7/0	169 million	18 (CI 3–324)	7,225 (CI 3,604–12,520)
2050	5/0	86 million	11 (CI 2–251)	8,495 (CI 2,524–19,690)
2060	5/0	77 million	9 (CI 2–214)	10,955 (CI 2,194–27,292)

*Table A-3C2. Projected RPU-level number of known hibernacula, and probability of population growth (λ)>1 (pPg) over time under **future** scenarios.*

RPU	Year	# of hibernacula	pPg
Eastern	2020	114	0%
	2030	2	12%
	2040	0	52%
	2050	0	50%
	2060	0	48%
Northern	2020	856	0%
	2030	81	1%
	2040	11	54%
	2050	7	72%
	2060	6	98%
Southern	2020	408	0%
	2030	41	11%
	2040	7	68%
	2050	4	72%
	2060	3	78%

*Table A-3C3. Projected RPU median abundance (90% CI) under **future** scenarios.*

RPU	2020	2030	2040	2050	2060
Eastern	1,891 (CI 1,786–1,996)	89 (CI 30–304)	29 (CI 0–183)	16 (CI 0–243)	12 (CI 0–467)
Northern	41,448 (CI 41,428–41,468)	4,860 (CI 3,608–6,541)	2,493 (CI 897–6,498)	2,863 (CI 271–10,955)	3,876 (CI 132–16,276)
Southern	24,559 (CI 23,980–25,138)	5,144 (CI 3,191–6,746)	4,553 (CI 1,639–7,654)	5,684 (CI 1,561–9,974)	6,020 (CI 1,579–12,855)

D: **Qualitative/Comparative Threat Analysis**

To estimate the proportion of TCB's range with wind mortality risk in 2020, we took the following approach:

1. Buffer extant hibernacula by avg. migration distance (126 km)
2. Buffer summer points by avg. migration distance (126 km)
3. Merge & dissolve buffered hibernacula and summer shapefiles into a "TCB occupied" area, clip TCB range by contiguous U.S. border for "TCB U.S. range", and clip TCB occupied area by TCB U.S. range.
4. Buffer and dissolve current turbines (Hoen et al. 2018, entire) by avg. migration distance for "wind threat" area

5. Clip wind threat area by TCB occupied area for "TCB wind risk" area
6. Compare TCB wind risk area with range area in U.S.: range area (4,605,467 km^2) and 2020 wind risk area: 2,449,924 km^2 (53% of U.S. range) (Figure A-3D1)

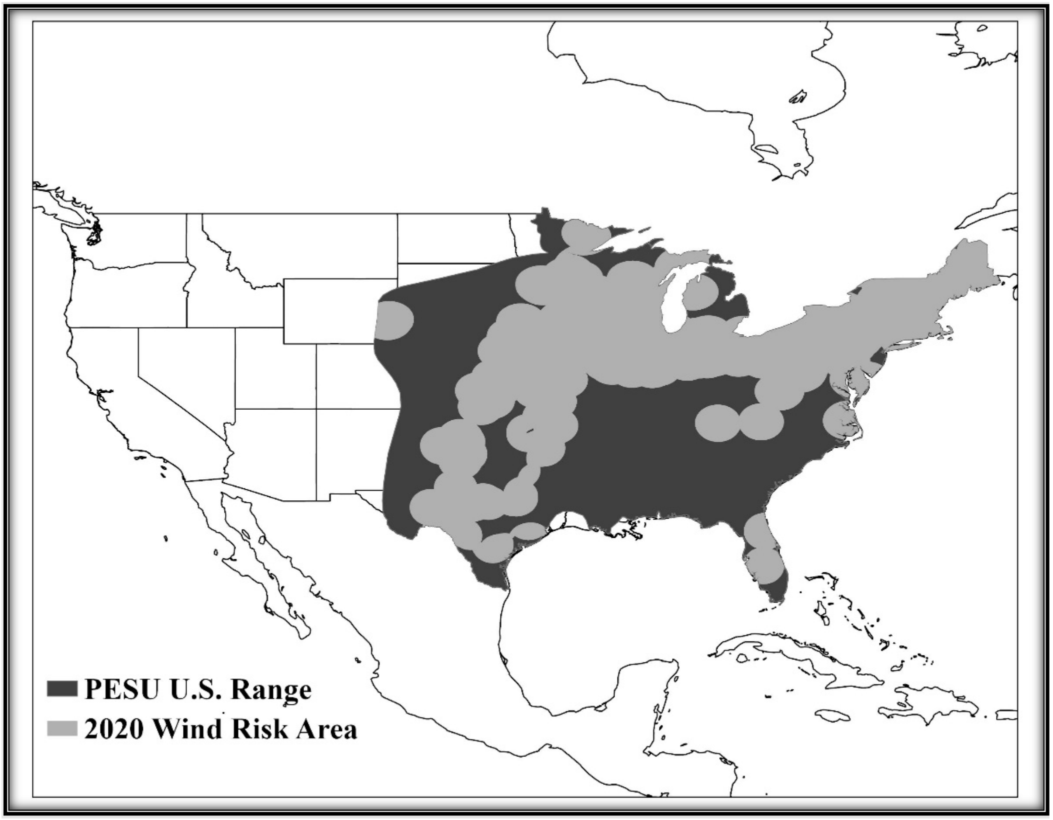

Figure A-3D1. Estimated extent of TCB's U.S. range with wind mortality risk.

To estimate the proportion of TCB's range with wind mortality risk in 2050 (per low and high build-out scenarios), we took the following approach:

1. 2050 Low Build-out Scenario:
    a. Buffer & dissolve 2050 High Onshore Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance (126 km) for "future wind threat: area. Note: Future MW summed by 11x11-km NREL grid so does not capture actual distribution of turbines on landscape.
    b. Clip wind threat area by TCB occupied areas for TCB 2050 low wind risk" area (U.S.)
    c. Compare TCB 2050 low wind risk areas with range area in U.S: range area (4,605,467 km^2) and 2050 low wind risk areas (1,720,963 km^2) (37% of U.S. range) (Figure A-3D2)
2. 2050 High Build-out Scenario:
    a. Buffer & dissolve 2050 Low Onshore Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance (126 km) for "future wind threat" area. Note: Future MW summed by 11x11-km NREL grid so does not capture actual distribution of turbines on landscape.

b. Clip wind threat area by TCB occupied areas for "TCB 2050 high wind risk" area (U.S.)
c. Compare TCB 2050 high wind risk areas with range area in U.S.: range area (4,605,467 km^2) and 2050 high wind risk areas (3,414,613 km^2)(74% of U.S. range) (Figure A-3D3)

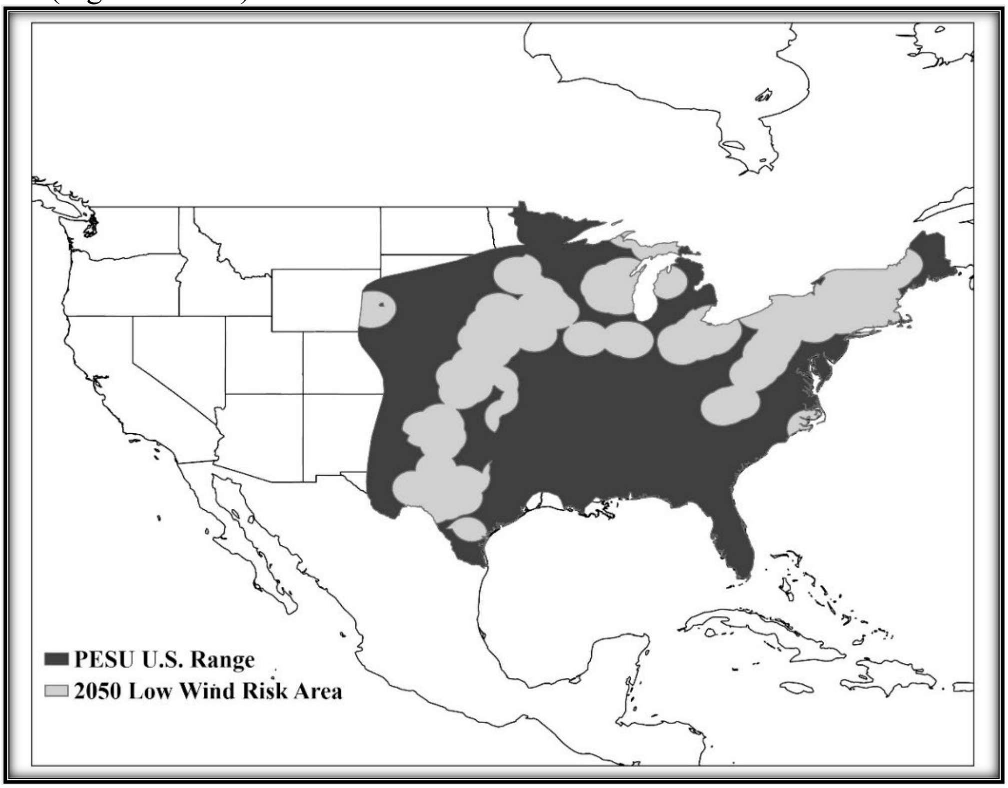

Figure A-3D2. Estimated extent of TCB's U.S. range with wind mortality risk in 2050 low build-out scenario.

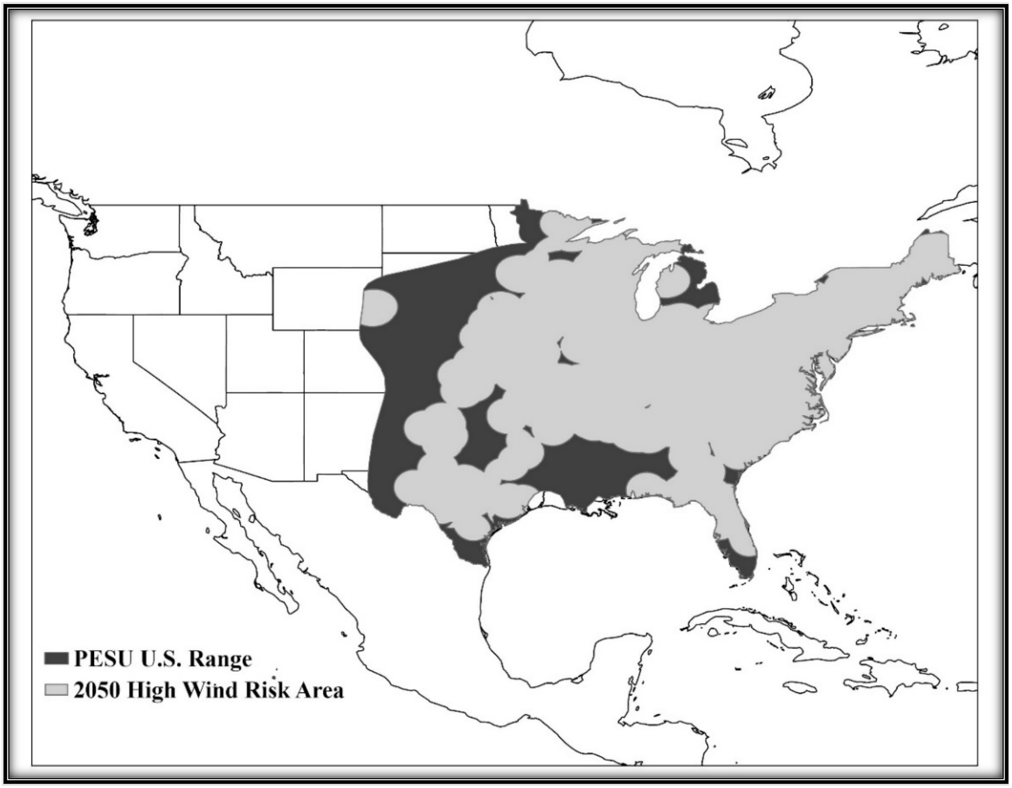

Figure A-3D3. Estimated extent of TCB's U.S. range with wind mortality risk in 2050 high build-out scenario.

To estimate the severity of impact from wind energy related mortality, we compared scenarios to baseline scenarios without wind energy mortality. Results are presented in Tables A-3D1 and A-3D2.

Table A-3D1 Projected median rangewide abundance given CURRENT wind energy mortality under 4 scenarios: 1) Pd model 1 and current wind energy related mortality, 2) Pd model 1 and no wind energy related mortality, 3) Pd model 2 and current wind energy related mortality ,and 4) Pd model 2 and no wind energy related mortality (all values derived from Wiens et al. 2022, entire).

	Median Rangewide Abundance			
Scenario	**2030**	**2040**	**2050**	**2060**
Pd Model 1 – Current mortality	10,623	10,355	17,379	23,657
Pd Model 1 – No mortality	13,416	13,823	25,032	36,388
% change	-21%	-25%	-31%	-35%
Pd Model 2 – Current mortality	24,195	19,599	17,809	17,252
Pd Model 2 – No mortality	29,846	24,245	22,629	21,955
% change	-19%	-19%	-21%	-21%

Table A-3D2. Projected median rangewide abundance given FUTURE wind energy mortality under 4 scenarios: 1) Pd model 1 and future wind energy related mortality, 2) Pd model 1 and no wind energy related mortality, 3) Pd model 2 and future wind energy related mortality, 4) Pd model 2 and no wind energy related mortality (all values derived from Wiens et al. 2022, entire).

	Median Rangewide Abundance			
Scenario	**2030**	**2040**	**2050**	**2060**
Pd Model 1 – low impact mortality	10,493	10,320	16,521	22,553
Pd Model 1 – future no mortality	13,789	14,226	24,766	36,277
% change	-24%	-27%	-33%	-38%
Pd Model 2 – high impact mortality	10,214	5,415	4,665	4,339
Pd Model 2 – future no mortality	14,797	7,397	5,719	4,811
% change	-31%	-27%	-18%	-10%

Appendix 4: Supplemental Threat and Future Scenario Information

A: WNS

Background

White-nose syndrome (WNS) is a disease of bats that is caused by the fungal pathogen *Pseudogymnoascus destructans (Pd)* (Blehert et al. 2009, entire; Turner et al. 2011, entire; Lorch et al. 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2016, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first observed in eastern New York in 2007 (with photographs showing presence since 2006; Meteyer et al. 2009, p. 411), although it is likely the pathogen existed in North America for a short time prior to its discovery (Keller et al. 2021, p. 3; Thapa et al. 2021, p. 17). Since May 2021, *Pd* and WNS have spread to 40 states and 7 provinces, with lesions indicative of disease confirmed in 12 species of North America bats, including TCB (Figure A-4A1, www.whitenosesyndrome.org; accessed online May 13, 2021; Hoyt et al. 2021, Suppl. Material). *Pd* invades the skin of bats, leading to significant morbidity and mortality that causes drastic declines in multiple species of hibernating bats.

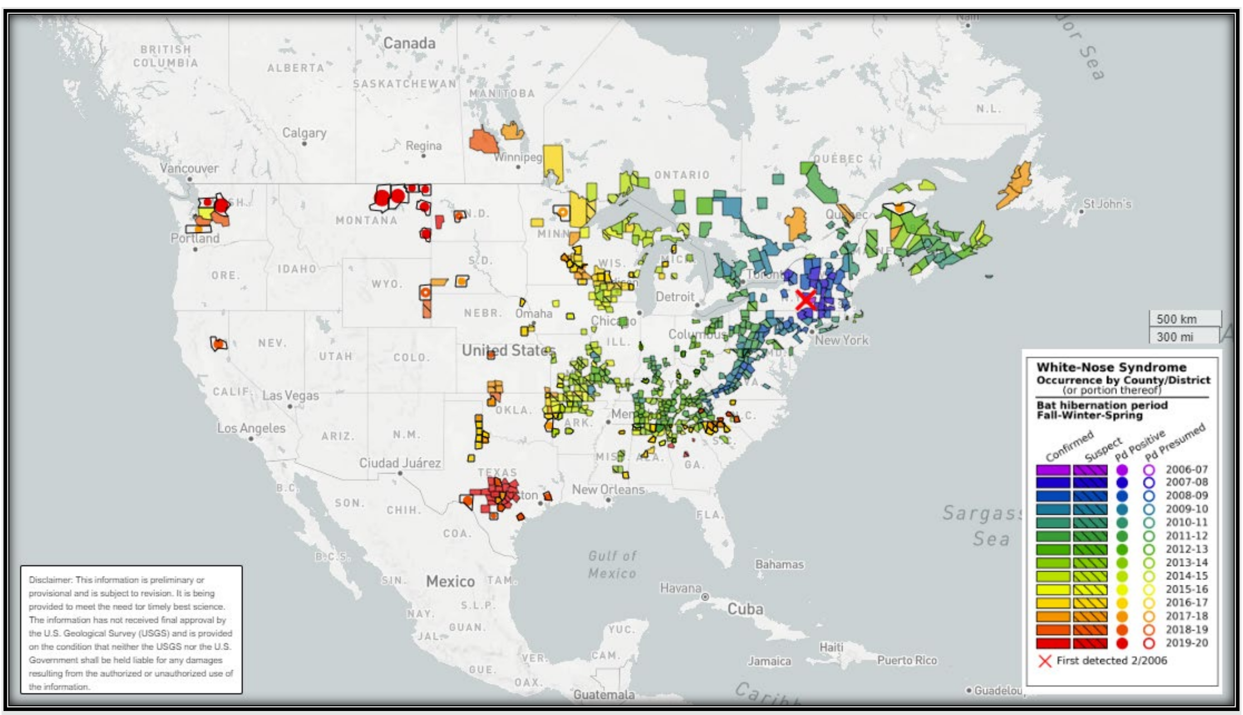

Figure A-4A1. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles). (www.whitenosesyndrome.org, accessed online: May 13, 2021)

As with any disease, there are three critical elements necessary for WNS to manifest: the pathogen, *Pd*; the host, hibernating bats; and a favorable environment for them to interact, mainly subterranean hibernacula (Turner et al. 2011, pp. 20–21).

- The *pathogen* that causes WNS, *Pseudogymnoascus destructans* (Gargas et al. 2009, pp. 151-152; Lorch et al. 2011, entire; Minnis and Lindner, 2013, p. 644) grows at cold temperatures ranging from 0–21 degrees C, with optimal growth temperature of 12–16 degrees C (Verant et al. 2012, p. 3), thus it is adapted to grow in conditions characteristic of bat hibernacula. It grows by invading the epidermis and underlying tissues of the face, ears and wings of bats (Meteyer et al. 2009, entire).
- The *hosts*, hibernating bats, are susceptible to infection by *Pd* in part because the physiological, physical and behavioral attributes associated with prolonged use of torpor present the opportunity for this cold-loving fungus to invade their tissues (Lorch et al. 2011, p. 2; Langwig et al. 2012, p. 4; Reeder et al. 2012, p. 4). In particular, hibernating bats overwinter in alternating states of torpor and euthermia (i.e., arousal) to survive prolonged periods without eating (McNab 1982, p. 171). To use limited fat stores efficiently, metabolic rates are greatly reduced, along with immune functioning and other physiological processes (Moore et al. 2011, p. 8).
- The *environment* where *Pd* and bats interact to cause disease is typically a winter roost location where bats engage in fall swarming and hibernation. The conditions of these locations overlap with the suitable growth requirements for *Pd* (Verant et al. 2012, p. 4). Hibernacula are often assumed to be caves and mines that provide overwinter shelter for large aggregations of hibernating bats, but these essential habitats take many forms and are used by individual bats to large, multi-species colonies. In North America, bats have been documented overwintering in caves, mines, rock crevices, talus, tunnels, bunkers, basements, bridges, aqueducts, trees, earthen burrows, leaf litter, and a variety of other roosts. For bats to hibernate successfully, the most important conditions are relatively stable- low temperatures, but generally above freezing, and high humidity (Perry 2013, p. 28). Notably, many North American hibernating bats select winter roosts that range between -4 and 16 degrees C (0 degrees C to 16.7 degrees C for TCB) (summarized in Webb et al. 1996, p. 763). The overlap of these roost conditions and suitable growth conditions for *Pd* (reported above), combined with the behavioral and physiological characteristics of their torpid state, are the primary factors making hibernating bats so susceptible to infection by *Pd*.

WNS is diagnosed histologically with the identification of "cup-like erosions" as *Pd* invades the skin tissue causing dehydration (Meteyer et al. 2009, p. 412). This fungal invasion destroys the protective skin tissue and disrupts water and electrolyte balance that is important to sustaining homeostasis through hibernation (Cryan et al. 2010, pp. 3–4; Warnecke et al. 2013, pp. 3–4). Likely in response to the homeostatic imbalance and irritation of the skin, *Pd* infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves (Reeder et al. 2012, p. 5; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). As a result, WNS leads to starvation as sick bats run out of fat needed to support critical biological functions.

Bats suffering from WNS may exhibit a variety of behavioral changes that can alter the course of morbidity from the disease. In addition to altered arousal patterns, bats have been observed relocating to different areas of hibernacula where conditions may be advantageous for hibernation or disadvantageous for *Pd* growth (Turner et al. 2011, p. 22; Langwig et al. 2012, p. 2; Johnson et al. 2016, p. 189). Observed changes in clustering behavior such that a greater proportion of bats in a colony are seen hibernating solitarily after WNS is present rather than huddled with roost mates may point to a behavioral factor that affects severity of WNS (Langwig et al. 2012, p. 2; Kurta and Smith 2020, p. 769), but may also be a maladaptive response to experiencing symptoms of WNS (Wilcox et al. 2014, p. 162). In many situations, infected bats have been documented exiting hibernacula earlier than usual and prior to when surface conditions are suitable for spring emergence. Early emergence has also been observed during daylight hours when diurnal predators such as hawks and ravens can take advantage of bats weakened by disease. It is possible that bats may find water to drink and insects to prey upon at this time, especially in more moderate climates, thus supplementing depleted energy reserves (Bernard and McCracken, 2017, p. 1492–1493), but in other areas, exposure to winter conditions and predation pose a significant threat to animals evacuating from hibernacula. Whether within the roost or on the landscape, WNS causes high rates of mortality during the hibernation season for multiple species including TCB (Turner et al. 2011, entire; Cheng et al. 2021, entire).

The weeks following emergence from hibernation also mark a critical period when bats incur energetic costs of clearing infection and recovering from over-winter sickness (Reichard and Kunz 2009, p. 461; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8). Meteyer et al. (2012, p. 3) proposed that bats with WNS can also suffer from immune reconstitution inflammatory syndrome, or IRIS. In this potentially fatal condition, deep or systemic infections that developed during hibernation while immune function was down-regulated trigger an excessive inflammatory response as immune function is upregulated in the spring (Meteyer et al. 2012, p. 5). Additionally, heavily compromised wing conditions resulting from overwinter infections and healing processes are likely to further limit foraging efficiency as the integrity of flight membranes is altered (Reichard and Kunz 2009, p. 462; Fuller et al. 2012, p. 6). These post-emergence complications can lead directly to mortality in addition to impacting reproductive success as a result of energetic constraints and trade-offs (Reichard and Kunz 2009, p. 462; Frick et al. 2010, p. 131; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8).

Transmission of Pd Among Bats

The fungus is spread via bat-bat and bat-environment-bat movement interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). Transmission occurs primarily in the fall and winter months when bats aggregate in hibernacula (Langwig et al. 2015b, p. 4). In spring, bats that survive a winter exposed to *Pd* can rid themselves of the fungus such that individuals are largely free of *Pd* at summer roosts (Dobony et al. 2011, p. 193; Langwig et al. 2015b, p. 4). However, it is not uncommon for some bats to be found carrying viable *Pd* later into summer (Dobony et al. 2011, p. 193; Ineson 2020, p. 104) and *Pd* is capable of remaining viable in hibernacula without bats for extended periods (Lorch et al. 2013, p. 1298). The cool, humid conditions of hibernacula likely serve as environmental reservoirs for the fungal pathogen where it can survive and even proliferate until bats return in the fall (Reynolds et al. 2015, p. 320; Hoyt et al. 2020, p. 7259). Generally, bats return to winter roosts in the fall and engage in social interactions that lead to rapid spread of *Pd* from the environmental reservoir to the population (Hoyt et al. 2020, p. 7256). However, because hibernacula may be used throughout the year by males and nonreproductive females who hibernate there, as well as by other species that are more transient, including long distance migrants, some transmission is likely to occur year round and by other mechanisms.

Expansion of Pd in North America

Since it was first detected in New York, the range of *Pd* in North America has increased steadily via bat to bat transmission, although activities of humans, including scientific research, recreational activity, and shipping are also likely to contribute to some short and long distance movements (Bernard et al. 2020, p. 5–6). Simply, *Pd* has spread from just a small number of sites in New York in 2007 to hundreds of locations across the continent in just 14 years. Several predictive models have identified biological, geological, climatic, ecological and behavioral variables which are correlated with the patterns and timing of *Pd* spread (Hallam and Federico, 2012, p. 270; Maher et al. 2012, p. 3; Alves et al. 2014, p. 2; Hefley et al. 2020, pp. 10–11). Putative barriers to *Pd* expansion have been hypothesized, but these generally have provided very short-term delays in *Pd's* steady progression into uncontaminated areas (Miller-Butterworth et al. 2014, p. 9; Hoyt et al. 2021 p. 3). While these obstacles to natural disease spread may delay arrival of *Pd*, when the fungus does pass to them either via dispersing bats or via inadvertent transport by humans, it has led to disease and continued spread of the fungus (Miller-Butterworth et al. 2014, p. 9; Lorch et al. 2016, p. 4). Because the above published models have fallen behind reality in their predictions, we used two models to describe past occurrence of *Pd* and to predict its future expansion in North America (see *Figure A-2A4, methods described above).*

Establishment of Pd

With the arrival of *Pd* at a new location, progression of the disease proceeds similarly to many emerging infectious diseases through stages of invasion, epidemic, and establishment (Langwig et al. 2015a, p. 196; Cheng et al. 2021, p. 5). During *invasion* (years 0-1), the fungus arrives on a few bats and spreads through the colony until most individuals are exposed to and carry *Pd*. As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7) in the population. Then 8 years after its arrival, *Pd* is determined to be *endemic* (Langwig et al. 2015a, p. 196; Cheng et al. 2022, entire). Although methods for detecting *Pd* have changed over time, it is apparent with few exceptions that morbidity and mortality associated with WNS occurs within a year or two after *Pd* has been observed in a population (Frick et al. 2017, pp. 627–629; Hoyt et al. 2020 p. 7259). With the publication by Muller et al. (2013, entire), the use of polymerase chain reaction (PCR) to confirm the presence of *Pd* became the gold standard for diagnosing WNS. This technique provided greater confidence in *Pd* detection and improved our understanding of the disease progression.

Langwig et al. (2015b, pp. 3–4) and Hoyt et al. (2020, p. 7257) quantified the proportion of bats on which *Pd* is detected (prevalence) and the amount of *Pd* on bats (load) in the years after *Pd* invades and establishes itself in a site. In general, when *Pd* is first detectable (by PCR), a relatively small number of bats carry the fungus in low loads. These values increase throughout the first winter at varying rates among species. By the end of the first winter, *Pd* is detectable both on bats and on surfaces of the roost. In the second year after detection, *Pd* loads and prevalence pick up near where they were the previous year; prevalence and load are at significantly higher levels in the fall and early winter, and prevalence approaches 1 (i.e., all bats are infected) by mid-winter for TCB (Frick et al. 2017, p. 627).

There are a few exceptions in which evidence of *Pd* has been detected in a site and then not detected at that site in subsequent years. These occurrences may represent failed invasions by *Pd.* In Iowa, for example, molecular tests revealed evidence suggestive of *Pd* being present, but WNS was not confirmed at that location for several more years. In California, *Pd* has not been detected in two subsequent years after initial evidence was detected (Osborn 2021, pers. comm.). There are also examples that do not fit the expected disease progression described above. At Tippy Dam in Michigan, *Pd* has been present for over 5 years without indication of WNS in little brown bat, although northern long-eared bat are no longer observed at this location (Kurta et al. 2020, p. 584). The factors contributing to this atypical scenario are under investigation. It has also been posited that WNS may have a southern limit where disease is less likely to impact populations (Hallam and Federico 2012, p. 277; Hoyt et al. 2021, pp. 6–7). For example, TCB in the coastal and far southern portions of its range may use shallower torpor or engage in periodic foraging through the winter, thus avoiding severe disease (Bernard et al. 2017, p. 8; Newman, 2020, pp. 21–22). However, Sirajuddin (2018, p. 19) found that skin temperatures of TCB in the south does fall within the optimal range of growth for *Pd* during winter. Notably, *Pd* has been detected on bats overwintering in culverts in Mississippi and WNS has not manifested in the colony (Cross 2019, entire). Nevertheless, the overwhelming pattern has been that WNS develops in a population soon after the arrival of *Pd*. Still, because environmental reservoirs of the pathogen play an important role in its transmission, hibernacula that become unsuitable for *Pd* during summer (e.g., too warm or dry) may reduce the amount of fungus in the environment between hibernation seasons, leading to lesser or delayed development of WNS (Hoyt et al. 2020, pp. 7257–7258). To date, these exceptions where colonies experience less severe impacts from WNS compared to the majority of colonies are not predictable based on geographic or biological features.

Impacts of WNS

The impacts of white-nose syndrome are severe among species that were the first observed with the disease. This pattern has remained true over a large area as *Pd* has continued to expand its range affecting previously unexposed colonies of hibernating bats. Four years after the discovery of WNS, Turner et al. (2011) estimated total declines of 75% for TCB at WNS infected winter colonies in Vermont, New York, and Pennsylvania. Later, with data from six states (Vermont, New York, Pennsylvania, Maryland, Virginia, West Virginia), Frick et al. (2015) estimated that median colony size decreased by 90% and TCB was extirpated from 10% of historical hibernacula (Frick et al. 2015, p. 5). Hoyt et al. (2021, p. 7) summarized overall TCB declines from WNS to be 95% in the Northeast and 99% the Midwest. Using data from 27 states and 2 provinces, the most complete dataset available at the time, Cheng et al. (2021, p. 7) reported similar patterns. They estimated that WNS has caused a 90–100% decline in TCB across 59% of the range (Figure 4.4.; Cheng et al. 2021, p. 7). Although there are ecological and environmental differences across the currently affected regions of North America, WNS has consistently caused significant declines in TCB populations (Figure 4.6), with very few examples of colonies that are avoiding impacts (Figure 4.6).

Conservation Measures Associated with WNS

There are multiple national and international efforts underway in attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts.

Efforts associated with the national response to WNS were initially aimed at determining the cause of the disease and reducing or slowing its spread. The response broadened and was formalized by the *National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-nose Syndrome in Bats* which provides the strategic framework for implementation of a collaborative, national response to WNS by state, Federal, Tribal and non-governmental partners (USFWS 2011). The U.S. plan integrates closely with a sister plan for Canada, assuring a coordinated response across much of North America. Implementation of the WNS National Plan is overseen by executive and steering committees comprising representation from the Department of Interior, Department of Agriculture, Department of Defense, and State wildlife agencies under the authority of a multi-species recovery team under the ESA, with the USFWS serving the lead coordinating role. In 2021, the WNS National Plan is being revised to reflect current state of knowledge and identify key elements to continue to effectively respond to this disease. Goals and actions address the greatest needs and knowledge gaps to be pursued, including: coordinated disease surveillance and diagnostic efforts; inter-programmatic data management; development and implementation of disease management, conservation and recovery strategies; and communication and outreach among partners and with the public. These efforts are also supported by the North American Bat Monitoring Program (NABat), which is co-led by USGS and USFWS, to integrate data across jurisdictional borders in support of population level information that supports management decisions at different scales. Actions under the National Plan are intended to be supported through multiple funding programs in different agencies. For several years, many state, Federal, Tribal, and private partners have annually provided funding and physical efforts or both toward WNS research. For its part, the USFWS supports management activities of many partners, research to address key information needs, and development and application of management solutions. The USFWS maintains a website (www.whitenosesyndrome.org) and social media accounts to address many of the communication needs for both internal and external audiences.

Over 100 state and Federal agencies, Tribes, organizations and institutions are engaged in this collaborative work to combat WNS and conserve affected bats. Partners from all the states in TCB's range, Canada, and Mexico are engaged in collaborations to conduct disease surveillance, population monitoring, and management actions in preparation for or response to WNS.

B: Wind

Background

Wind power is a rapidly growing portion of North America's clean energy sector due to its small footprint, lack of carbon emissions, changes in state's renewable energy goals, and recent technological advancements in the field allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association (AWEA) 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often have tens or hundreds of turbines installed in a given area, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of October 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

Wind related mortality of TCB, while often overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and regional levels. The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[8] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat mortalities continue to be documented at wind power installations across North America.

Mechanism Behind Bat Mortality

Most bat mortality at wind energy projects is caused by direct collisions with moving turbine blades (Grodsky et al. 2011, p. 920; Rollins et al. 2012, p. 365). Barotrauma--a rapid air pressure change causing tissue damage to air-containing structures such as the lungs—may also contribute to bat mortality (Baerwald et al. 2008, pp. 695–696; Cryan and Barclay 2009, p. 1331; Rollins et al 2012, p. 368-369; Peste et al. 2015, p. 11), although impact trauma is likely the cause of most wind-related bat mortality (Lawson et al. 2020; entire). Grodsky et al. (2011, p. 924) further hypothesize that direct collision with turbine blades may cause delayed lethal effects (i.e., injured bats may leave the search area before succumbing to injuries; turbines may damage bats' ears, negatively affecting their ability to echolocate, navigate, and forage), thus causing an underestimation of true bat mortality.

Bats may be attracted to turbines (Solick et al. 2020, entire; Richardson et al. 2021, entire), though support for this is limited. Some hypotheses for bat attraction to wind turbines include the sound of moving blades, blade motion, insect aggregations near these structures, turbines as potential roost structures, and turbines as mating locations (Kunz et al. 2007, pp. 317-319, 321; National Research Council 2007, p. 97; Cryan and Barclay 2009, pp. 1334–1335; Cryan et al. 2014, p. 15128). Horn et al. (2008a, p. 14; 2008b, p. 126) observed bats flying within the turbine blade's rotor swept zone at wind projects in New York and West Virginia and noted that bats were actively feeding and foraging around moving and non-moving blades (2008b, p. 130), while Cryan et al. (2014, p. 15127) observed bats altering course towards turbines using thermal imagery.

Bat mortality tends to exhibit a seasonal pattern, with mortality peaking generally in the late summer and early fall (Erickson et al. 2002, p. 39; Arnett et al. 2008, p. 65; Taucher et al. 2012, pp. 25–26; Bird Studies Canada et al. 2018, pp. 28, 32, 33, and 46). Based on our analysis, 6.5%, 25.5%, and 68% of bat fatalities occur during the spring, summer, and fall periods, respectively (USFWS 2016, pp. 4-12–4-15). Temperature and wind speed may also indirectly influence bats collision risk with wind turbines. Bat activity is higher during nights of low wind speed and warmer temperatures (Arnett et al. 2006, p. 18), and is lower during periods of rain, low

[8] Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, pp. 32 and 69).

temperatures, and strong winds (Anthony et al. 1981, 154–155; Erkert 1982, pp. 201–242; Erickson and West 2002, p. 22; Lacki et al. 2007, p. 89).

Bat Mortality

Bat mortality varies across wind facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise most of all known bat fatalities at wind facilities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other bat species. However, there is notable spatial overlap between TCB occurrences and wind facilities and notable TCB mortality documented (Figure 4.7). Based on October 2020 installed MW capacity (Hoen et al. 2018, entire; USFWS unpublished data), we estimated 3,227 TCB are killed annually at wind facilities (Table 4.1; Figure A-2A6; Udell et al. 2022, pp. 265–266). Similarly, Whitby et al. (2022, entire) analyses suggest that the impact of wind related mortality is discernible in the ongoing decline of TCB. Comparing a no wind (and no WNS) baseline scenario to current and future wind (and no WNS) scenarios, the percent change in abundance relative to the baseline no wind scenario ranges from a 19–21% decrease by 2030 under the current wind scenario up to a 38% decrease by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, pp. 151–153) found a decline in the predicted relative abundance of TCB as wind energy risk index increased.

Conservation Measures

To reduce bat fatalities, some facilities "feather" turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein et al. 2021, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed or at a higher threshold, typically referred to as curtailment. The effectiveness of feathering below various cut-in speeds differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with raising cut-in speeds by 1.0–3.0 meters per second (m/s) above the manufacturer's cut-in speed (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for TCB has not been documented.

Our wind threat analysis incorporated available curtailment data for existing facilities, and to a limited degree, accounted for future curtailment (see Appendix A-2A). Although effective, curtailment results in energy and revenue losses, which may limit the viability of widespread implementation (Hein and Straw 2021, p. 28). Based on available data (USFWS, unpublished data), most current curtailment is implemented as part of Habitat Conservation Plans developed to support Incidental Take Permits or Technical Assistance Letters pursuant to the Endangered Species Act and detailing methods to avoid incidental take of Indiana bat; however, these areas with risk to Indiana bat do not fully overlap with those where TCB and other species may be susceptible to mortality.

There are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. For example, the use of ultrasonic acoustic bat deterrents mounted on turbine towers, blades, and nacelles is an emerging research field showing some promise at reducing bat fatalities (Arnett et al. 2013, entire; Romano et al. 2019, entire; Schirmacher et al. 2020, entire; Weaver et al. 2020, entire; Berthinussen et al. 2021, pp. 88–91). Acoustic-activated "smart" curtailment aims to focus operational curtailment when bat activity is detected in real time (e.g., Hayes et al. 2019, entire; Berthinussen et al. 2021, pp. 105–106; Hein and Straw 2021, pp. 29–30). Additionally, USGS is testing whether illuminating turbines with dim ultraviolet light may deter bats from approaching them (Cryan et al. 2016, entire; Berthinussen et al. 2021, p. 91; Hein and Straw 2021, pp. 23–24). Further, researchers have tested applying a textured coating to the surface of the turbine to alter bats' perception of the turbine (Bennett and Hale 2019, entire; Berthinussen et al. 2021, pp. 87–88; Hein and Straw 2021, p. 24). These and other methods of reducing bat mortality are still in the research phase, and to date, there are no broadly proven and accepted measures to reduce the severity of impacts beyond various operational strategies (e.g., feathering turbine blades when bats are most likely to be active).

C: Climate Change

Background

There is growing concern about impacts to bat populations in response to climate change (Jones et al. 2009, entire; Jones and Rebelo 2013, entire, O'Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats including changes in hibernation, mortality from extreme drought, cold, or rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed potential impacts of climate change on foraging, roosting, reproduction, and biogeography of bats and also discussed extreme weather events and indirect effects of climate change. However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). In particular, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success. While there may be a variety of ways that climate change directly or indirectly effects TCB, here we summarize information on the effect of increasing temperatures and changes in precipitation.

Increased Annual Temperature

Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., annual average temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986 to 2016 relative to 1901 to 1960 (Hayhoe et al. 2018, p. 86). At a regional scale, each National Climate Assessment region also increased in temperature during that time with the largest changes in the West with average increases of more than 1.5 degrees F (0.8 degrees C) in Alaska, the Northwest, the Southwest and the Northern Great Plans and the least change in the Southeast (Hayhoe et al. 2018, p. 86).

Increased annual temperatures are likely to change bat activity and phenology. For example, increased winter temperatures may reduce hibernation period due to longer fall activity or earlier spring emergence (Jones et al. 2009, p. 99). Rodenhouse et al. (2009, p. 250) suggest that hibernation may be shortened by 4 to 6 weeks by the end of this century. Reduced hibernation periods may decrease the duration that individual bats are exposed to *Pd* and effects from WNS (Langwig et al. 2015a, p. 5).

With increasing temperatures, earlier spring emergence has been documented for cave-roosting bats in Virginia (Muthersbaugh et al. 2019, p. 1). After earlier arrival to summer habitat, if spring weather remains favorable (warm, dry and calm nights providing suitable foraging conditions for bats), this could result in earlier parturition (Racey and Swift 1981, pp. 123–125; Jones et al. 2009, p. 99; Linton and MacDonald 2018, p. 1086) and increased reproductive success (Frick et al. 2010, p. 133; Linton and MacDonald 2018, p. 1086). However, earlier emergence increases the risk of exposure to lethal cold snaps in Spring (Jones et al. 2009, p. 99).

Increased temperatures may expand the suitable window for nightly foraging opportunities, thereby increasing per night caloric intake. Low ambient temperatures reduce flying insect activity and bat foraging (Anthony et al. 1981, p. 155), while higher average temperatures may result in more frequent suitable foraging nights, which is particularly important during the fall when bats are trying to accumulate extra body fat for winter hibernation.

Bats that hibernate in temperate regions require temperatures above freezing but cool enough to save energy through torpor (Perry 2013, p. 28). Increased ambient surface temperatures change hibernacula temperatures which then influences their ability to meet the needs of hibernating bats. However, increased ambient surface temperatures will not affect all hibernacula or all parts of a given hibernaculum equally. Hibernaculum microclimate is influenced by a variety of factors including the size, complexity, and location of the site (Tuttle and Stevenson 1977, pp. 109–113). In addition, temperatures of microsites near entrances are strongly correlated to external ambient temperatures compared to microsites deep within hibernacula (Dwyer 1971, p. 427; Boyles 2016, p. 21). Therefore, changes in ambient temperatures are anticipated to result in the greatest changes to portions of hibernacula nearest entrances. In Texas, external temperature had a greater influence on microclimate temperatures in culverts than in caves, likely as a result of culvert design (generally being straight, with two entrances to allow for air flow) and maintenance (clearing of brush around entrances), whereas not all caves have multiple entrances or may have additional barriers (vegetation) to reduce airflow (Leivers et al. 2019, p. 5). Overall, culverts and caves/mines with little complexity have greatest potential for being impacted by increasing external temperatures.

In warmer regions, caves and mines that trap cold air produce beneficial conditions for hibernacula, while in colder regions sites that trap warm air will be more suitable (Perry 2013, p. 33; Kurta and Smith 2014, p. 595). Consequently, a northern site that is suitable today in part for its ability to trap warm air while surface temperatures are very low may become unsuitable as mean annual surface temperature increases.

Indiana bats have been documented to use a wide variety of microclimates within hibernacula and Boyles (2016, p. 34) suggests that the most valuable caves for protection might be the ones with the widest variety of microclimates available. Briggler and Prather (2003, p. 411) similarly found that more TCB were found in caves with wide temperature gradients available. These more complex hibernacula will be less influenced by changes in surface ambient temperatures.

Variations in ambient temperature increase energy expenditure of hibernating bats (Boyles and McKechnie 2010, p. 1645); therefore, stable microsites may be advantageous (Johnson et al. 2021, entire). Increased ambient temperatures may reduce reliance on relatively stable temperatures associated with underground hibernation sites (Jones et al. 2009, p. 99). However, variation in ambient temperature (e.g., increased temperatures in the spring) may decrease the energetic costs of arousing from hibernation and serve as a signal that surface conditions are suitable for emergence and foraging (Boyles 2016, p. 36).

Increased hibernacula temperatures may influence overwinter survival rates. If more frequent bat arousals occur, bats will burn through fat reserves more quickly. While insect abundance may also increase in winter, it is unknown whether they will become sufficiently abundant to offset the increased energetic costs associated with more frequent arousal by bats (Rodenhouse et al. 2009, p. 251; Jones and Rebelo 2013, p. 464). Changes to hibernacula temperatures could potentially alter the severity of WNS in these sites (Martínková et al. 2018, p. 1747). For example, a hibernaculum with temperature below the optimal growth rate for *Pd* could shift into the optimal temperature range, thus increasing infection at the site.

Lastly, increased temperatures may result in range shifts of bats and forest communities, and increases in invasive species. With increasing temperatures, a poleward range expansion of temperate-zone species is predicted (Humphries et al. 2004, p. 154). Kuhl's pipistrelle (*Pipistrellus kuhlii*) has already undergone a substantial northward range shift over the past 15 years (Jones et al. 2009, p. 100), and Lundy et al. (2010, entire) suggested that the migratory Nathusius' pipistrelle (*Pipistrellus nathusii*) has expanded its range in the United Kingdom in response to climate change and will likely continue to expand. The ranges of European bats are forecasted to show considerable shifts, with species in the Boreal Zone experiencing the greatest change and risk of extinction (Rebelo et al. 2010, p. 568). Many species have little or no overlap between their current and predicted range and face enhanced extinction risk (Rebelo et al. 2010, p. 572).

Any northern range shifts, however, will be limited based on availability of suitable hibernacula and energetic requirements for hibernation and migration. Humphries et al. (2002, p. 315) predicted that minimum accumulated fat stores of little brown bats are currently inadequate for surviving hibernation throughout the northern portions of the Canadian provinces and the maximum possible fat stores are inadequate for most of Alaska and Canadian territories. When considering a predicted increase of 6 to 8 degrees C (10.8 to 14.4 degrees F), the region of suitable hibernation is expected to expand with a northward shift of approximately 6 km (3.7 mi) per year over the next 80 years (Humphries et al. 2002, pp. 315–316) (Figure A-4C1).

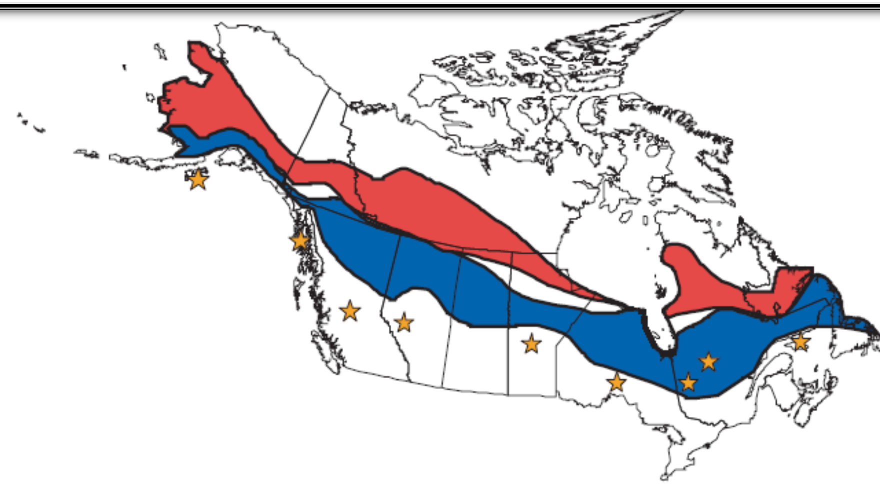

Figure 3 Observed and predicted *M. lucifugus* range distributions in northern North America. Confirmed northern locations of hibernacula (refs 10, 14, 19 and 29, and M. Gauthier, personal communication) are indicated with yellow stars, and the predicted distributional limit of hibernacula is indicated by the blue area; the width of the area represents variation in predictions resulting from cave temperatures exceeding average annual temperature by 0–2 °C, fat stores varying from 1.5 to 3.5 g, and all other parameters varying by up to ±5%. Thus, the lower margin of the blue zone indicates the limit beyond which to the south almost all bats could successfully hibernate, and the upper margin indicates the limit beyond which to the north almost no bats could successfully hibernate. Observed hibernacula approach and only marginally exceed the lower margin of this zone and there is a significant positive correlation between the latitude of observed hibernacula and the lower margin of this zone (predictions generated for each hibernaculum longitude, $n = 9$, $r = 0.93$, $P < 0.001$; intercept not significantly different from 0). The red area represents the predicted 2080 winter range limit of *M. lucifugus* due to global warming. Climate projections are based on the Hadley Centre coupled global climate model (HadCM2—GAX)[12]. The width of the red zone represents the same as does the blue zone.

Figure A-4C1. Observed and predicted little brown bat range distributions in northern North America (from Humphries et al. 2002, Figure 3).

While more northerly sites may become suitable for hibernation, there may be other constraints on successful recruitment at higher latitudes. The active season is shorter in higher latitudes or elevations which may be particularly important for juveniles. Juvenile little brown bats take longer than adults to gain sufficient fat stores for hibernation and shorter active seasons limit their capacity to grow and fatten before their first winter (Kunz et al. 1998, pp. 10–13; Humphries et al. 2002, p. 315). Higher elevations have similar climatic influences as higher latitudes and significantly fewer reproductive female little brown bats are captured at higher elevations in Pennsylvania, West Virginia and Virginia with a similar pattern for TCB in West Virginia (Brack et al. 2002, pp. 24–26).

While bats may be more flexible than other mammals in shifting their ranges, given their ability to fly, the ability of individuals to reach new climatically suitable areas will be impacted by loss and fragmentation of habitat (Thomas et al. 2004, p. 147). The availability of suitable roosts may be one of the most limiting resources for bats (Scheel et al. 1996, p. 453). This may be of special concern for tree-dwelling bats since the rate of climate change may be too fast to allow the development of mature forests in the new climatically suitable areas in the north (Rebelo et al. 2010, p. 573).

Changes in Precipitation

Increased temperatures interact with changes in precipitation patterns and results may differ regionally. Annual average precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the Southwest and Southeast (Easterling et al. 2017, p. 208; Hayhoe et al. 2018, p. 88) (Figure A-

4C2). The frequency and intensity of heavy precipitation events across the U.S. have increased more than increases in average precipitation (Hayhoe et al. 2018, p. 88).

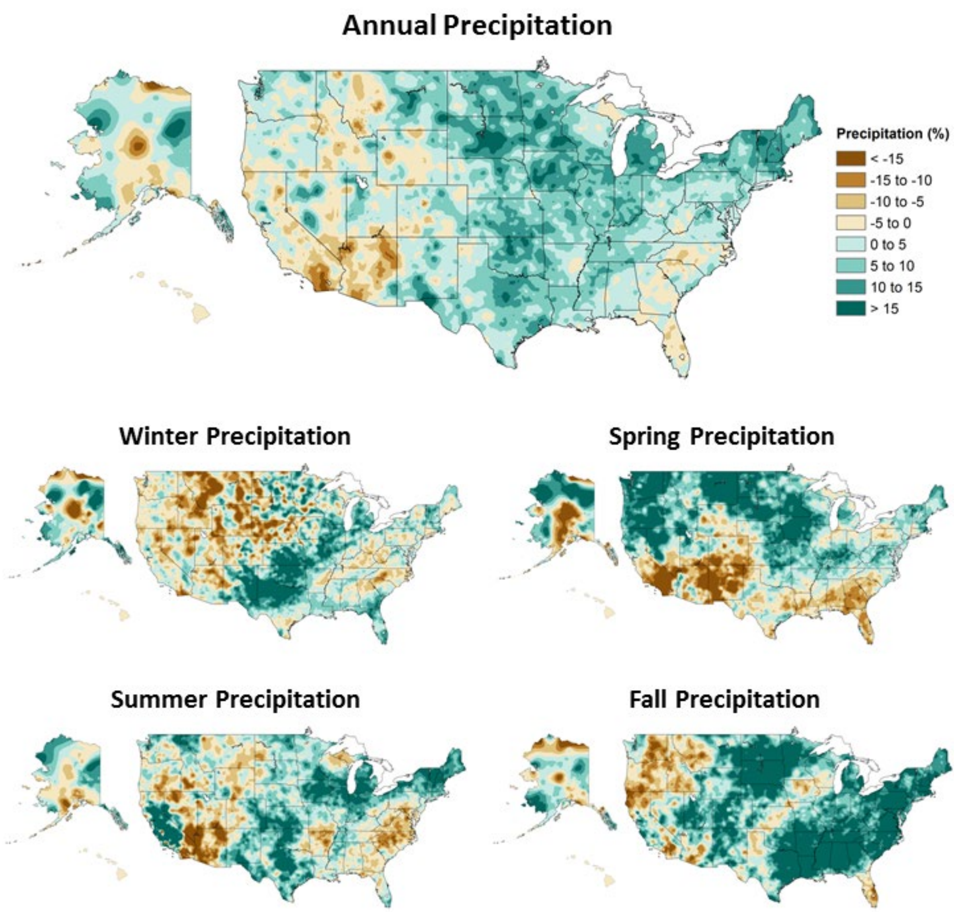

Figure A-4C2. Annual and seasonal changes in precipitation over the U.S. Changes are the average for present-day (1986–2015) minus the average for the first half of the last century (1901–1960 for the contiguous U.S., 1925–1960 for Alaska and Hawaii) divided by the average for the first half of the century (from Easterling et al. 2017, Figure 7.1).

In arid regions, any further reductions in water availability from human uses, reductions in snowpack, or droughts will amplify existing constraints. Spring snow cover extent and maximum snow depth has declined in North America and snow water equivalent and snowpack has declined in the western U.S. (Hayhoe et al. 2018, p. 90). Bats rely on access to free water for thermoregulation, foraging, and reproduction (Adams and Hayes 2008, pp. 1117–1119). In the Rocky Mountains, for example, drought and reduced standing water appears associated with decreased reproduction in bats (Adams 2010, entire). Years that were hotter and drier had a higher incidence of nonreproductive females (Adams 2010, pp. 2440–2442) (Figure A-4C3). While cooler and wetter springs resulted in shifts in parturition dates (Grindal et al. 1992, p. 342; Linton and MacDonald 2018, p. 1086), drought years resulted in an overall reduction in the percentage of bats that were reproductive <u>at all</u> (Adams 2010, p. 2442). Readily available water sources appear to be particularly important during lactation (Adams and Hayes 2008, pp. 1117–1120).

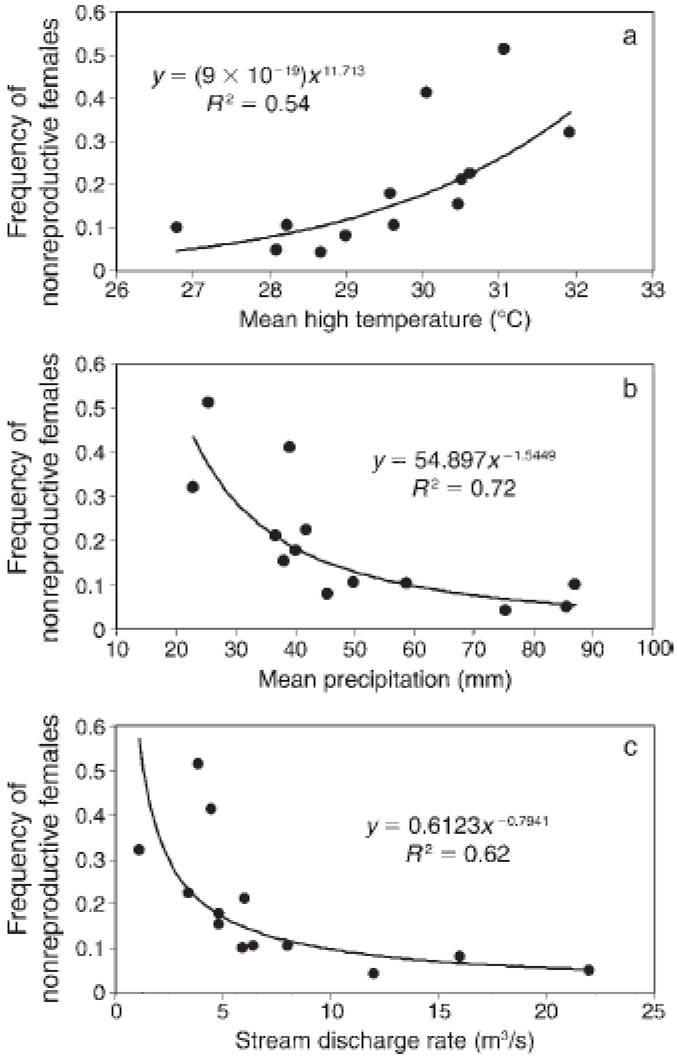

Figure A-4C3. Relationships between the frequency of nonreproductive females captured from 1996 through 2008 and (a) mean high temperature, (b) mean precipitation, and (c) stream discharge rate (from Adams 2010, Figure 2).

In temperate regions, increased cumulative annual rainfall may lead to increases in the abundance of insects such as dipterans and lepidopterans and is correlated with higher survival rates for the little brown bat (Frick et al. 2010, pp. 131–133). Frick et al. (2010, p. 133) suggest that increased insect abundance associated with higher moisture availability was the likely driver and this relationship may vary based on the timing of precipitation. Drying summer conditions may negatively impact aquatic insect prey and therefore, bats in the northeastern U.S. (Rodenhouse et al. 2009, p. 250; Frick et al. 2010, p. 133). Small mammals with high energy demands like bats, may be particularly vulnerable to changes in food supply (Rodenhouse et al. 2009, p. 250).

More precipitation has been falling as rain rather than snow in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 90). For example, increased winter temperatures are associated with decreases in Great Lakes ice cover and increases in winter precipitation occurring as rain. The extent and duration of lake ice on the Great Lakes are two of the principal factors controlling the amount of lake-effect snow (provided the air temperatures are sufficiently cool). When large areas of the lakes are covered with ice, the moisture cycle that generates lake-effect snow systems is greatly diminished (Brown and Duguay 2010, p. 692). During the first half of the 20th century there was an increase in snowfall in the Great Lakes Basin; however, recent studies have shown a decline through the latter half of the 20th and early 21st century (Baijnath-Rodino et al. 2018, p. 3947). Similarly, Suriano et al. (2019, pp. 4) found a reduction in snow depth in the Great Lakes Basin of approximately 25% from 1960–2009. Trends in snowfall and snow depth during this timeframe are variable by subbasin (Suriano et al. 2019, pp. 5–6) and there was a significant increase of the number of ablation events (i.e., snow mass loss from melt, sublimation, or evaporation) in many areas (Suriano et al. 2019, pp. 6–7). These events are associated with rapid snow melt and often lead to localized flooding. Hibernacula that already faced periodic flooding would be expected to have an increased risk in these areas.

While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During the precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795). Given these consequences, bats are likely to reduce their foraging bouts during these heavy rain events.

There is a balancing act that insectivorous bats perform, balancing the costs of flight, thermoregulation and reproduction versus energetic gains from foraging. When female bats arrive at maternity areas in the spring, they are stressed after a lengthy hibernation period, a potentially long migration, and the demands of early pregnancy. During this period when their energetic and nutritional requirements are highest, food (flying insects) is relatively scarce, due to cool and wet weather (Kurta 2005, p. 20). Adverse weather, such as cold spells, increases energetic costs for thermoregulation and decreases availability of insect prey (the available energy supply). Bats may respond to a negative energy balance by using daily torpor which conserves consumed and stored energy, and probably minimizes mortality. This has significant implications for their survival or reproduction.

Also, as mentioned above, increased rainfall during pregnancy and lactation may delay parturition or reduce reproductive success (Racey and Swift 1981, pp. 123–125; Grindal et al. 1992, p. 128; Burles et al. 2009, pp. 135–136; Linton and MacDonald 2018, p. 1086). Some females may not bear pups in years with adverse weather conditions (Barclay et al. 2004, p. 691). Young bats who are born and develop later in the season have less time to develop to successfully forage and to build the fat stores needed to meet the energy demands of migration and hibernation (Humphrey 1975, p. 339). Frick et al. (2010, pp. 131–132) found that little brown bats born even a few weeks later in the summer have significantly lower first-year survival rates and are significantly less likely to return to the maternity colony site to breed in their first year.

Early in the summer, females are under heavy energy requirements to supply their developing fetuses. After giving birth, the adult females experience increased energy needs due to the requirements of lactation and the need to return to the roost during night foraging times to feed their nonvolant pups (Murray and Kurta 2004, p. 4).

Later in the summer as the pups become volant, these inexperienced and relatively inefficient flyers must expend increased levels of energy as they are growing and learning to feed. Once weaned, young-of-the-year bats must consume enough on their own to migrate to hibernacula and store sufficient fat for the coming winter.

Interaction with WNS-affected Bats

Regardless of the source of increased stress (e.g., reduced foraging, reduced free standing water), because of WNS, there are additional energetic demands for bats. Because WNS causes premature fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, pp. 2–3) and have wing damage (Meteyer et al. 2009, p. entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging while also bringing the energetic cost of healing (Davy et al. 2017, p. 705; Fuller et al. 2020, p. 8; Meierhofer et al. 2018, p. 487).

Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. Bats may use torpor to conserve energy during cold, wet weather when insect activity is reduced and increased energy is needed to thermoregulate. However, use of torpor reduces healing opportunities as immune responses are suppressed (Field et al. 2018, p. 3731).

Dobony et al. (2011, entire) observed a little brown bat colony prior to and after onset of WNS impacts and found evidence of lower reproductive rates in the years immediately after WNS was first documented to affect the colony. Francl et al. (2012, p. 36) observed a reduction in juveniles captured pre- and post-WNS in West Virginia, suggesting similarly reduced reproductive rates. Meierhofer et al. (2018, p. 486) found higher resting metabolic rates in WNS-infected (vs. uninfected) little brown bats suggesting additional energy costs during spring in WNS survivors.

Future climate conditions

Over the next few decades, annual average temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C), relative to 1985–2015 regardless of future scenario (Hayhoe et al. 2018, p. 86; Figure A-4C4). Larger increases are projected by late century of 2.3–6.7 degrees F (1.3–3.7 degrees C) under RCP4.5 and 5.4–11.0 degrees F (3.0–6.1 degrees C) and 5.4 to 11.0 degrees F (3.0–6.1 degrees C) under RCP8.5, relative to 1986–2015 (Hayhoe et al. 2018, p. 86). For the period of 2070–2099 relative to 1986–2015, precipitation increases of up to 20–30% are projected in winter and spring for northcentral U.S. and Alaska, respectively, with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al.

2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

TCB's response to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as the little brown bat (see above). This includes reduced reproduction due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years in the Northeast (Frick et al. 2010, pp. 131–133). However, the timing of rain events is also important as reduced reproduction has been observed during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will likely vary throughout TCB's range and on how much the annual temperature actually rises in the future.

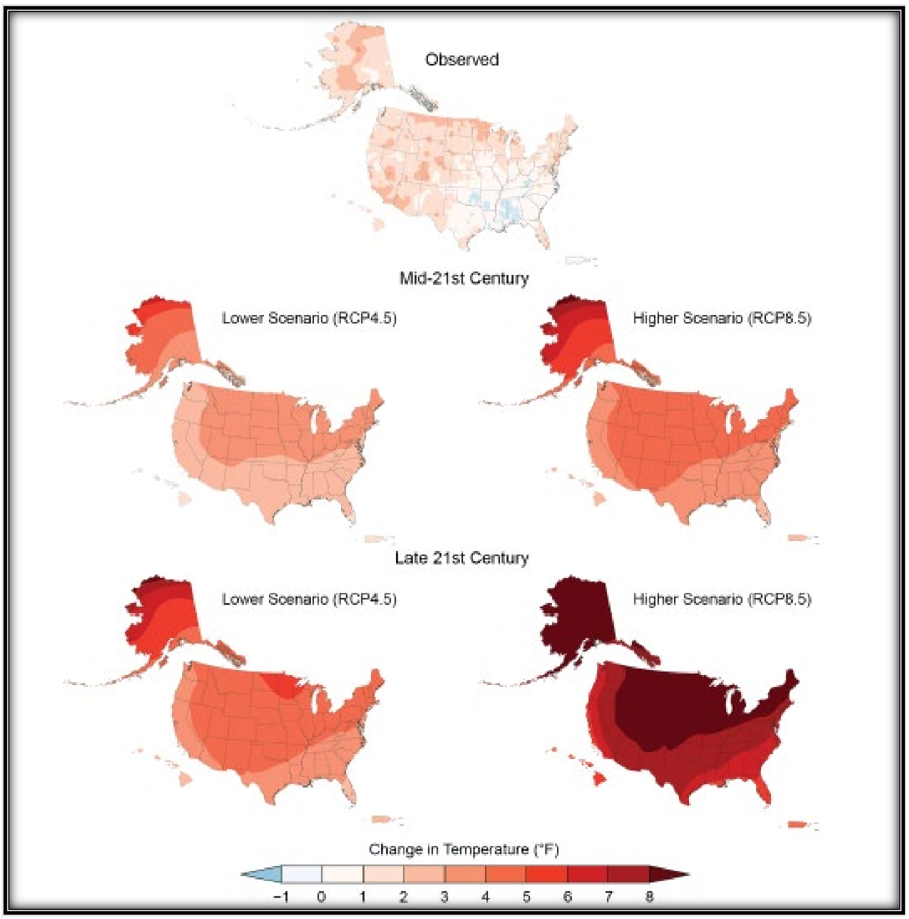

Figure A-4C4. Observed and Projected Changes in Annual Average Temperature (from Hayhoe et al. 2018, Figure 2.4).

Climate change may additionally impact TCB in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid, southern pine beetle) (Haavik 2019).

Climate change has also resulted in a rise of global sea level by about 7 to 8 inches (16 to 21 centimeters) since 1993 and relative to the year 2000, sea level is very likely to rise 1 to 4 feet (0.3 to 1.3 meters) (Hayhoe et al. 2018, p. 83). Relative sea level rise is projected to be greater than the global average along the coastlines of the Northeast and western Gulf of Mexico (Hayhoe et al. 2018, p. 99), which may reduce access to cave roosts along low-lying coastal areas (Jones et al. 2009, p. 101).

Additionally, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of TCB's range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success.

D: Habitat Loss

Background

As discussed in Chapter 2, TCB require suitable forest habitat for roosting, foraging, and commuting between those habitats during spring, summer, and fall. Wetlands and water features are also important for foraging and serve as drinking water sources. There are a variety of reasons for roosting, foraging, and commuting habitat loss within the range of TCB. Hammerson et al. (2017, entire) assessed the scope and severity of threats to bats and determined the highest projected threats included: invasive species and diseases (particularly WNS); energy production and mining, especially wind energy; human intrusions and disturbance of primarily cave- or mine-dwelling species; and biological resource use, such as forest conversion. Tree cutting and wetland loss can occur from a variety of sources (e.g., development, energy production and transmission, transportation projects). These activities leading to the loss of roosting and foraging habitat are increasing across TCB's range (USFWS 2015, p. 17991; Oswalt et al. 2019, p. 17) and may result in impacts to TCB.

Past and Current Habitat Loss

The USFS (2014, p. 7) summarized U.S. forest trends and found a decline from 1850 to the early 1900s, and a general leveling off since that time; therefore, conversion from forest to other land cover types has been fairly stable with conversion to forest (cropland reversion/plantings). In addition, the USFS reviewed U.S. forest trends through 2017 and found forest area trended upward from 1987–2012, but since 2012 appears to have reached a plateau (2019, p. 4).

In addition to reviewing these reports, we examined more recent (2006–2016) changes in various NLCD landcover classes within each RPU in the continental U.S. Forest landcover increased overall (primarily based on increases in coniferous forest in the Southern RPU). However, deciduous forest landcover decreased across all RPUs by 768,903 ha (1,900,000 ac) for an average loss of 76,890 ha (190,000 ac) per year and coniferous forest decreased in both the Northern and Eastern RPUs (Table A-4D1). Other cover types that provide foraging opportunities such as emergent wetland cover types also decreased across all RPUs by 687,966 ha (1,700,000 ac).

Table A-4D1. Changes in land cover types in acres (NLCD 2006-2016) by TCB RPU.

NLCD Lower 48 2006-2016	**TCB Representation Units – Δ Acres**			
Land Cover Type	**Northern**	**Eastern**	**Southern**	**All Units**
No Data	0	0	0	0
Open Water	403201	-54478	158744	507467
Developed, Open Space	227144	47603	328783	603530
Developed, Low Intensity	305072	80143	493413	878628
Developed, Medium Intensity	397935	106263	580635	1084833
Developed, High Intensity	176968	43008	223158	443134
Barren Land	28274	-6241	-2119	19913
Deciduous Forest	-299557	-464127	-1195633	-1959317
Evergreen Forest	-151629	-18069	1778105	1608407
Mixed Forest	335074	-15987	140023	459110
Shrub/Scrub	490801	386026	-1212135	-335309
Grassland/Herbaceous	-3632147	25255	-543963	-4150855
Pasture/Hay	-3257867	-253408	-1459287	-4970562
Cultivated Crops	5070264	90132	848862	6009257
Woody Wetlands	777936	74636	645345	1497918
Emergent Herbaceous Wetlands	-871468	-40755	-783931	-1696154
Forest change over 10 years	661824	-423547	1367840	1606117

Forest ownership varies widely across the species' range in the U.S. As of 2017, private landowners owned approximately 60% of forests (Oswalt et al. 2019, p. 7). Private lands do not carry the same level of regulatory certainty as do Federal lands, a factor that must be considered when assessing risk of forest loss now and in the future (USFWS 2015, p. 17990). Private land ownership is approximately 81% in the East and 30% in the western U.S. (USFS 2014, p. 15).

Future Habitat Loss

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources. This assessment was influenced by a set of future scenarios with varying assumptions with regard to global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010–2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million ha (16–34 million acres or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million ha (9–21 million acres) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through approximately 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8-7). However, TCB are found roosting in mature forest stands significantly more often than in younger stands given that regenerating young forests lack the structural diversity preferred by roosting TCB (e.g., broken tree branches with dead leaf clusters) (Veilleux et al. 2003, p. 1072; Perry and Thill 2007, p. 978; Thames 2020, pp. 32–34). In addition, where roosting and foraging habitat is removed, impacts are greater to the species where the species has been impacted by WNS.

Impacts to bats

Forest removal may result in the following impacts to TCB: loss of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging due to habitat fragmentation of remaining forest patches, fragmentation of maternity colonies due to removal of travel corridors, and direct injury or mortality (during active season tree removal).

Loss of roosts →death or injury
TCB may be directly affected by forest habitat loss by removal of occupied roost trees (Belwood 2002, p. 193; McAlpine et al. 2021, p. 2) or loss of roosting and foraging habitat (Farrow and Broders 2011, p. 177). While roosting bats can sometimes flee during tree removal, removal of occupied roosts is likely to result in direct injury or mortality to some bats (McAlpine et al. 2021, p. 2). This is particularly likely during cool spring months (when bats enter torpor) and if flightless pups or inexperienced flying juveniles are also present.

Loss of roosts → colony fragmentation → smaller colonies → reduced thermoregulation, reduced information sharing → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival
- reduced adult survival

Loss of roosts, foraging habitat, or travel corridors → displacement → increased flights → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival
- reduced adult survival

Although loss of a roost is a natural occurrence that temperate bat species must cope with regularly due to the ephemeral nature of tree roosts, the loss of many roosts or an entire home range may result in impacts at the colony level. Bats switch roosts for a variety of reasons, including temperature, precipitation, predation, parasitism, sociality, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264). TCB are known to switch roosts (Veilleux and Veilleux 2004a, p. 197; Quinn and Broders 2007, p. 19; Poissant et al. 2010, p. 374); therefore, TCB likely can tolerate some loss of roosts, provided suitable alternative roosts are available (see Chapter 2). However, loss of central or important roosts has caused colony fragmentation in the northern long-eared bat. For example, Silvis et al. (2015, pp. 6–12) found a loss of approximately 17% of roosts may begin to cause colony fragmentation; however, we have no additional information specific to TCB. One of the most prominent advantages of colonial roosting is the thermoregulatory benefit (Humphrey et al. 1977, pp. 343–344; Kurta et al. 1996, entire). Therefore, smaller colonies are expected to provide fewer thermoregulatory benefits for adults in cool spring temperatures and for nonvolant pups at any time.

If bats are required to search for new roosting or foraging habitat and to find the same habitats as the rest of their colony finds in the spring, it is reasonable to conclude that this effort places additional stress on pregnant females at a time when fat reserves are low or depleted and they are already stressed from the energy demands of migration and pregnancy. In addition, removal of roosting or foraging habitat may result in longer travel distances between sites used for roosting and foraging. The increased energetic cost of longer commuting distances may result in maternity colony disruption and may be particularly important for pregnant and lactating females (Lacki et al. 2007, p. 89) and therefore, reproductive success. TCB emerge from hibernation with their lowest annual fat reserves and return to their summer home ranges. Loss or alteration of roosting or foraging habitat puts additional stress on species with strong summer site fidelity (Allen 1921, p. 54; Veilleux and Veilleux 2004a, p. 197). Reproduction is one of the most energetically demanding periods for temperate-zone bats (Broders et al. 2013, p. 1174). Female TCB produce a maximum of two pups per year; therefore, loss of just two pups results in loss of that entire year's recruitment for females. Limited reproductive potential severely limits the ability of bat populations to respond quickly to perturbations.

Interaction with WNS-affected Bats

Similar to climate change, there are interacting effects of habitat loss with effects from WNS. Regardless of the source of increased stress on bats (roost or foraging habitat removal), because of WNS, there are additional energetic demands for bats associated with healing (Fuller et al. 2020, p. 7). Because WNS causes more frequent arousals (Reeder et al. 2012, pp. 6–9) and fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, p. 7001) and have wing damage (Meteyer et al. 2009, entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging. Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. With increased flights to find suitable habitat or between roosting and foraging habitat comes a trade-off for sufficient energy for survival, recovering from WNS, successful pregnancy or successful rearing of pups.

Roosting/Foraging/Commuting Habitat Loss Conservation Measures

All states have active forestry programs with a variety of goals and objectives. Several states have established habitat protection buffers around known Indiana bat hibernacula that will also serve to benefit TCB by maintaining sufficient quality and quantity of swarming habitat. Some states conduct some of their own forest management activities in the winter within known federally listed endangered and/or threatened bat home ranges, as a measure that would protect maternity colonies and nonvolant pups during summer months. The USFWS routinely works with project sponsors and Federal agencies to minimize the amount of forest loss associated with their projects and to provide mitigation for impacts associated with forest loss within the range of the federally listed Indiana bat. Examples of largescale efforts to address impacts associated with habitat loss include: rangewide transportation consultation for Indiana bats and northern long-eared bat[9], NiSource Habitat Conservation Plan[10], and rangewide in-lieu fee program for Indiana bats. Many of the beneficial actions associated with these and similar efforts may benefit TCB if they occur in overlapping ranges.

Depending on the type and timing of activities, forest management can be beneficial to bat species (e.g., maintaining or increasing suitable roosting and foraging habitat). Forest management that results in heterogeneous (including forest type, age, and structural characteristics) forest habitat appears to benefit North American tree roosting bats (Silvis et al. 2016, p. 37). For example, creation of small canopy openings could increase solar exposure to roosts, leading to warmer conditions that result in more rapid development of young (Perry and Thill 2007, p. 224). Preserving mature forest habitats should allow for increased roosting opportunities (Veilleux et al. 2003, p. 1072; Perry and Thill 2007, p. 978; Thames 2020, pp. 32–34) which may increase survival or reproductive success. Consequently, we should continue to pursue tried and true management approaches, such as providing for the continual recruitment of mature forest in landscapes with a variety of well-connected forested habitat types.

Summary

In summary, U.S. forest area trends have remained relatively stable with some geographic regions experiencing more forest loss than others in the recent past. In the future, forest loss is expected to continue, whether from commercial or residential development, energy production, or other pressures. Impacts from forest habitat removal to individuals or colonies would be expected to range from minor (e.g., removal of a small portion of foraging habitat in largely forested landscapes with robust TCB populations) to significant (e.g., removal of roosting habitat in highly fragmented landscapes with small, disconnected populations). In areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), impacts from forest removal would likely be higher given decreased roosting opportunities and potential loss of connectivity between roosting and foraging habitat.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, TCB require hibernation sites with specific microclimates and TCB exhibit high interannual fidelity to their hibernacula. Therefore, the complete loss of or modification of winter roosts (such that the site is no longer suitable) can result in impacts to individuals or at the population level. In addition, disturbance within hibernacula can render a site unsuitable or can pose harm to individuals using the site. Human entry or other disturbance to hibernating bats results in additional arousals from hibernation which require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable.

[9] https://www.fws.gov/midwest/endangered/section7/fhwa/index.html
[10] https://www.fws.gov/midwest/endangered/permits/hcp/nisource/

Modifications to bat hibernacula (e.g., erecting physical barriers to control cave and mine access, intentional or accidental filling or sealing of entries, or creation of new openings) can alter ability of bats to access the site (Spanjer and Fenton 2005, p. 1110) or affect the airflow and alter microclimate of the subterranean habitat, and thus the ability of the cave or mine to support hibernating bats, such as TCB. These well-documented effects on cave-hibernating bat species were discussed in the USFWS's *Indiana Bat Draft Recovery Plan* (USFWS 2007, pp. 71–74). In addition to altering the thermal or humidity regime and ability of the site to support hibernating bats, bats present during any excavation or filling can be crushed or suffocated. Sources of these stressors include fill from adjacent activities, mining, and intentional closures of abandoned mines or cave openings to restrict access.

Conservation Measures Addressing Hibernacula Loss and Disturbance

Protecting TCB from disturbance during winter is essential because increased arousals from hibernation require greater energy expenditures at a time when food and water resources are scarce or unavailable. This is even more important for hibernacula impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Boyles and Willis 2010, p. 96).

One method of reducing disturbance at bat hibernacula is through installation of bat-friendly gates that allow passage of bats while reducing disturbance from human entry as well as avoiding changes to the cave microclimate from air restrictions (Kilpatrick et al. 2020, p. 6). Many state and Federal agencies, conservation organizations, and land trusts have installed bat-friendly gates to protect important hibernation sites. The National Park Service has proactively taken steps to minimize effects to underground bat habitat resulting from vandalism, recreational activities, and abandoned mine closures (Plumb and Budde 2011, unpublished data). Further, the USFS has closed hibernacula during the winter hibernation period, primarily due to the threat of WNS, although this will reduce disturbance to bats in general inhabiting these hibernacula (USFS 2013, unpaginated). Because of concern over the importance of bat roosts, including hibernacula, the American Society of Mammalogists developed guidelines for protection of roosts, many of which have been adopted by government agencies and special interest groups (Sheffield et al. 1992, p. 707). Also, regulations, such as the Federal Cave Resources Protection Act (16 U.S.C. 4301 *et seq*.), protects caves on Federal lands. Finally, many Indiana bat hibernacula have been gated and some have been permanently protected via acquisition or easement, which provides benefits to other bats that use these sites for hibernation.

Appendix 5. Supplemental Future Scenario Descriptions

A summary of the low and high impact scenarios is described below and summarized in Table A-5.1.

Table A-5.1. TCB composite plausible future scenarios.

Plausible Scenario	*Pd* Occurrence Model	WNS Impact Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition
Low impact	*Pd* Occurrence Model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	Regional-specific
High impact	*Pd* Occurrence Model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	Regional-specific

WNS

For current projections, we used the two *Pd* occurrence models (see Appendix 2) to assign a WNS stage to all known hibernacula. Table A-5.2 provides the current (2020) number of winter colonies in each of the five WNS stages.

Table A-5.2. Number of TCB colonies in 2020 per WNS stage under Pd occurrence models 1 and 2.

Model	Pre-arrival	Invasion	Epidemic	Established	Post-established
Pd occurrence model 1	0 (0%)	32 (2%)	421 (22%)	756 (39%)	738 (38%)
Pd occurrence model 2	286 (15%)	63 (3%)	324 (17%)	271 (14%)	997 (51%)

The difference between the low and high impact scenarios is based on past year of arrival of *Pd* and future rate of *Pd* spread. We used *Pd* Occurrence model 1 (Wiens et al. 2022, pp. 226–229) in our low impact scenario and *Pd* Occurrence model 2 (Hefley et al. 2020, entire) in our high impact scenario. As *Pd* expands its range, we expect bat populations to be impacted similarly across the species' range. Thus, we apply the same WNS impacts schedule in low and high impact scenarios. Each hibernaculum's population abundance trajectory is divided into three segments with differing λ values: a pre-*Pd*-arrival λ typically ≳1, a *Pd*-arrival λ typically <1, and a post-established λ that can be less than, greater than, or approximately equal to 1. From years since arrival (YSA) 0–6, λ varied annually based on results of the status and trends model. We used site specific estimates to the extent possible, although relatively few colonies had sufficient data from counts more than 6 YSA. Therefore, for YSA>6, λ was estimated as the average predicted rate of change in that time period and is held constant through YSA=15 (low impact scenario) and through YSA=40 (high impact scenario). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts under the high impact scenario to 40 years (i.e., the time throughout which WNS will affect survival in the population). To understand the sensitivity of the results to the duration of the disease dynamic and to fully capture the uncertainty, we used the shortest reasonable disease dynamic duration in the low impact scenario. Based on current data (i.e., data from hibernacula documented with WNS in 2008 continue to show impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect populations after *Pd* arrives. After YSA=15 (low impact) or YSA=40 (high impact), λ is assumed to return to pre-WNS rates (i.e., no further WNS impacts applied).

Wind

U.S. Current and Future Wind Capacity

We obtained current wind capacity data for the U.S. from the USWTDB (version 3.2; Hoen et al. 2018, entire) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we considered projections for 2030, 2040 and 2050 from 4 potential sources: (1) the U.S. Department of Energy (USDOE) April 2015 Wind Vision report (USDOE 2015) and downloadable data; (2) the U.S. Energy Information Administration (USEIA) January 2020 Annual Energy Outlook (AEO) report (USEIA 2020) and downloadable data; (3) the USFWS April 2016 Draft Midwest Wind Multi-Species Habitat Conservation Plan (USFWS 2016); and (4) the National Renewable Energy Laboratory (NREL)'s 2020 Standard Scenarios Report (Cole et al. 2020, entire) and downloadable data.

After exploring these data sets and their stated purposes and underlying assumptions and consulting with experts from the USEIA, USDOE, and NREL, we ultimately decided that the NREL Standard Scenarios would serve best for the purposes of our analysis. According to the Standard Scenarios report, it is "*one of a suite of National Renewable Energy Laboratory (NREL) products aiming to provide a consistent and timely set of technology cost and performance data and define a scenario framework that can be used in forward-looking electricity analyses by NREL and others. The long-term objective of this effort is to identify a range of possible futures for the U.S. electricity sector that illuminate specific energy system issues. This is done by defining a set of prospective scenarios that bound ranges of technology, market, and macroeconomic assumptions and by assessing these scenarios in NREL's market models to understand the range of resulting outcomes, including energy technology deployment and production, energy prices, and emissions*" (Cole et al. 2020, p. iii).

In addition to a Mid-case Scenario, which uses the reference, mid-level, or default assumptions for all scenario inputs, represents a reference case, and provides a useful baseline for comparing scenarios and evaluating trends, the NREL's 2020 report presents 46 power sector scenarios for the contiguous U.S. (CONUS) that consider the present day through 2050. The NREL report notes, "*the Standard Scenarios are not "forecasts," and we make no claims that our scenarios have been or will be more indicative of actual future power sector evolution than projections made by others*"(Cole et al. 2020, p. 1); however, our experts advised that although the NREL report doesn't calculate a level of probability associated with any given scenario, the Mid-case Scenario is a justifiably reasonable baseline scenario for future wind deployment to use in our analysis.

After further exploring the NREL Standard Scenarios data, we discussed with USDOE and NREL experts the option of using high and low deployment bounds rather than, or in addition to, a reasonable central projection (i.e., Mid-case Scenario). Our experts agreed that this approach would help to capture some of the uncertainty associated with modeled projections; however, we were cautioned not to simply use the lowest and highest deployment scenarios since some scenarios might best be thought of as edge cases intended to show the sensitivity of the model to tweaks in assumptions rather than realistic characterizations of future deployment. Instead, we were advised to use the High and Low Wind Cost Scenarios as a reasonable combination of scenarios for our SSA analysis, and ultimately decided to apply them as lower and upper bounds, respectively, for the U.S. projections.

The Mid-case, High Wind Cost, and Low Wind Cost Scenarios each implement a slightly different set of assumptions for electricity demand, fuel prices, electricity generation and technology costs, financing, resource and system conditions and more. Under the High Wind Cost Scenario (our lower bound or "Low Build-out Scenario"), other energy technologies become more cost competitive compared to new wind energy facilities or repowering existing sites. As wind turbines reach their end of life, more are retired than are replaced with newer machines, condensing where wind energy is deployed to only the most optimal sites that present the fewest barriers and the greatest return on investment (Straw 2021, pers. comm.). Therefore, under this scenario, the distribution of wind turbines across the species' range by 2050 is reduced compared to 2020 build-out and total wind capacity decreased for several regions (Table A-5.3), although total U.S. wind capacity is projected to increase slightly. Under the Low Wind Cost Scenario (our upper bound, or "High Build-out Scenario), repowering existing wind energy facilities or installing new wind facilities is more cost competitive compared to other energy technologies, resulting in a broader future distribution of wind turbines across the U.S. and higher overall capacity compared to 2020 build-out (Table -5.3, Figures 4.9-4.11). For a summary of input assumptions used in the Standard Scenarios see Appendix A.1 from the 2020 Standard Scenarios report (https://cambium.nrel.gov/). We assumed total curtailed MW per NREL grid cell would remain unchanged into the future unless MW capacity declined; in these cases, we reduced grid cell curtailment proportionally (e.g., if MW capacity is projected to decline from 10 to 1 MW and currently there is curtailment on 9 MW, there would be 0.9 MW with curtailment and 0.1 MW without curtailment; Udell et al. 2022, entire).

Canada Current and Future Wind Capacity

We obtained current wind capacity data for Canada from the Canada Wind Turbine Database (CWTD). To obtain current and future wind capacity for Canada, the SSA wind team considered current buildout and projections for 2030, 2040 and 2050 from two sources: (1) The Canadian Wind Energy Association (CanWEA); and (2) The Canada Energy Regulator (CER) Canada's Energy Future 2019 Report (CER 2019). We decided that the CanWEA data would not serve well for our analysis because adequate projections were lacking through the future decades (2020–2050) for most provinces as well as the entire country.

The CER Canada's Energy Future 2019 (EF 19) report is an annual report published by the Government of Canada starting in 2013 and presents projections for wind energy buildout and future capacity through 2040 through updated baseline projections from previous years. According to the report "*the Energy Futures series explores how possible energy futures might unfold for Canadians over the long term. Energy Futures uses economic and energy models to make these projections. They are based on assumptions about future trends in technology, energy and climate policies, energy markets, human behavior and the structure of the economy.*" The baseline projections EF 19 are based on one future projection scenario called the Reference Case. According to the report, the Reference Case is "*based on a current economic outlook, a moderate view of energy prices and technological improvements, and climate and energy policies announced and sufficiently detailed for modeling at the time of analysis*" (CER 2019, p. 1).

After we had selected the EF 2019 data for our analysis, the CER published an updated report (EF 20 report) in November 2020 (CER 2020). Similar to previous reports, the EF 20 report presents projections for wind energy buildout and future capacity through updated baseline projections from previous years. Unlike its predecessors, the EF 20 projects buildout scenarios through 2050, ten years longer than previous years. Additionally, unlike previous reports, the EF 20 Report analyzes two buildout scenarios rather than one: the Evolving Scenario and the Reference (baseline) Scenario. According to the report, the Evolving Scenario "*considers the impact of continuing the historical trend of increasing global action on climate change throughout the projection period. Globally, this implies lower demand for fossil fuels, which reduces international market prices. Advancements in low carbon technologies lead to improved efficiencies and lower costs. Within Canada, we assume a hypothetical suite of future domestic policy developments that build upon current climate and energy policies*". The 2020 Reference Scenario "*provides an update to what has traditionally been the baseline projection in the Energy Futures series, the Reference Scenario. The scenario considers a future where action to reduce GHG emissions does not develop beyond measures currently in place. Globally, this implies stronger demand for fossil fuels, resulting in higher international market prices compared to the Evolving Scenario. Low carbon technologies with existing momentum continue to improve, but at a slower rate than in the Evolving Scenario*" (CER 2020, p. 4).

In addition to being more up-to-date than the 2019 data, the dual buildout scenarios included in the 2020 update presented an opportunity to analyze a range of scenarios rather than a single projection and set of assumptions. Therefore, we assigned the Evolving Scenario as an upper bound buildout scenario and the Reference Scenario as a lower bound scenario for our analysis.

Table A-5.3. Wind capacity (MW) by USFWS Region and Canadian Province under 2020 and 2050 low and high scenario build-out.

Location	**Wind Capacity (MW)**		
	2020 Build-out	**2050 Low Build-out (% change)**	**2050 High Build-out (% change)**
Region 3	27,387	15,198 (-45%)	141,573 (+417%)
Region 6	21,280	40,944 (+92%)	83,033 (+290%)
Region 5	6,116	7,252 (+19%)	68,946 (+1027%)
Region 1	7,459	1,422 (-81%)	19,102 (+156%)

Location	Wind Capacity (MW)		
Region 8	2,466	1,414 (-43%)	20,624 (+736%)
Region 4	240	391 (+63%)	38,083 (+15768%)
Region 2	39,964	40,511 (+1%)	116,346 (+191%)
U.S. Total	104,912	107,132 (+2%)	487,707 (+365%)
Alberta	1,746	6,699 (+284%)	10,286 (+489%)
British Columbia	732	1,252 (+71%)	1,967 (+169%)
Manitoba	258	476 (+85%)	851 (+230%)
Ontario	5,436	5,646 (+4%)	12,300 (+126%)
Quebec	4,330	5,830 (+35%)	6,930 (+60%)
Atlantic Canada	873	1,408 (+61%)	2,394 (+174%)
Saskatchewan	221	3,256 (+1373%)	5,781 (+2516%)
Canada Total	13,597	24,569 (+81%)	40,510 (+198%)
U.S. + Canada	118,509	131,701 (+11%)	528,217 (+346%)

ADDITIONAL REFERENCES CITED IN APPENDICES

Adams, R.A. and M.A. Hayes. 2008. Water Availability and Successful Lactation by Bats as Related to Climate Change in Arid Regions of Western North America. Journal of Animal Ecology 77(6):1115–1121.

Alves, D.M.C.C, L.C. Terribile, and D. Brito. 2014. The Potential Impact of White-Nose Syndrome on the Conservation Status of North American Bats. PLoS ONE 9(9): e107395. doi:10.1371/journal.pone.0107395

American Wind Wildlife Institute (AWWI). 2020. AWWI Technical Report: 2nd Edition: Summary of Bat Fatality Monitoring Data Contained in AWWIC. Washington, DC. Available at www.awwi.org. © 2020 American Wind Wildlife Institute.

Anthony, E.L.P., M.H. Stack, and T.H. Kunz. 1981. Night roosting and the nocturnal time budget of the little brown bat, *Myotis lucifugus*: Effects of reproductive status, prey density, and environmental conditions. Oecologia 51:151–156. https://doi.org/10.1007/BF00540593

Arnett, E. B., J. P. Hayes, and M. M. P. Huso. 2006. Patterns of preconstruction bat activity at a proposed wind facility in south-central Pennsylvania. An annual report submitted to the Bats and Wind Energy Cooperative. Bat Conservation International, Austin, Texas. 46 pp. Available at: https://batsandwind.org/wp-content/uploads/precon_pa.pdf

Arnett, E.B., W.K. Brown, W.P. Erickson, J.K. Fiedler, B.L Hamilton, T.H. Henry, A. Jain, G.D. Johnson, J. Kerns, R.R. Koford, C.P. Nicholson, T.J.O'Connell, M.D. Piorkowski, and R.D. Tankersley, Jr. 2008. Patterns of bat fatalities at wind energy facilities in North America. Journal of Wildlife Management 72:61–78.

Baerwald, E.F., G. H. D'Amours, B.J. Klug, and R.M.R. Barclay. 2008. Barotrauma is a significant cause of bat fatalities at wind turbines. Current Biology18(16): 695–696.

Baijnath-Rodino, J.A., C. Duguay, and E. LeDrew. 2018. Climatological trends of snowfall over the Laurentian Great Lakes Basin. International Journal of Climatology 38:3942–3962.

Barclay, R.M.R., J. Ulmer, C.J.A. MacKenzie, M.S. Thompson, L. Olson, J. McCool, E. Cropley, and G. Poll. 2004. Variation in the reproductive rate of bats. Canadian Journal of Zoology 82:688–693.

Belwood, J.J. 2002. Endangered bats in suburbia: observations and concerns for the future. In *The Indiana bat: biology and management of an endangered species* (A. Kurta and J. Kennedy, editors). Bat Conservation International, Austin, Texas, 193–198.

Bennett, V.J. and A.M. Hale. 2018. Texturizing wind turbine towers to reduce bat mortality DE-EE007033. Texas Christian University. Final Technical Report. Available at: https://www.osti.gov/servlets/purl/1502997.

Bernard, R.F. and G.F. McCracken. 2017. Winter behavior of bats and the progression of white-nose syndrome in the southeastern United States. Ecology and Evolution 7:1487–1496. https://doi.org/10.1002/ece3.2772

Bernard, R.F., E.V. Willcox, K.L. Parise, J.T. Foster, and G.F. McCracken. 2017. White-nose syndrome fungus, *Pseudogymnoascus destructans*, on bats captured emerging from caves during winter in the southeastern United States. BMC Zoology 2(12):1–11.

Bird Studies Canada, Canadian Wind Energy Association, Environment and Climate Change Canada and Ontario Ministry of Natural Resources and Forestry. 2018. Wind Energy Bird and Bat Monitoring Database Summary of the Findings from Post-construction Monitoring Reports. 56 pp. Available at: https://www.bsceoc.org/resources/wind/2018_Database_Summary_Report.pdf

Boyles, J. G. and A. E. McKechnie. 2010. Energy Conservation in Hibernating Endotherms: Why "Suboptimal" Temperatures Are Optimal. Ecological Modelling 221:1644–1647.

Boyles, J.G. 2016. Microclimate of Indiana bat hibernacula in southern Indiana. Unpublished technical report. Prepared by Eko Consulting LLC, Cobden, Illinois for U.S. Fish and Wildlife Service, Indiana Ecological Services Field Office, Bloomington, Indiana. 41 pp.

Brack, V. Jr., C.W. Stihler, R.J. Reynolds, C.M. Butchkoski, and C.S. Hobson. 2002. Effect of climate and elevation on distribution and abundance in the mideastern United States. Pp. 21–28 in *The Indiana bat: biology and management of an endangered species* (A. Kurta and J. Kennedy, editors). Bat Conservation International, Austin, Texas.

Broders, H.G., L.E. Burns, and S.C. McCarthy. 2013. First Records of the Northern Myotis (*Myotis Septentrionalis*) from Labrador and Summer Distribution Records and Biology of Little Brown Bats (*Myotis lucifugus*) in Southern Labrador. The Canadian Field Naturalist 127:266–269.

Brown, L. and C. Duguay. 2010. The response and role of ice cover in lake-climate interactions. Progress in Physical Geography 34:671–704. 10.1177/0309133310375653.

Burles, D.W., R.M., Brigham, R.A., Ring, and T.E. Reimchen. 2009. Influence of weather on two insectivorous bats in a temperate Pacific Northwest rainforest. Canadian Journal of Zoology 87:132–138.

Canada Energy Regulator (CER). 2019. Canada's Energy Future 2019: Energy Supply and Demand Projections to 2040. 56 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2019/2019nrgftr-eng.pdf

Canada Energy Regulator (CER). 2020. Canada's Energy Future 2020: Energy Supply and Demand Projections to 2050. 104 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2020/canada-energy-futures-2020.pdf

Canadian Wind Energy Association (CanWEA). 2021, entire. Available at: https://canwea.ca/wind-energy/national/

Carter, T.C. and G.A. Feldhamer. 2005. Roost tree use by maternity colonies of Indiana bats and northern long-eared bats in southern Illinois. Forest Ecology and Management 219:259–268.

Cryan, P. M. and R.M.R. Barclay. 2009. Causes of bat fatalities at wind turbines: Hypotheses and predictions. Journal of Mammalogy 90:1330–1340. https://doi.org/10.1644/09-MAMM-S-076R1.1

Cryan, P.M., C. Meteyer, J.G. Boyles, and D.S. Blehert. 2010. Wing pathology of white-nose syndrome in bats suggests life-threatening disruption of physiology. BioMed Central Biology 8(1):135–143.

Cryan, P.M., P.M. Goressen, C.D. Hein, M.R. Schirmacher, R.H. Diehl, M.M. Huso, D.T.S. Hayman, P.D. Fricker, F.J. Bonaccorso, D.H. Johnston, K. Heist, and D.C. Dalton. 2014. Behavior of bats at wind turbines. Proceedings of the National Academy of Sciences 111: 15,126–15,131. https://doi.org/10.1073/pnas.1406672111

Cryan, P.M., P. Marcos Gorresen, D. Dalton , S. Wolf , and F. Bonaccorso. 2016. Ultraviolet illumination as a means of reducing bat activity at wind turbines. Presented at the Wind Wildlife Research Meeting XI in Broomfield, Colorado. 2 December 2016. Available at https://www.nationalwind.org/wp-content/uploads/2017/04/83_Cryan.pdf

Davy, C.M., G.F. Mastromonaco, J.L. Riley, J.H. Baxter-Gilbert, H. Mayberry, C.K.R. Willis. 2017. Conservation implications of physiological carry-over effects in bats recovering from white-nose syndrome. Conservation Biology 31(3): 615–624. DOI: 10.1111/cobi.12841. Epub 2016 Dec 5. PMID: 27641049.

Dobony, C.A., A.C. Hicks, K.E. Langwig, R.I. von Linden, J.C. Okoniewski, and R.E. Rainbolt. 2011. Little brown myotis persist despite exposure to white-nose syndrome. Journal of Fish and Wildlife Management 2:190–195. http://dx.doi.org/10.3996/022011-JFWM-014

Dwyer, P.D. 1971. Temperature regulation and cave-dwelling in bats: An evolutionary perspective. Mammalia 35(3):424–455.

Easterling, D.R., K.E. Kunkel, J.R. Arnold, T. Knutson, A.N. LeGrande, L.R. Leung, R.S. Vose, D.E. Waliser, and M.F. Wehner, 2017: Precipitation change in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock (editors)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 207–230, doi: 10.7930/J0H993CC.

Erickson, W., G. Johnson, D. Young, D. Strickland, R. Good, M. Bourassa, K. Bay, and K.

Sernka. 2002. Synthesis and comparison of baseline avian and bat use, raptor nesting and mortality information from proposed and existing wind developments. WEST, Inc. 2003 Central Avenue, Cheyenne, Wyoming 82001.

Erickson, J.L. and S.D. West. 2002. The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. Acta Chiropterologica 4:17–24.

Erkert, H.G. 1982. Ecological aspects of bat activity rhythms. Pages 201–242 in Ecology of bats. T. H. Kunz (editor). Plenum Press, New York, New York, USA. 425 pp.

Field, K.A., B.J. Sewall, J.M. Prokkola, G.G. Turner, M. Gagnon, T.M. Lilley, J.P. White, J.S. Johnson, C.L. Hauer, and D.M. Reeder. 2018. Effect of torpor on host transcriptomic responses to a fungal pathogen in hibernating bats. Molecular Ecology 27: 3727–3743. https://doi.org/10.1111/mec.14827

Frick, W.F., D.S. Reynolds, and T.H. Kunz. 2010. Influence of climate and reproductive timing on demography of little brown myotis *Myotis lucifugus*. Journal of Animal Ecology 79:128–136.

Frick, W.F., S.J. Puechmaille, J.R. Hoyt, B.A. Nickel, K.E. Langwig, J.T. Foster, K.E. Barlow, T. Bartonička, D. Feller, A. Haarsma, C. Herzog, I. Horaček, J. Kooij, B. Petrov, R. Reynolds, L. Rodrigues, C.W. Stihler, G.G. Turner, and A.M. Kilpatrick. 2015. Disease alters macroecological patterns of North American bats. Global Ecology and Biogeography 24:741–749.

Fuller, N.W., J.D. Reichard, M.L. Nabhan, S.L. Fellows, L.C. Pepin, and T.H. Kunz. 2012. Free-ranging little brown myotis (*Myotis lucifugus*) heal from wing damage associated with white-nose syndrome. EcoHealth 8:154–162.

Gargas, A., M.T. Trest, M. Christensen, T.J. Volk, and D.S. Blehert. 2009. *Geomyces destructans* sp. nov. associated with bat white-nose syndrome. Mycotaxon 108:147–154.

Grodsky, S.M., M.J. Behr, A. Gendler, D. Drake, B.D. Dieterle, R.J. Rudd, and N.L. Walrath. 2011. Investigating the causes of death for wind turbine-associated bat fatalities. Journal of Mammalogy 92:917–925.

Haase, C.G., N.W. Fuller, Y.A. Dzal, C.R. Hranac, D.T.S. Hayman, C.L. Lausen, K.A. Silas, S.H. Olson, and R.K. Plowright. 2020. Body mass and hibernation microclimate may predict bat susceptibility to white-nose syndrome. Ecology and Evolution 11:506–515.

Hallam, T.G. and P. Federico. 2012. The Panzootic White-nose Syndrome: An Environmentally Constrained Disease? Transboundary and Emerging Diseases 59:269–278.

Hayes, M.A., L.A. Hooton, K.L. Gilland, C. Grandgent, R.L. Smith, S.R. Lindsay, J.D. Collins, Susan M. Schumacher, P.A. Rabie, J. C. Gruver, and J. Goodrich-Mahoney. 2019. A smart curtailment approach for reducing bat fatalities and curtailment time at wind energy facilities. Ecological Applications: e01881.

Horn, J. W., E. B. Arnett, M. Jensen, and T. H. Kunz. 2008a. Testing the effectiveness of an experimental acoustic bat deterrent at the Maple Ridge wind farm. The Bats and Wind Energy Cooperative and Bat Conservation International, Austin, TX. 30 pp.

Horn, J. W., E. B. Arnett, and T. H. Kunz. 2008b. Behavioral responses of bats to operating wind turbines. Journal of Wildlife Management 72:123–132.

Humphrey, S.R. 1975. Nursery roosts and community diversity of nearctic bats. Journal of Mammalogy 56(2):321–346.

Humphrey, S., A. Richter, and J. Cope. 1977. Summer habitat and ecology of the endangered Indiana Bat, *Myotis sodalis*. Journal of Mammalogy 58(3):334–346. doi:10.2307/1379332

Humphries, M., D. Thomas and J. Speakman. 2002. Climate-mediated energetic constraints on the distribution of hibernating mammals. Nature 418:313–316. https://doi.org/10.1038/nature00828

Humphries, M.M., J. Umbanhowar, and K.S. McCann. 2004. Bioenergetic Prediction of Climate Change Impacts on Northern Mammals. Integrative and Comparative Biology 44(2):152–162. https://doi.org/10.1093/icb/44.2.152

Huso, M., T. Conkling, D. Dalthorp, M. Davis, H. Smith, A. Fesnock, and T. Katzner. 2021. Relative energy production determines effect of repowering on wildlife mortality at wind energy facilities. Journal of Applied Ecology DOI: 10.1111/1365-2664.13853

Ineson, K.G. 2020. Demography of a recovery. Doctor of Philosophy. University of New Hampshire. Durham, New Hampshire.

Johnson, J.S., M.R. Scafini, B.J. Sewall, and G.G. Turner. 2016. Hibernating bat species in Pennsylvania use colder winter habitats following the arrival of white-nose syndrome. Pp. 181–199 in (C.M. Butchkoski, D.M. Reeder, G.G. Turner, and H.P. Whidden, editors) Conservation and Ecology of Pennsylvania's Bats. The Pennsylvania Academy of Sciences.

Johnson, J.S., A.S. Blomberg, J.G. Boyles, and T.M. Lilley. 2021. The Winter Worries of Bats: Past and Present Perspectives on Winter Habitat and Management of Cave Hibernating Bats. Pp. 209–221 in (B.K. Lim, M.B. Fenton, R.M. Brigham, S. Mistry, A. Kurta, E.H. Gillam, A. Russel, and J. Ortega (editors) 50 Years of Bat Research, Foundations and New Frontiers. North American Society for Bat Research.

Keller, S., J.M. Lorch, B. Berlowski-Zier, A. Ballman, and D.S. Blehert. 2021. Analysis of

Archival Specimens Confirms White-nose Syndrome in Little Brown Bats (*Myotis lucifugus*) from New York, USA, in Spring 2007. Journal of Wildlife Diseases 57:1–4.

Kilpatrick, A.M., J.R. Hoyt, R.A. King, H.M. Kaarakka, J.A. Redell, J.P. White, and K.E. Langwig. 2020. Impact of censusing and research on wildlife populations. Conservation Science and Practice 2:e264. https://doi.org/10.1111/csp2.264

Kunz, T.H., J.A. Wrazen, and C.D. Burnett. 1998. Changes in body mass and fat reserves in pre-hibernating little brown bats (*Myotis lucifugus*). Écoscience 5(1):8–17. DOI: 10.1080/11956860.1998.11682443.

Kunz, T.H., E.B. Arnett, W.P. Erickson, A.R. Hoar, G.D. Johnson, R.P. Larkin, M.D. Strickland, R.W. Thresher, and M.D. Tuttle. 2007. Ecological impacts of wind energy development on bats: questions, research needs, and hypotheses. Frontiers in Ecology and Environment 5(6):315–324.

Kurta, A. 2005. Roosting ecology and behavior of Indiana bats (*Myotis sodalis*) in summer. Pp. 29–42 in K.C. Vories and A. Harrington (editors), Proceedings of the Indiana bat and coal mining: a technical interactive forum. Office of Surface Mining, U.S. Department of the Interior, Alton, Illinois.

Kurta, A., K. J. Williams, and R. Mies. 1996. Ecological, behavioural, and thermal observations of a peripheral population of Indiana bats (*Myotis sodalis*). Pp. 102–117 in Bats and forests (R. M. R. Barclay and R. M. Brigham, editors). Research Branch, Ministry of Forests, Province of British Columbia, Victoria, British Columbia, Canada.

Kurta, A. and S.M. Smith. 2014. Hibernating bats and abandoned mines in the Upper Peninsula of Michigan. Northeastern Naturalist 21(4):587–605.

Kurta, A. and S.M. Smith. 2020. Changes in population size and clustering behavior of hibernating bats in the Upper Peninsula of Michigan after arrival of white-nose syndrome. Northeastern Naturalist 27:763–772.

Kurta, A., R.W. Foster, B.A. Daly, A.K. Wilson, R.M. Slider, C.D. Rockey, J.M. Rockey, B.L. Long, G.G. Auteri, J.D. Collins, J.P. White, H.M. Kaarakka, J.A. Redell, and D.M. Reeder. 2020. Exceptional Longevity in Little Brown Bats Still Occurs, despite Presence of White-Nose Syndrome. Journal of Fish and Wildlife Management 11:583–587.

Lacki, M.J., S.K. Amelon, and M.D. Baker. 2007. Foraging ecology of bats in forests. Pp. 83–127 in Lacki, M.J., J.P. Hayes, and A. Kurta (editors). 2007. Bats in forests: conservation and management. John Hopkins University Press, Baltimore, Maryland, USA.

Langwig, K.E., W.E. Frick, R. Reynolds, K.L. Parise, K.P. Drees, J.R. Hoyt, T.L. Cheng, T.H. Kunz, J.T. Foster, and A.M. Kilpatrick. 2015b. Host and pathogen ecology drive the seasonal dynamics of a fungal disease, white-nose syndrome. Proceedings of the Royal Society B 282:1–7.

Lawson M., D. Jenne, R. Thresher, D. Houck, J. Wimsatt, and B. Straw. 2020. An investigation into the potential for wind turbines to cause barotrauma in bats. PLoS ONE 15(12): e0242485. https://doi.org/10.1371/journal.pone.0242485

Leivers, S.J., M.B. Meierhofer, B.L. Pierce, J.W. Evans, and M.L. Morrison. 2019. External temperature and distance from nearest entrance influence microclimates of cave and culvert-roosting tri-colored bats (*Perimyotis subflavus*). Ecology and Evolution 9(24):14042–14052. https://doi.org/10.1002/ece3.5841

Linton, D.M. and D.W. MacDonald. 2018. Spring weather conditions influence breeding phenology and reproductive success in sympatric bat populations. Journal of Animal Ecology 87:1080–1090.

Lorch, J.M., D.L. Lindner, A. Gargas, L.K. Muller, A.M. Minnis, and D.S. Blehert. 2013. A culture-based survey of fungi in soil from bat hibernacula in the eastern United States and its implications for detection of Geomyces destructans, the causal agent of bat white-nose syndrome. Mycologia 105:237–252.

Lorch, J.M., J.M. Palmer, D.L. Linder, A.E. Ballman, K.G. George, K. Griffin, S. Knowles, J.R. Huckabee, K.H. Haman, C.D. Anderson, P.A. Becker, J.B. Buchanan, J.T. Foster, and D.S. Blehert. 2016. First detection of white-nose syndrome in western North America. mSphere 1(4). DOI: 10.1128/mSphere.00148-16

Lucan, R.K., M. Weiser, and V. Hanak. 2013. Contrasting effects of climate change on the timing of reproduction and reproductive success of a temperate insectivorous bat. Journal of Zoology 290:151–159.

Lundy, M., I. Montgomery, and J. Russ. 2010. Climate change-linked range expansion of Nathusius' pipistrelle bat, *Pipistrellus nathusii* (Keyserling and Blasius, 1839). Journal of Biogeography 2010(37):2232–2242

Maher, S.P., A.M. Kramer, J.T. Pulliam, M.A. Zokan, S.E. Bowden, H.D. Barton, K. Magori, and J.M. Drake. 2012. Spread of white-nose syndrome on a network regulated by geography and climate. Nature Communications 3:1306.

Martínková, N., J. Pikula, J. Zukal, V. Kovacova, H. Bandouchova, T. Bartonička, A.D. Botvinkin, J. Brichta, H. Dundarova, T. Kokurewicz, N.R. Irwin, P. Linhart, O.L. Orlov, V. Piacek, P. Škrabánek, M.P. Tiunov and A. Zahradníková jr. 2018. Hibernation temperature-dependent *Pseudogymnoascus destructans* infection intensity in Palearctic bats. Virulence 9: 1734-1750. DOI: 10.1080/21505594.2018.1548685.

McAlpine, D.F., J.L. Bullied, and P.D. Seymour. 2021. A Maternity Roost of Silver-Haired Bats (*Lasionycteris noctivagans*) in New Brunswick: First Evidence of Parturition in Atlantic Canada. Northeastern Naturalist 28(1):1–6.

McNab, B. 1982. Evolutionary Alternatives in Physiological Ecology of Bats. Pp. 151–200 in (Kunz, T.H., editor) Ecology of Bats. Plenum Press, New York.

Meierhofer, M.B., J.S. Johnson, K.A. Field, S.S. Lumadue, A. Kurta, J.A. Kath, and D.M. Reeder. 2018. Bats recovering from white-nose syndrome elevate metabolic rate during wing-healing in spring. Journal of Wildlife Diseases 54(3):480–490. https://doi.org/10.7589/2017-08-195

Minnis, A.M. and D.L. Lindner. 2013. Phylogenetic evaluation of *Geomyces* and allies reveals no close relatives of *Pseudogymnoascus destructans*, comb. nov., in bat hibernacula of eastern North America. Fungal biology 117: 638–649.

Moore, M.S., J.D. Reichard, T.D. Murtha, B. Zahedi, R.M. Fallier, and T.H. Kunz. 2011. Specific Alterations in Complement Protein Activity of Little Brown Myotis (*Myotis lucifugus*) Hibernating in White-Nose Syndrome Affected Sites. PLoS ONE (11):e27430.

Muller, L.K., J.M. Lorch, D.L. Lindner, M. O'Connor, A. Gargas, and D.S. Blehert. 2013. Bat white-nose syndrome: a real-time TaqMan polymerase chain reaction test targeting the intergenic spacer region of *Geomyces destructans*, Mycologia 105:(2):253–259. DOI: 10.3852/12-242

Murray, S.W. and A. Kurta. 2004. Nocturnal activity of the endangered Indiana bat (*Myotis sodalis*). Journal of Zoology 262:197–206. http://search.ebscohost.com/login.aspx?direct=true&db=aph&AN=12207821&site=ehost-live

Muthersbaugh, M.S., W.M. Ford, A. Silvis, and K.E. Powers. 2019. Activity Patterns of Cave-Dwelling Bat Species during Pre-Hibernation Swarming and Post-Hibernation Emergence in the Central Appalachians. Diversity 11(9):159. https://doi.org/10.3390/d11090159

National Research Council. 2007. Environmental Impacts of Wind-Energy Projects. Washington, DC: The National Academies Press. https://doi.org/10.17226/11935.

Osborn, S. California Department of Fish and Game (personal communication April 30, 2021).

Oswalt, S.N., W.B Smith, P.D. Miles, and S.A. Pugh. 2019. Forest Resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 223 pp. https://doi.org/10.2737/WO-GTR-97

Peste, F., A. Paula, L.P. da Silva, J. Bernardino, P. Pereira, M. Mascarenhas, H. Costa, J. Vieira, C. Bastos, C. Fonseca, and M.J.R. Pereira. 2015. How to mitigate impacts of wind farms on bats? A review of potential conservation measures in the European context. Environmental Impact Assessment Review 51:10–22.

Perry. R.W. 2013. A review of factors affecting cave climates for hibernating bats in temperate North America. Environmental Reviews 21:28–39. dx.doi.org/10.1139/er-2012-0042

Plumb, G. and P. Budde 2011. Unpublished data from U.S. Fish and Wildlife Service Region 3 data request regarding status of 7 cave bat species (sent 03/30/2011).

Rabie, P.A., D. Riser-Espinoza, J. Studyvin, D. Dalthorp, and M. Huso. 2021. Performance of the GenEst mortality estimator compared to the Huso and Shoenfeld Estimators. American Wind Wildlife Institute Technical Report. 10 March 2021. 28 pp. Available at https://awwi.org/wp-content/uploads/2021/04/Performance-of-the-GenEst-Statistical-Mortality-Estimator.pdf

Racey, P.A. and S.M. Swift. 1981. Variations in gestation length in a colony of pipistrelle bats (*Pipistrellus pipstrellus*) from year to year. Journal of Reproductive Fertility 61:123–129.

Rebelo, H., P. Tarroso, and G. Jones. 2010. Predicted impact of climate change on European bats in relation to their biogeographic patterns. Global Change Biology 16:561–576. doi:10.1111/j.1365-2486.2009.02021.x

Reynolds, H.T., T. Ingersoll, and H.A. Barton. 2015. Modeling the environmental growth of *Pseudogymnoascus destructans* and its impact on the white-nose syndrome epidemic. Journal of Wildlife Diseases 51: 318–331.

Rodenhouse, N.L., L.M. Christenson, D. Parry, and L.E. Green. 2009. Climate change effects on native fauna of Northeastern forests. Canadian Journal of Forest Research 39:249–263.

Rollins, K.E., D.K. Meyerholz, G.D. Johnson, A.P. Capparella and S.S. Loew. 2012. A Forensic Investigation into the Etiology of Bat Mortality at a Wind Farm: Barotrauma or Traumatic Injury? Veterinary Pathology 49:362–371.

Romano, W.B., J.R. Skalski, R.L. Townsend, K.W. Kinzie, K.D. Coppinger, and M.F. Miller. 2019. Evaluation of an acoustic deterrent to reduce bat mortalities at an Illinois wind farm. Wildlife Society Bulletin 43(4):608–618.

Scheel, D, T.L.S. Vincent, and G.N. Cameron. 1996. Global Warming and the Species Richness of Bats in Texas. Conservation Biology 10(2)452–464.

Schirmacher, M.R. 2020. Evaluating the Effectiveness of Ultrasonic Acoustic Deterrents in Reducing Bat Fatalities at Wind Energy Facilities. U.S. Department of Energy, DE-EE0007036.

Sheffield S.R., J.H. Shaw, G.A. Heidt, and L.R. McClenaghan. 1992. Guidelines for the protection of bat roosts. Journal of Mammalogy 73: 707–710.

Silvis A., W.M. Ford, and E.R. Britzke. 2015. Effects of Hierarchical Roost Removal on

Northern Long-Eared Bat (*Myotis septentrionalis*) Maternity Colonies. PLoS ONE 10(1):e0116356. doi:10.1371/journal.pone.0116356

Sirajuddin, P. 2018. Vulnerability of tri-colored bats (*Perimyotis subflavus*) to white-nose syndrome in the Southeastern United States. Master of Science. Clemson University.

Solick, D., D. Pham, K. Nasman, and K. Bay. 2020. Bat activity rates do not predict bat fatality rates at wind energy facilities. Acta Chiroptera 22(1):135–146.

Straw, B. U.S. Geological Survey (personal communication, April 6, 2021).

Suriano, Z.J., D.A. Robinson, and D.J. Leathers. 2019. Changing snow depth in the Great Lakes basin (USA): Implications and trends. Anthropocene 26:1–11. https://doi.org/10.1016/j.ancene.2019.100208

Taucher, J., T. Librandi Mumma, and W. Capouillez. 2012. Pennsylvania Game Commission Wind energy Voluntary Cooperation Agreement Third Summary Report. 79 pp.

Thapa, V., Turner, G.G., and M.J. Roossinck. 2021. Phylogeographic analysis of *Pseudogymnoascus destructans* partitivirus-pa explains the spread dynamics of white-nose syndrome in North America. PLoS Pathogens 17(3):e1009236.

Thomas, C., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. rasmus, M. Ferreira de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A. Townsend Peterson13, O.L. Phillips, and S.E. Williams. 2004. Extinction risk from climate change. Nature 427:145–148. https://doi.org/10.1038/nature02121

Tuttle, M.D. and D.E. Stevenson. 1977. Variation in the cave environment and its biological implications. Pp. 108–121, in National Cave Management Symposium Proceedings. Albuquerque: Adobe Press, 1977.

U.S. Department of Energy (USDOE). 2015. Wind Vision: A New Era for Wind Power in the United States. 289 pp. Available at: https://www.energy.gov/sites/prod/files/WindVision_Report_final.pdf

U.S. Energy Information Administration (USEIA). 2020. Annual Energy Outlook 2020 with projections to 2050. 161 pp. Available at: https://www.eia.gov/outlooks/aeo/pdf/AEO2020%20Full%20Report.pdf

U.S. Fish and Wildlife Service (USFWS). 2011. A national plan for assisting states, Federal agencies, and tribes in managing white-nose syndrome in bats. U.S. Fish and Wildlife Service. Hadley, MA. 21 pp.

USFWS. 2016. Draft Midwest Wind Energy Multi-species

Habitat Conservation Plan in collaboration with the States of Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, and Wisconsin and the American Wind Energy Association. Public Review Draft. April 2016. Available at: https://www.fws.gov/midwest/endangered/permits/hcp/r3wind/DraftHCPandEIS.html

U.S. Forest Service (USFS). 2013. Adaptive Management Strategy for White-nose Syndrome, Available at: http://www.fs.usda.gov/detail/r2/home/?cid=stelprdb5319926. Accessed online October 21, 2020.

USFS. 2014. U.S. Forest Resource Facts and Historical Trends. Forest Service, FS-1035, 64 pp.

Verant, M.L., J.G. Boyles, W. Waldrep Jr., G. Wibbelt, and D.S. Blehert. 2012. Temperature Dependent Growth of Geomyces destructans, the Fungus That Causes Bat White-Nose Syndrome. PLoS ONE 7(9):e46280.

Warnecke, L., J.M. Turner, T.K. Bollinger, J.M. Lorch, V. Misra, P.M. Cryan, G. Wibbelt, D.S. Blehert, and C.K.R. Willis. 2012. Inoculation of bats with European *Geomyces destructans* supports the novel pathogen hypothesis for the origin of white-nose syndrome. Proceedings of the National Academy of Sciences, 109:6999-7003.

Weaver, S.P., C.D. Hein, T.R. Simpson, J.W. Evans, and I. Castro-Arellano. 2020. Ultrasonic deterrents significantly reduce bat fatality at wind turbines. Global Ecology and Conservation 24:e01099.

Webb, P.I., J.R. Speakman, and P.A. Racey. 1996. How hot is a hibernaculum? A review of the temperatures at which bats hibernate. Canadian Journal of Zoology 74:761–765.

Wilcox, A., L. Warnecke, J.M. Turner, L.P. McGuire, J.W. Jameson, V. Misra, T.C. Bollinger, and C.K.R. Willis. 2014. Behaviour of hibernating little brown bats experimentally inoculated with the pathogen that causes white-nose syndrome. Animal Behaviour 88:157–164.

Wilder, A.P., W.F. Frick, K.L. Langwig, and T.H. Kunz. 2011. Risk factors associated with mortality from white-nose syndrome among hibernating bat colonies. Biology Letters 7: 950–953. http://doi.org/10.1098/rsbl.2011.0355

Species Status Assessment Report for the Pearl River Map Turtle (*Graptemys pearlensis*)

Version 1.1

April 2021

U.S. Fish and Wildlife Service
Mississippi Basin Region
Atlanta, GA

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EXECUTIVE SUMMARY

The Pearl River map turtle (*Graptemys pearlensis*) is a freshwater turtle that inhabits rivers and large creeks with sand and gravel bottoms in the Pearl River drainage in Mississippi and Louisiana. The species was separated from the Pascagoula map turtle (*Graptemys gibbonsi*) in 2010, shortly after *G. gibbonsi* was petitioned for federal listing. This SSA is intended to provide the biological support for the decision on whether to propose to list the species as threatened or endangered. For the Pearl River map turtle to survive and reproduce, individuals need suitable habitat that supports essential life functions at all life stages. Several elements appear to be essential to the survival and reproduction of individuals: mainstem and tributary reaches within the Pearl River system that have sandbars, adequate flow, adequate supply of invertebrate prey items including insects and mollusks, and an abundance of emergent and floating basking structures of various sizes. Threats to the species include construction of reservoirs and dams, sand mining, excessive levels of development and agriculture, collection for the pet trade, invasive species, and long-term climate impacts such as prolonged flooding, drought, and sea level rise. Positive influences on the species include state laws and regulations that protect individual turtles and their habitat, and implementation of forestry best management practices, such as maintenance of riparian forested cover.

We delineated five resilience units of Pearl River map turtles based on HUC8 watersheds and in accordance with guidance from species experts (Upper Pearl, Middle Pearl-Silver, Middle Pearl-Strong, Bogue Chitto, and Lower Pearl). Historically, the majority of the range of the species was likely connected in a single interbreeding biological population, but we used the five analysis units in the SSA to most accurately describe trends in resiliency, forecast future resiliency, and capture differences in stressors among units. A recent genetics study for the species showed no distinct genetic structure and no evidence of isolation by distance, thus we consider the entire range of the Pearl River map turtle to be a single representative unit. We assessed current resilience using the following population and habitat factors: occupied tributaries, density, water quality, water engineering projects, forested riparian cover, and protected areas. The conditions of each of these factors were combined to classify the resilience of each population as high, moderate, or low. There are currently three Pearl River map turtle populations with moderate resilience, and two populations with low resilience (Figure EX-1).

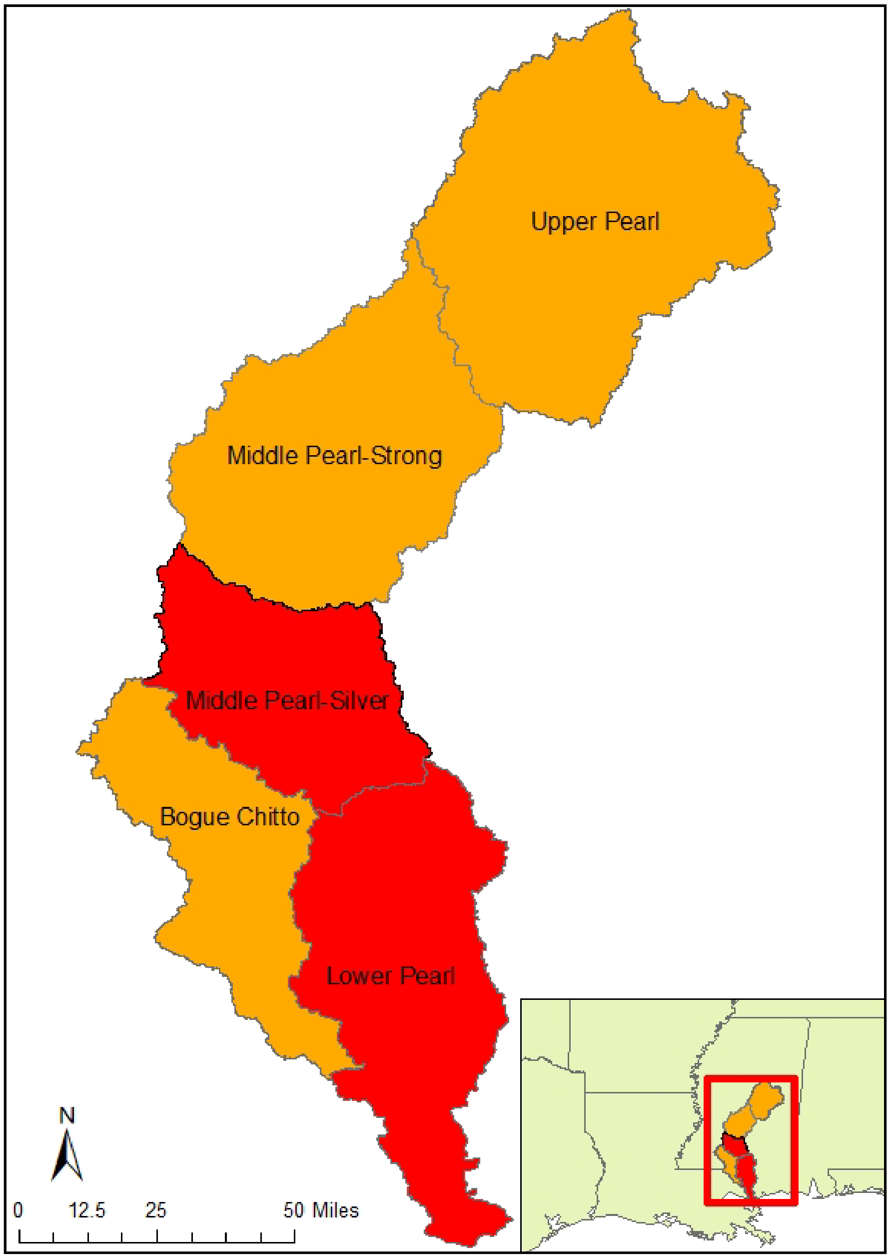

Figure EX-1-Resilience of the five units of Pearl River map turtles: low (red), moderate (orange), high (green).

To assess the future condition, we projected the primary current threats of land use (agriculture and development), potential future water engineering projects, and sea level rise into the future under six plausible scenarios, out to two different time steps: 2040 (20 years) and 2070 (50 years). The six scenarios capture the range of uncertainty in the changing human population footprint on the landscape, current emission models, implementation of water engineering projects, and how the Pearl River map turtle will respond to these changing conditions. Based on these factors, there are two populations predicted to substantially decrease in resilience across most of the future scenarios: Lower Pearl and Middle Pearl Strong. The Lower Pearl unit faces a myriad of future threats, including impacts from sea level rise in the southern portion of the unit, substantial increases in development and agriculture, and loss of forested cover. For the Middle Pearl Strong, the magnitude of decrease is most closely tied to the One Lake project. If the One Lake Project moves forward within the next 50 years, areas in and around the project site are anticipated to be substantially negatively impacted. The Bogue Chitto, Middle Pearl Silver, and Upper Pearl units are anticipated to maintain their current resilience, or only slightly decrease, as the main threats assessed are not anticipated to increase markedly in the future. One caveat for the Bogue Chitto unit is that there is uncertainty in levels and patterns of mining in the future, and this has been identified as a current threat in some portions of the unit.

Table EX-1-Anticipated magnitude and direction of change in resilience for Pearl River map turtles based on land use, climate change, and future water project scenarios.

Unit	Future Resilience
Bogue Chitto	Likely to maintain moderate resilience across all scenarios
Lower Pearl	Low resilience is expected to further decrease across all scenarios
Middle Pearl Silver	Likely to maintain low resilience across all scenarios
Middle Pearl Strong	Moderate resilience likely to decrease substantially
Upper Pearl	Likely to maintain moderate resilience across all scenarios

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 – INTRODUCTION AND ANALYTICAL FRAMEWORK

The Pearl River map turtle (*Graptemys pearlensis*) inhabits rivers and large creeks with sand and gravel bottoms in the Pearl River drainage in Mississippi and Louisiana (Lindeman 2013, p. 298). The species was separated from the Pascagoula map turtle (*Graptemys gibbonsi*) in 2010 (Ennen et al. 2010, entire), shortly after *G. gibbonsi* was petitioned for federal listing under the Endangered Species Act of 1973, as amended (Act), as a part of the 2010 Petition to List 404 Aquatic, Riparian and Wetland Species from the Southeastern United States by the Center for Biological Diversity (CBD 2010, p. 559-562). The Species Status Assessment (SSA) framework (USFWS 2016, entire) summarizes the information compiled and reviewed by the U.S. Fish and Wildlife Service (Service), incorporating the best available scientific and commercial data, in order to conduct an in-depth review of the species' biology and threats, evaluate its biological status, and assess the resources and conditions needed to maintain long-term viability. The intent is for the SSA to be easily updated as new information becomes available and to support all functions of the Endangered Species Program from Listing to Consultations to Recovery.

The Pearl River map turtle SSA is intended to provide the biological support for the decision on whether to propose to list the species as threatened or endangered and, if so, to determine whether it is prudent to designate critical habitat in certain areas. Importantly, the SSA Report is not a decisional document by the Service but provides a review of available information strictly related to the biological status of the Pearl River map turtle. The listing decision will be made by the Service after reviewing this document and all relevant laws, regulations, and policies, and the results of a proposed decision will be announced in the Federal Register, with appropriate opportunities for public input.

For the purpose of this assessment, we generally define viability as the ability of the species to sustain populations in its natural systems over time. Using the SSA framework (Figure 1.1), we consider what the species needs to maintain viability by characterizing the status of the species in terms of its resiliency, redundancy, and representation (Wolf et al. 2015, entire).

To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308-311; Wolf et al. 2015, entire). To sustain populations over time, a species must have the capacity to withstand:

(1) environmental and demographic stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is better able to adapt to novel changes and to tolerate environmental stochasticity and catastrophes. In general, species viability will increase with increases in resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308-311; Wolf et al. 2015, entire; Smith et al. 2018, p. 306).

- **Resiliency** is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature and rainfall), periodic disturbances within the normal range of variation (fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

  We can best gauge resiliency by evaluating population level characteristics such as: demography (abundance and the components of population growth rate -- survival, reproduction, and migration), genetic health (effective population size and heterozygosity), connectivity (gene flow and population rescue), and habitat quantity, quality, configuration, and heterogeneity. Also, for species prone to spatial synchrony (regionally correlated fluctuations among populations), distance between populations and degree of spatial heterogeneity (diversity of habitat types or microclimates) are also important considerations.

- **Redundancy** is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely (Mangal and Tier 1993, p. 1083).

We can best gauge redundancy by analyzing the number and distribution of populations relative to the scale of anticipated species-relevant catastrophic events. The analysis entails assessing the cumulative risk of catastrophes occurring over time. Redundancy can be analyzed at a population or regional scale, or for narrow-ranged species, at the species level.

- **Representation** is the ability of a species to adapt to both near-term and long-term changes in its physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (pathogens, competitors, predators, etc.) environments. This ability to adapt to new environments-- referred to as adaptive capacity--is essential for viability, as species need to continually adapt to their continuously changing environments (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either [1] moving to new, suitable environments or [2] by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270). The latter (evolution) occurs via the evolutionary processes of natural selection, gene flow, mutations, and genetic drift (Crandall et al. 2000, p. 290-291; Sgro et al. 2011, p. 327; Zackay 2007, p. 1).

  We can best gauge representation by examining the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas. In assessing the breadth of variation, it is important to consider both larger-scale variation (such as morphological, behavioral, or life history differences which might exist across the range and environmental or ecological variation across the range), and smaller-scale variation (which might include measures of interpopulation genetic diversity). In assessing the dispersal ability, it is important to evaluate the ability and likelihood of the species to track suitable habitat and climate over time. Lastly, to evaluate the evolutionary processes that contribute to and maintain adaptive capacity, it is important to assess [1] natural levels and patterns of gene flow, [2] degree of ecological diversity occupied, and [3] effective population size. In our species status assessments, we assess all three facets to the best of our ability based on available data.

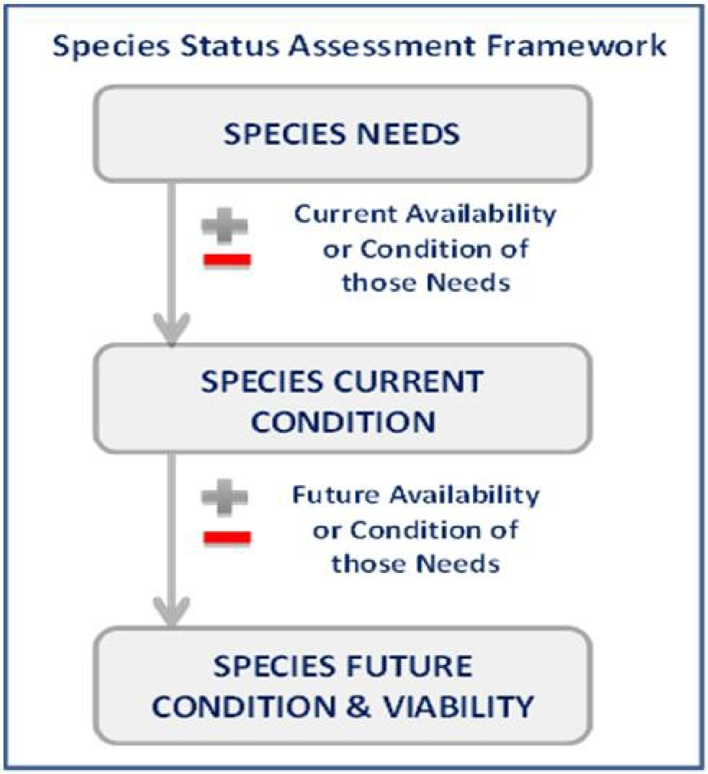

Figure 1.1. Species Status Assessment Framework

To evaluate the biological status of the Pearl River map turtle, both currently and into the future, we assessed a range of conditions to allow us to consider the species' resiliency, redundancy, and representation (together, the 3Rs). This SSA provides an assessment of biology and natural history, and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the species.

The format for this SSA includes: (1) Species Biology (2) Species Needs (3) Influences on Viability (4) Current Conditions and (5) Future Conditions. This document is a compilation of the best available scientific and commercial information, and a description of past, present, and likely future risk factors to the Pearl River map turtle.

CHAPTER 2 – SPECIES BIOLOGY AND INDIVIDUAL NEEDS

In this chapter, we provide biological information about the Pearl River map turtle, including its taxonomic history, morphological description, historical and current distribution and range, and known life history. We then outline the resource needs of individuals.

2.1 Taxonomy

Kingdom: Animalia
Division: Chordata
Class: Reptilia
Order: Testudines
Family: Emydidae
Genus: *Graptemys*
Species: *Graptemys pearlensis*
Common name: Pearl River map turtle (= Pearl map turtle)

Before 1992, all megacephalic (big-headed) map turtles from the Pearl, Pascagoula, Mobile Bay, and Escambia-Conecuh river systems were recognized as the Alabama map turtle (*Graptemys pulchra*) (Baur 1893, p. 675-676; Lovich and McCoy 1992, p. 294). That changed when morphological features were analyzed among those river systems, resulting in the name *G. pulchra* being restricted to the Mobile Bay drainages, individuals from the Escambia-Conecuh River system being elevated to a new species *G. ernsti* (Escambia map turtle), and individuals from the Pascagoula and Pearl River systems being elevated to the new species *G. gibbonsi* (Pascagoula map turtle, Lovich and McCoy 1992, pp. 296-306). A molecular systematics study supported the division of *G. pulchra* into three species, although *G. gibbonsi* was only represented in the analysis by genetic material collected from individuals in the Pearl River drainage (Lamb et al. 1994, p. 554-559). The Pearl River map turtle was taxonomically separated from the Pascagoula map turtle (*G. gibbonsi*) in 2010 based on morphological and genetic differences (Ennen et al. 2010, p. 109-110). This separation was subsequently supported with a molecular analysis of the phylogeny of all map turtles (*Graptemys*; Thomson et al. 2018, p. 65). In contrast to these studies, another study found *G. gibbonsi* and *G. pearlensis* genetically undifferentiated, both in nuclear and in mitochondrial DNA, and that previous analyses may represent population level variation rather than taxonomic variation (Praschag et al. 2017, p. 680-681). For the purposes of this assessment, the Pearl River map turtle is recognized as a separate species from the Pascagoula map turtle, Escambia map turtle, and Alabama map turtle, and is considered a valid species because of morphological and genetic (mtDNA and nuclear genes) support and recognition by the herpetological community (Iverson et al. 2017, p. 86; Ennen et al. 2012, p. 889.881-889.884).

2.2 Species Description

Map turtles (genus *Graptemys*) are named for the intricate pattern on the carapace that often resembles a topographical map. In addition to the intricate pattern, the shape of the carapace (top half of shell) in map turtles is very distinctive. The carapace is keeled, and many species show some type of knobby projections or spikes down the vertebral scutes (located down the midline of the carapace). Marginal scutes (located along the edge of the carapace) are also serrated and this trait is most pronounced in the three species of closely-related microcephalic (small-headed) map turtles, including the black-knobbed map turtle (*G. nigrinoda)*, ringed map turtle (*G. oculifera*), and yellow-blotched map turtle (*G. flavimaculata*). Among the megacephalic clade of map turtles these traits are present but less pronounced in adults (Lovich and McCoy 1992, p. 293). The megacephalic clade of map turtles includes the Pascagoula map turtle, Barbour's map turtle (*G. barbouri*), the Escambia map turtle, the Pearl River map turtle, and the Alabama map turtle. This group has the largest adult female body size and greatest degree of sexual dimorphism in the genus *Graptemys*, with adult females being over two times the carapace length of adult males on average. Megacephalic map turtles are defined by the following combination of characteristics (Lovich and McCoy 1992, p. 294).

1. Large female size reaching 29.5 – 33.0 centimeters (cm) (11.6 – 13.0 inches (in)) in maximum carapace length;
2. Extreme sexual dimorphism in adults, SDI ([sexual dimorphism index], size of larger sex divided by size of smaller sex, Gibbons and Lovich 1990, p. 2-3) is 2.42 – 2.58;
3. Head is broad, particularly in adult females, with greatly expanded alveolar surfaces (flattened surfaces for crushing; turtles do not have teeth) of jaws;

4. Head pattern consists of an interorbital blotch (between the eyes on the dorsal surface of the head) and large postorbital blotches (posterior to the eye); and
5. Vertebral scutes with salient spines.

Figure 2.1. A mature male Pearl River map turtle (*Graptemys pearlensis*).

Distinguishing among the Pearl River map turtle (Figure 2.1), Pascagoula map turtle, Barbour's map turtle, Escambia map turtle, and Alabama map turtle is easiest via locality information, but there is some genetic, morphological, and pattern variation among the three species. In the Pascagoula map turtle and the Pearl River map turtle, there is a single, yellow, vertical bar near the center of the dorsal surface of each marginal scute, while the Alabama map turtle has a spot surrounded by concentric yellow semicircles near the center of the dorsal surface of each marginal scute (Lovich and McCoy 1992, p. 302-303; Ennen et al. 2010, p. 100). The yellow bars on the dorsal surface of the marginals tend to be wider in the Pascagoula map turtle than in the Pearl River map turtle, and are bordered by concentric dark rings, which are often absent or faded in the Pearl River map turtle (Ennen et al. 2010, p. 101-102). In many individuals, the yellow bars on the posterior-most marginal scutes in the Pascagoula map turtle and the Pearl

River map turtle do not connect the outer margin of the carapace with the vertebral scutes, and on average, Pascagoula map turtle specimens have longer vertical bars on the posterior-most marginal scutes than in Pearl River map turtles (Ennen et al. 2010, p. 101). The darkened borders along the seams between the ventral surfaces of each marginal scute are relatively narrow in the Pascagoula map turtle when compared to the Alabama map turtle and the Pearl River map turtle (Lovich and McCoy 1992, p. 302-303; Ennen et al. 2010, p. 101-102). On the head of all three species, large, light-colored postorbital blotches are connected via a large interorbital blotch and supraoccipital (on the dorsal surface of the head but posterior to the eye) spots are absent (Lovich and McCoy 1992, p. 302-303). In the Pascagoula map turtle and more commonly in the Pearl River map turtle, the anterior end of the interorbital blotch often forms a three-pronged nasal trident shape that is absent in the Alabama map turtle and Barbour's map turtle, but especially prominent in the Escambia map turtle (Lovich and McCoy 1992, p. 302-303; Ennen et al. 2010, p. 101-102). In a 2010 study, interorbital blotches in 66 percent of Pascagoula map turtle specimens and 79 percent of Pearl River map turtle specimens exhibited a nasal trident shape (Ennen et al. 2010, p. 102). Both Pascagoula map turtles and Pearl River map turtles have a black or dark brown vertebral stripe along the keel of the carapace. In Pearl River map turtles, the stripe is continuous in most individuals but in Pascagoula map turtles the stripe is usually broken up by the ground color of the carapace (Ennen et al. 2010, p. 103). Hatchlings of megacephalic map turtles have more vivid colors and higher contrast than adults, are more circular, have a more pronounced dorsal keel, and each marginal scute is serrated (Lovich et al. 2009, p. 029.3). The midline plastron lengths for the Pearl River map turtles range from 5.8 to 9.8 cm (2.3 to 3.9 in) in mature males and reach 21.5 cm (8.5 in) in mature females (Lindeman 2013, p. 299).

2.3 Range and Distribution

The Pearl River map turtle is endemic to the Pearl River drainage in Mississippi and Louisiana. Counties with known records for the species in the state of Mississippi include: Attala, Copiah, Hancock, Hinds, Lawrence, Leake, Lincoln, Madison, Marion, Neshoba, Pearl River, Pike, Rankin, Smith, Simpson, and Walthall. Two parishes in Louisiana, St. Tammany and Washington, have records for the species. When the Pearl River map turtle was described as its own taxon in 2010, its distribution in the Pearl River basin was not well known. Between 2015 and 2018, Lindeman (2019, entire) conducted surveys to document the extent of the range of the species. During this study, Lindeman documented an occupancy of 1279.6 river kilometers (rkm) (795.1 rm) in the drainage, with 647.0 km (402.0 rm) in tributaries and 632.6 km (393.1 rm) in the main channels of the Pearl and West Pearl rivers (Lindeman 2019, p. 19-47). Within this total distribution, presence in 188.3 rkm (117.0 rm) of the occupied range was unknown prior to these surveys (Lindeman 2019, p. 6-19). In an effort to determine the extent of the species distribution, Lindeman surveyed 16 connecting tributaries of the Pearl River and eight additional streams upstream of the known range but did not document presence in them (Lindeman 2019, p. 20). In his report, Lindeman advised not to overestimate the importance of the recent range extensions, pointing out that while the study increased the known range by 15 percent, the range-wide population estimates indicate that the newly reported stream reaches only account for 9 percent of the estimated global population of Pearl River map turtles because most of the extensions were in areas where visual surveys indicated either low or very low population densities (Lindeman 2019, p. 25).The occupied range of the Pearl River map turtle includes portions of the Pearl River, West Pearl River, Bogue Chitto, East Pearl River, Yockanookany River, Strong River, Holmes Bayou, Pearl Navigation Canal, Lobutcha Creek, Tuscolometa Creek, Pelahatchie Creek, Purvis Creek, Pushepatapa Creek, Topisaw Creek, Magees Creek, Hobolochitto Creek, and West Hobolochitto Creek (Figure 2.2). This species has also been reported in upper reaches of the Ross Barnett Reservoir (where conditions are more lotic and less lentic). It is likely that the species will be discovered in smaller creeks in the future but likely at low population densities (Selman pers. comm. 2020b, p. 16).

Figure 2.2. Known range of the Pearl River map turtle as of 2019 (Lindeman 2019, p. 50). Rivers and tributaries are drawn to the known upstream extent of the range of the species.

2.4 Diet

The diet of the Pearl River map turtle varies between females and males, and females grow proportionally larger heads and jaws as they age. A recent study on fecal samples of Pearl River map turtles found that mature females consume mostly Asian clams (*Corbicula fluminea*), while males and unsexed juveniles eat insects, with mature males specializing in caddisfly larvae and consuming more mollusks than juveniles (Vučenović and Lindeman 2020, entire). Moderately large native mussels had previously been reported in fecal samples from adult female Pearl River map turtles (R. Jones, pers. comm. reported in Lindeman 2013, p. 298-299), but an account from before the Pearl River map turtle was split from the Pascagoula map turtle also identified Asian clams (*Corbicula fluminea*) as a preferred prey item (Ernst, pers. obs. reported in Lovich 2009, p. 305-306). In stomach contents of specimens from the Pearl River, two adult males contained only insects, while a juvenile female contained only snails and clams (Cagle 1952, p. 228). In fecal samples from a site on the Pearl River, the diet for both sexes of all sizes combined was composed of 44 percent fish, 25 percent mollusks, and 25 percent insects, which is broader than the megacephalic species to the east, the Pascagoula map turtle; apparently Pearl River map turtles did not specialize as much on mollusks as the Pascagoula map turtle (McCoy and Vogt, unpubl. data reported in Lovich et al. 2009, p. 029.4). Several accounts published before the Pearl River map turtle and Pascagoula map turtle were split indicate that juveniles, small females, and mature males are predominantly insectivorous (Dundee and Rossman 1989, p. 187; Lovich et al. 2009, p. 029.4; J. Vučenović and P. Lindeman, unpubl. data reported in Lindeman 2019, p. 22-31). Another observation from before the species were taxonomically divided found stomach contents of immature females containing less than 15 percent mollusks by volume and found no mollusks in stomachs of males (Cagle, 1952 data reported in Lindeman 2000, p. 553-563). Snails are occasionally consumed by *G. pearlensis*, and made up ~1% by volume of female samples (adult and juvenile) in a recent study (Vučenović and P. Lindeman 2020, p. 19).

2.5 Behavior and Activity

Many behaviors of the Pearl River map turtle are poorly understood, but some have been documented from observations of the species; we describe behaviors of similar species in lieu of

Pearl River map turtles when available. At night, numerous males and juveniles have been observed clinging to submerged branches and tree limbs just below the water surface (Cagle 1952, p. 227), so it has been suggested that males and juveniles forage during the day and rest at night (Lovich 2009, p. 303). Female foraging behavior is unknown, but a study on the megacephalic species to the east (the Pascagoula map turtle) found wood fragments in fecal samples from males but absent from females, suggesting that males probably forage on submerged deadwood structure, while researchers inferred from the molluscivorous diet of mature females that females forage along sand or gravel directly at the river bottom (Selman and Lindeman 2015, p. 794-795).

Demographic and seasonal biases in basking behavior have not been studied for this species, but many existing publications that estimate abundance or relative abundance for map turtles, in general, rely on observation counts of basking individuals, so understanding basking behavior is important. A study on the sympatric (overlapping in distribution) microcephalic species, the ringed map turtle (*Graptemys oculifera*), found that Pearl River map turtles were more frequently seen basking later in the afternoon than ringed map turtles, and suggested that more Pearl River map turtles might have been detected if more surveys were conducted after 1500 hours (3 pm) (Dickerson and Reine 1996, p.8). Ecologically similar Pascagoula map turtles were found to also bask later in the afternoon (particularly males) (Selman and Lindeman 2015, pp. 789-793).

Available information on movements, home range, and dispersal is limited. However, an early account that combined data on the Pearl River map turtle and the Pascagoula map turtle states that males and juveniles often stay in the same area at night where they were found during the day, and that researchers returning to areas that had basking map turtles during the day resulted in frequent night captures by hand, while areas without basking individuals yielded no captures (Cagle 1952, p. 227).

2.6 Reproduction

Information on Pearl River map turtle reproductive biology is lacking; we have little information on species specific clutch frequency, age of maturity, and other reproductive parameters. Pearl River map turtles excavate nests and lay their eggs on sandbars and beaches along river banks during the late spring and early summer months. A gravid female containing calcified eggs was collected from the Pearl River on June 8, 1951 (Cagle 1952, p. 228). In unpublished data on Pearl River map turtles and Pascagoula map turtles from the Chickasawhay River combined, McCoy and Vogt found gravid females from May to August (reported in Lovich et al. 2009, p. 029.4). A study on ringed map turtles reported incidentally finding three fresh nests from Pearl River map turtles during the month of June, identified by hatching them in captivity (Ennen et al. 2016, p. 094.4-094.6).

Five eggs incubated in captivity hatched in 62.8 days on average, and three clutches incubated in natural nests averaged 62 days to hatching; time from deposition to nest emergence by hatchlings in the natural clutches ranged from 67-79 days and averaged 69.3 days. The five lab-incubated hatchlings averaged 3.66 cm (1.44 in) in carapace length (Jones, unpublished data, summarized in Ennen et al. 2016, p. 094.4-094.6). Hatchlings typically emerge from the nest within three hours after sunset. During trials releasing hatchlings by hand, hatchlings moved to the nearest shade if released during the bright sunlight of day, even if released near the water (Anderson 1958, p. 212-215). Juvenile growth rates have not been studied, but one juvenile female from the Pearl River was 2.27 cm (0.89 in) in carapace length upon hatching, 4.24 cm (1.67 in) after one growing season, and 5.93 cm (2.33 in) after the second growing season (Cagle 1952, p. 228). Another early study on the megacephalic clade reported hatchling mean plastron length as 3.02 cm (1.19 in), and the following mean lengths among 29 individuals for subsequent years, estimated by counting growth rings: 4.37 cm (1.72 in) in the first year, 5.11 cm (2.01 in) in the second year, 6.09 cm (2.40 in) in the third year, and 6.74 cm (2.65 in) in the fourth year (Cagle 1952, p. 227).

At a site on the Pearl River 2 km (1.2 mi) downstream of the Strong River, the smallest gravid female had a carapace length of 20.5 cm (8.1 in) but authors suggested that maturation age and size may vary. A total of 22 clutches at that site ranged from 3 to 9 eggs (mean = 6.4 eggs; Vogt et al. 2019, p. 557-558). An average clutch size of 6.4 eggs with a range of 4-9 eggs was reported for the Pearl River map turtle and stated that females probably produce multiple clutches per year (Ennen et al. 2016, pp. 094.4-094.6). During the dissection of an adult female (plastron length of 17.0 cm (6.7 in)) specimen from the Pearl River, three calcified eggs were found and there was evidence of an additional clutch developing; the three eggs were 4.27-4.73 cm (1.68-1.86 in) in length and 2.5-2.7 cm (0.98-1.06 in) in width (Cagle 1952, p. 228). In another study, the length, width, and mass among 33 eggs averaged 4.01 cm (1.58 in), 2.68 cm (1.06 in), and 16.4 grams (0.58 ounces), respectively (Ennen et al. 2016, pp. 094.4-094.6). Unpublished data combining Pearl River map turtles and Pascagoula map turtles found a 22.4 cm (8.8 in) mean carapace length among eight gravid females (reported in Lovich et al. 2009, p. 029.4).

2.7 Habitat

There are a lack of studies relating demographic parameters such as occupancy and abundance, however general habitat features have been defined. Pearl River map turtles occur in sand and gravel-bottomed rivers and creeks with dense accumulations of deadwood; they have not been documented in oxbow lakes or other floodplain habitats. They were notably absent from lakes where their sympatric microcephalic species, the ringed map turtle, is present, but do occur at the upstream reach of Ross Barnett Reservoir, an impoundment of the Pearl River (Lindeman 2013, p. 298). Accounts from before the Pearl River map turtle and Pascagoula map turtle were taxonomically divided described ideal habitat as rivers and creeks with sand or gravel bottoms, abundant basking structures, and swift currents (Lovich 2009, p. 304; USFWS 2006, p. 2). Although some species of *Graptemys* appear to handle some salinity increases, there is some evidence that the group is largely intolerant of brackish and saltwater environments (Selman and Qualls 2008 pp. 228-229; Lindeman 2013, pp. 396-397). We do not know the physiological effects of salinity on the Pearl River map turtle, however we do know that although there are some cases of microcephalic species entering brackish water, thus far there are no records of macrocephalic species exhibiting this behavior (Agha 2018, p. 7). Future research is needed to understand the impacts of sea level rise on the Pearl River map turtle (Agha et al. 2019, entire).

Emergent deadwood serves as thermoregulatory basking structure, foraging structure for males and juveniles (Selman and Lindeman 2015, pp. 794-795), and as an overnight resting place for males and juveniles (Cagle 1952, p. 227). Moderate-to-high basking densities of Pearl River map turtles were always associated with moderate-to-high deadwood densities, but some sites with ample deadwood structure did not have high densities of basking map turtles, indicating that those sites may lack other important characteristics (Lindeman 1999, pp. 37-40). Most deadwood structure observed on surveys was not occupied by basking turtles, and structures that were occupied typically had enough surface area for additional turtles (Lindeman 1999, pp. 37-40). The importance of deadwood and its source in riparian forests to the abundance of riverine turtle species is documented (Sterrett, et al. 2011, entire). Comparisons have not been studied in map turtle abundance between deadwood-rich river reaches that have ample adjacent forest compared to river reaches that have little adjacent forest and less deadwood (Lindeman 2019, p. 31).

An early account with combined data for Pearl River map turtles and Pascagoula map turtles of all sizes suggested that the turtles select logs or debris emerging from deep water as basking structures and dive into the water at the slightest disturbance (Cagle 1952, p. 227), but demographic differences in basking behavior have not been studied in Pearl River map turtles. In several other species of map turtle, females occupy deeper waters farther from shore than males (Pluto and Bellis 1986, pp. 24-30; Craig 1992, pp. 43-49; Jones 1996, pp. 378-384). In Pascagoula map turtles, the megacephalic species one drainage to the east of Pearl River map turtle, the only well-documented habitat difference among demographic groups is that of basking structure choice (Selman and Lindeman 2015, pp. 789-793). In this study, observers found that adults typically basked on structures that had water between the structure and the bank; females preferred large logs or floating logs and basked less on smaller branches, while mature males basked on large logs and smaller branches about equally. Juveniles usually basked closer to the bank and preferred smaller branches and tangles, while tangles were rarely used by adult males or females. Adults have been observed basking directly on the bank during flood periods when deadwood structures were under high water.

Nesting habitat has been described. At a beach on the Pearl River downstream of the Strong River, a nest was found in fine sand 82 feet (ft) (25 meters (m)) from the water (Vogt et al. 2019, p. 557). Three confirmed Pearl River map turtle nests found on sandbars along the Pearl River were dug in relatively fine sand ranging from 23-180ft (7 to 55m) from the water's edge and averaging 5.2ft (1.6m) from the closest vegetation (Ennen et al. 2016, pp. 094.4-094.6). Another account states that nests are typically near the vegetation lines of sandbars (Anderson 1958, pp. 212-215).

Pearl River map turtles have been observed in several streams that were thought to be too shallow or narrow for the genus *Graptemys* by previous researchers (McCoy and Vogt 1979, p. 5; Selman et al. 2009, p. 34) and this indicates a potential difference in habitat selection. While the Pearl River map turtle is present near the same dense accumulations of deadwood and large sandbars as the ringed map turtle, the Pearl River map turtle ranges farther upstream into smaller tributaries (Lindeman 2013, p. 298) and notable differences in relative abundance between the two species have been documented in specific streams on several occasions. Pearl River map turtle density was greater on mainstem reaches and large tributaries than on small tributaries, but they were outnumbered by ringed map turtles in mainstem reaches (Lindeman 2019, pp. 13-18). A suggested explanation for the difference in tributary tolerance between Pearl River map turtles and ringed map turtles is dietary requirements; ringed map turtles graze on algae, sponges, and other invertebrates on submerged deadwood, a community that is sunlight dependent, while Pearl River map turtles rely more heavily on benthic mussels that may be more tolerant of shaded tributary streams (Lindeman 2019, pp. 22-23).

2.8 Population Structure

While some population structure information has been reported for Pearl River map turtles, demographic bias among the various survey methodologies is likely (basking surveys, basking traps, and hand capture from emergent basking structures at night). In addition, demographic differences between sexes and between age classes in habitat and basking behaviors indicate that ratio estimates based on visual surveys and collecting efforts may not accurately represent the actual population structure (Lindeman 2019, pp. 25-32). Male to female sex ratio reports for adult Pearl River map turtles vary. In 1952, Cagle reported that among 98 individuals captured via dipnetting at night along the Pearl River, 5 were adult females, 12 were adult males, 6 were immature females, and 75 were juveniles, indicating a general sex ratio of 1.1:1, an operational sex ratio (ratio of mature males to mature females) of 2.4:1 (Cagle 1952, p. 233). A 1999 basking survey study on the Bogue Chitto reported that females outnumbered males at most sites (up to 1:3) and that adults outnumbered juveniles up to 1:45 (Shively 1999, p. 4). A later study reported a general sex ratio for captures larger than minimum male size of 1.8:1 and an operational sex ratio of 4.6:1 (Lindeman 2019, pp. 18-25). Another study reported 39 individuals captured to be comprised of 28 males, 10 females, and 1 juvenile, but without indication of the level of maturity of the individuals (Selman and Jones 2017, pp. 29-34). Buhlmann reported observations of 34 basking individuals to be 47 percent mature females (2014, pp. 17-18). Typically, male map turtles mature in 2-3 years, while females mature much later (Lindeman 2013, p. 109). Maturity for adult female Pearl River map turtles may occur around 9 years of age (Vogt et al. 2019, pp. 557-558). Hatchlings and 1-year-old juveniles accounted for 18 percent of all Pearl River map turtles captured during trapping sessions (Lindeman 2019, p. 18-25), but most were captured via hand or dipnet, a method which may be demographically biased (Cagle 1952, p. 227). A study on another megacephalic species (the Pascagoula map turtle) suggested that variation in population structure at different sites may be driven by dietary differences and prey availability (Selman and Lindeman 2015, pp. 785-786), and prior surveys on other map turtle species have documented mostly males in degraded sections or impoundments (Buhlmann 2014, p. 11).

Pearl River map turtle population structure may be influenced by temperature-dependent sex determination (TDSD) during egg development in the nest, in which the sex determination of individuals is influenced by the temperature of incubation. While TDSD has not been studied in this species, it has been documented in the Ouachita map turtle (*Graptemys ouachitenis*), the false map turtle (*Graptemys pseudogeographica*) and the common map turtle (*Graptemys geographica*), and likely affects all members of the genus (Vogt and Bull 1982, entire; Vogt and Bull 1984, entire; Bull 1985, entire). In a laboratory study of 14 genera of turtles including the false and Ouachita map turtles, incubation temperatures of 77 degrees F (25 degrees C) yielded all males and incubation temperatures from 87.8 degrees F (31 degrees C) or higher yielded all females (Vogt and Bull 1982, entire).

2.9 Predators

Little has been documented on predators of this species. American alligators (*Alligator mississippiensis*) and humans are the only significant predators of mature individuals (Ennen et al. 2016, p. 094.6), and persecution by humans through shooting basking turtles has been related to local population reduction in some turtles (Marion 1986, p. 51; Buhlman and Gibbons 1997, p. 222). Predation of Barbour’s map turtle by the river otter has been reported (Sterrett 2009, p. 73) and this mammal may be a predator of juvenile and adult Pearl River map turtles as well. A juvenile Pearl River map turtle was found in the stomach contents of a spotted bass (*Micropterus punctulatus*) (Carr and Messinger 2002, pp. 201-202). Alligator snapping turtles also prey on small turtles, including *Graptemys*; turtle remnants were found in the gastrointestinal tract of alligator snapping turtles in Arkansas and Louisiana, including the Pearl River, and small turtles of multiple species have been identified as one of the most common prey items (Elsey 2009, p. 447). Nests and juveniles are likely to be the most vulnerable to predation, with the latter probably preyed upon by wading birds and small mammals (Selman and Lindeman 2015, p. 796, Lovich 2009, p. 306). Many large and small mammals such as raccoons, opossums, feral swine, and skunks, as well as snakes and bird species (such as crows) prey upon nests of other map turtle species, including the sympatric ringed map turtle (Lovich 2009, p. 306; Lindeman 2013, entire; Jones 2006, pp. 197-207), and are likely to be predators of Pearl River map turtles as well. Red imported fire ants have also been documented invading turtle nests in the southeastern United States and can cause nest failure and hatchling mortality (Buhlmann and Coffman 2001, entire).

2.10 Summary of Resource Needs

A summary of information related to the needs of the Pearl River map turtle's demographic life stage (or similar species as a proxy) is presented in Table 2.1., however more research is needed in some areas to improve our understanding of the Pearl River map turtle.

Table 2.1. Individual resource needs of the Pearl River map turtle.

Life Stage	Resources needed for individuals to complete each life stage	Resource Function	Information Source
Fertilized Eggs (Nesting: May-August; hatching: July-November)	- Presence of gravid females - Patches of fine sand adjacent to adult habitat with sparse vegetation, typically on sandbars - Adequate incubation temperatures to yield an appropriate sex ratio** - Adequate river flow to prevent nest mortality due to flooding**	Breeding	- Ennen et al. 2016, pp. 094.4-094.6 - Lindeman 2013, p. 275; Lovich et al. 2009, p. 029.4 - Valenzuela et al. 2019, pp. 2-8 - Horne et al. 2003, p. 732; Geller 2012, pp. 210-211; Dieter et al. 2014, pp. 112-117; Eisemberg et al. 2016, p. 6
Hatchlings (July-November)	- Adequate abundance of invertebrate prey* - Adequate abundance of emergent branches and tangles near the riverbank**	Feeding, Sheltering, Dispersal	- LovHarding ich et al. 2009, p. 029.4 - Lindeman 2019, pp. 22-31 - Selman and Lindeman 2015, pp. 789-793
Adult Males (2+ years) and **Juveniles** (1-2 years)	- Adequate abundance of insect prey - Emergent logs, branches, and tangles near the bank**	Feeding, Sheltering, Dispersal	- Cagle 1952, p. 228 - Selman and Lindeman 2015, pp. 789-793
Adult Females (9+ years) and **sub-adult Females** (2-9 years)	- Adequate abundance of native mussels or Asian clams - Deeper, sand or gravel-bottomed stretches for foraging - Emergent logs and branches **	Feeding, Sheltering, Dispersal	- Cagle 1952, p. 228 - Selman and Lindeman 2015, pp. 789-793

* information source combines Pearl River map turtle data with the Pascagoula map turtle

** data were collected on a different megacephalic map turtle species or all megacephalic map turtles.

*** data were collected on a different species of emydid turtle or all aquatic turtles.

CHAPTER 3 – SPECIES NEEDS FOR VIABILITY

In order to assess the current and future condition of the species it is necessary to identify the individual, population, and species needs (Figure 3.1). As defined earlier, resiliency is the ability to withstand stochastic disturbances. In this chapter, we consider the Pearl River map turtle's ecological needs at the individual, population and species level, and discuss these needs in relation to resiliency, redundancy, and representation.

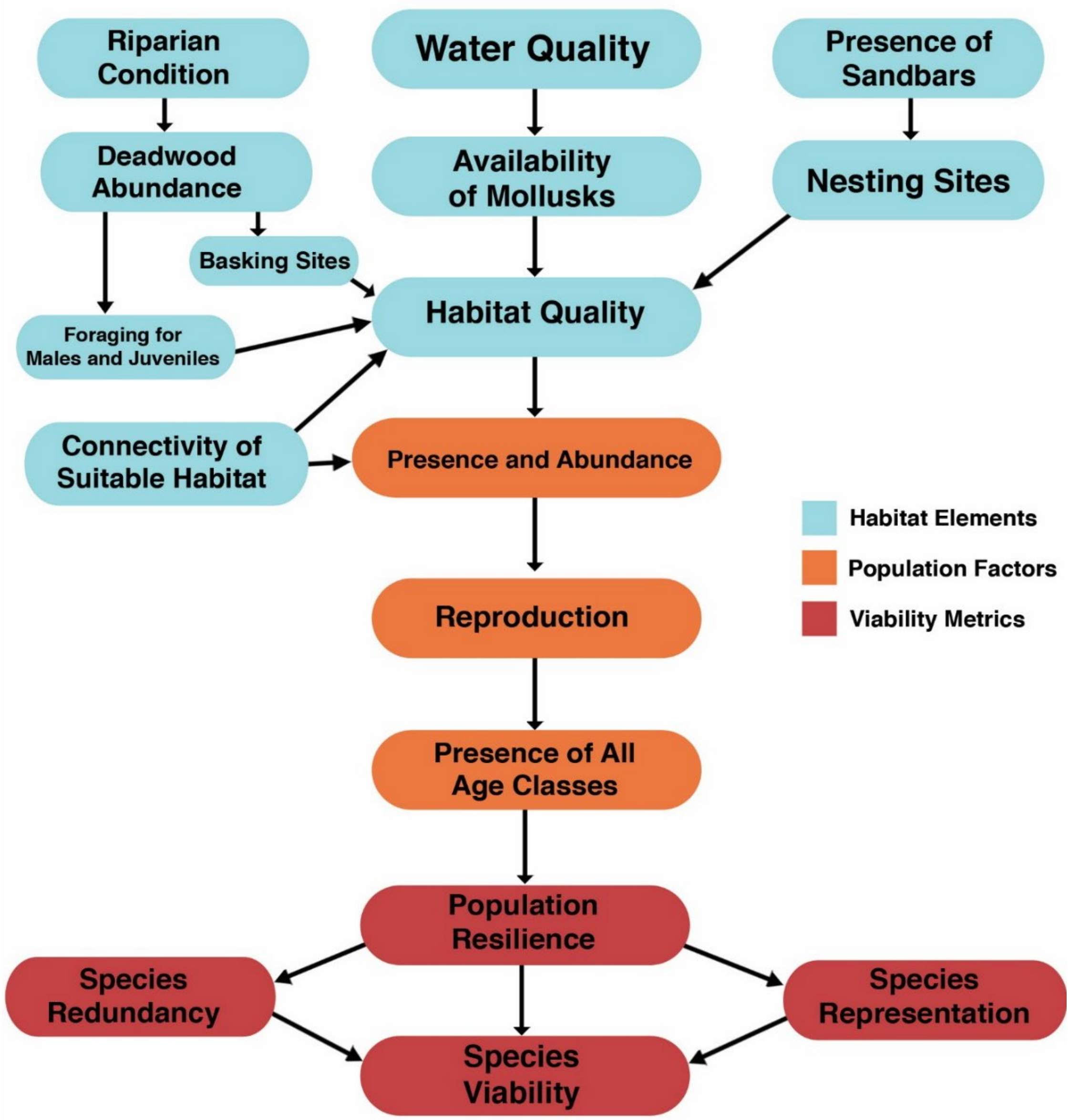

Figure 3.1. Influence diagram depicting the Pearl River map turtle's needs considering the primary habitat elements and population factors that influence the species' viability.

3.1 Individual Needs

For the Pearl River map turtle to survive and reproduce, individuals need suitable habitat that supports essential life functions at all life stages. Several elements appear to be essential to the survival and reproduction of individuals, as discussed in Section 2.10: mainstem and tributary reaches within the Pearl River system that have sandbars, adequate flow, adequate supply of invertebrate prey items including insects and mollusks, an abundance of emergent and floating basking structures of various sizes, and sand or gravel substrates.

3.2 Population Needs

For populations to be resilient, the needs of individuals (sandbars, natural hydrologic regimes, adequate supply of invertebrate prey items, basking structures, and sand, gravel, or rocky substrates) must be met at a larger scale. Tributary and mainstem reaches with suitable habitat uninterrupted by impoundments must be of adequate size to support a large enough population of individuals to avoid issues associated with small population sizes, such as inbreeding depression.

3.3 Species Needs

For the species to be viable, there must be adequate redundancy (suitable number of populations and connectivity to allow the species to withstand catastrophic events) and representation (genetic and environmental diversity to allow the species to adapt to changing environmental conditions). Redundancy improves with increasing numbers of populations (natural or reintroduced) distributed across the species range, and connectivity (either natural or human-facilitated) allows connected populations to "rescue" each other after catastrophes. Representation improves with the persistence of populations spread across the range of genetic and/or ecological diversity within the species. Long-term viability will require resilient populations to persist into the future; for the Pearl River map turtle, this will mean maintaining quality tributary and mainstem habitat and water quality to support many redundant populations across the species range, while preventing barriers to dispersal between populations such as dams or impoundments.

CHAPTER 4 – INFLUENCES ON VIABILITY

The following discussion provides a summary of the factors that have historically affected the species and are affecting or could be affecting the current and future condition of the Pearl River map turtle throughout some or all of its range. Research on delayed sexual maturity suggests that, as generally long-lived species, turtle populations may have a limited ability to respond to chronic stochastic disturbances such as prolonged flooding or drought (Congdon et al 1993, entire; Lovich et al. 2017, pp. 6-8). Effects due to climate change along with water quality issues provide major potential influences on the long-term viability of the species. Other factors that may influence species' viability are collection for the pet trade, invasive species and data deficiency. Conservation measures are considered positive influences on the species' viability and are also summarized in this chapter.

4.1 Climate Change

In the southeastern United States, climate change is expected to result in a high degree of variability in climate conditions with more frequent drought, more extreme heat (resulting in increases in air and water temperatures), increased heavy precipitation events (e.g., flooding), more intense storms (e.g., frequency of major hurricanes increases), and rising sea level and accompanying storm surge (IPCC 2013, entire). Warming in the southeast is expected to be greatest in the summer which is predicted to increase drought frequency, while annual mean precipitation is expected to increase slightly, leading to increased flooding events (IPCC 2013, entire; Alder & Hostetler 2013, unpaginated). This variability in climate may affect ecosystem processes and communities by altering the abiotic conditions experienced by biotic assemblages resulting in potential effects on community composition and individual species interactions (DeWan, et al., 2010, p. 7). These changes have the potential to impact Pearl River map turtles and/or their habitat.

Climate change could intensify or increase the frequency of drought events, such as the one that occurred in the southeastern U.S. in 2007. Based on down-scaled climate models for the Southeast U.S., the frequency, duration, and intensity of droughts are likely to increase in the southeastern U.S. as a result of global climate change (Keellings and Engstrom 2019, pp. 4-6). Stream flow is strongly correlated with important physical and chemical parameters that limit the distribution and abundance of riverine species (Power et al. 1995, entire; Resh et al. 1988, pp. 438-439) and it affects the hydrology of the systems by regulating the ecological integrity of flowing water systems (Poff et al. 1997, p. 770).

Since 1996, there has been an increase in the frequency of hurricane landfalls in the southeastern United States that is predicted to continue for some years into the future (Goldenberg et al. 2001, p. 475; Emanuel 2005, entire; Webster et al. 2005, p. 1845). Individual storm characteristics play a large role in the types and temporal extent of impacts (Greening et al. 2006, p. 878). For example, direction and speed of approach, point of landfall and intensity, all influence the magnitude of storm surge and resultant flooding (Weisberg and Zheng 2006, p. 164) and consequent environmental damage. Areas with higher levels of anthropogenic land use activity are more susceptible to environmental damage and long-term effects from storms (Mallin and Corbett 2006, pp. 1057-1058).

As a result of climate change, the world's oceanic surface-waters and land are warming. Higher temperatures reduce the density of the water causing it to expand. This process of "thermal expansion," exacerbated by an influx of melt water from glaciers and polar ice fields, is causing sea levels to rise. During the 20th century, global sea level rose by 0.17m at an average annual rate of 0.002m per year, which was ten times faster than the average during the previous 3,000 years (IPCC 2007, p. 30-31). The rate of sea level rise continues to accelerate and is currently believed to be about 0.003m per year (Church and White 2006, pp. 2-4). It is estimated that sea level will rise by a further 0.18 to 0.59m by the century's end (IPCC 2007, p. 46). However, some research suggests the magnitude may be far greater than previously predicted due to recent rapid ice loss from Greenland and Antarctica (Overpeck et al. 2006, entire; Rignot and Kanagaratnam 2006, pp. 989-990). Accounting for this accelerated melting, sea level could rise by between 0.5 and 1.4m by 2100 (Rahmstorf et al. 2007, p. 709). Sea level rise is likely to impact downstream Pearl River map turtle populations, as local scenarios based on downscaled climate models predict between 2-10ft (0.6-3.0m) of sea level rise in the vicinity of the Pearl River (NOAA 2020, unpaginated).

Despite the recognition of climate effects on ecosystem processes, there is uncertainty about what the exact climate future for the southeastern United States will be and how the ecosystems and species in this region will respond. It should be recognized that the greatest threat to many species from climate change may come from synergistic and compounding effects. That is, factors associated with a changing climate may act as risk multipliers by increasing the risk and severity of more imminent threats.

The dual stressors of climate change and direct human impact have the potential to impact aquatic ecosystems by altering stream flows and nutrient cycles, eliminating habitats, and changing community structure (Moore et al. 1997, pp. 942). Increased water temperatures and a reduction in stream flow are the climate change effects that are most likely to affect stream communities (Poff, 1992, entire), and each of these variables is strongly influenced by land use patterns. For example, in agricultural areas, lower precipitation may trigger increased irrigation resulting in reduced stream flow (Backlund et al. 2008, pp. 42-43). In forested areas, logging patterns influence instream temperatures through the direct effects of shading. Reductions in temperature by vegetative cover may be particularly important in low-order streams, where canopy vegetation significantly reduces the magnitude and variation of the stream temperature compared with that of clear-cut regions (Ringler and Hall 1975, pp. 111-121).

While river flooding under natural hydrologic conditions may be important for sandbar construction and deposition of nesting sand on riverine beaches (Dieter et al. 2014, p. 112-117), an increase in hurricane frequency and stochastic catastrophic floods could cause an increase in nest mortality. Nest mortality from flooding has not been studied in the Pearl River map turtle but has been documented in several other riverine turtle species. A study on the sympatric yellow-blotched map turtle revealed that nest mortality from flooding can be as high as 86.3 percent in some years (Horne et al. 2003, p. 732). In a study on nests of the Ouachita map turtle (*Graptemys ouachitensis*), two ten-day floods (in 2008 and 2010) were believed to have caused the complete mortality of all nests existing before the floods, as hatchlings were found dead inside eggs after the flood. However, a shorter flooding event in 2011 (~4 days of inundation) caused no known nest mortalities (Geller 2012, p. 210-211). A study on freshwater turtles in South America indicated that as flooding incidents have increased since the 1970's, the number of days that nesting sandbars remain above the inundation threshold has been steadily and significantly decreasing, causing steep declines in the number of hatchlings produced per year (Eisemberg et al. 2016, p. 6).

Another area where climate change may affect the viability of the Pearl River map turtle is through TDSD during egg development in the nest. In turtle species with TDSD, increasing seasonal temperatures may result in unnatural sex ratios among hatchlings. This could be an important factor as climate change drives increasing temperatures. Since male map turtles with TDSD develop at lower temperatures than females, rising temperatures during developmental periods may result in sex ratios that are increasingly female-biased.

4.2 Water Quality

Degradation of stream and wetland systems through reduced water quality and increased concentrations of contaminants can affect the occurrence and abundance of freshwater turtles (DeCatanzaro and Chow-Fraser 2010, p. 360). Infrastructure development increases the percentage of impervious surfaces, reducing and degrading terrestrial and aquatic habitats. Increased water volume and land-based contaminants (e.g. heavy metals, pesticides, oils) flow into aquatic systems, modifying hydrological and sedimentation regimes of rivers and wetlands (Walsh et al. 2005, entire). Aquatic toxicants can have both immediate and long-term negative impacts on species and ecosystems. Despite these effects, species vary widely in their tolerances and abilities to adapt to water quality degradation, including variation in stress and immune responses (French et al.2008, pp. 5-6), population structure (Patrick and Gibbs 2010, pp. 795-797), survival and recruitment (Eskew et al. 2010, pp. 368-371), and ultimately distribution and abundance (Riley et al. 2005, pp. 6-8).

Sedimentation and pollution can have adverse impact on the mollusk populations that the closely-related Pascagoula map turtles prey upon (Box and Mossa 1999, entire). Inputs of point (point source discharge from particular pipes, discharges, etc.) and nonpoint (diffuse land surface runoff) source pollution across the range are numerous and widespread. Point source pollution can be generated from inadequately treated effluent from industrial plants, sanitary landfills, sewage treatment plants, active surface mining, drain fields from individual private homes, and others (USFWS 2000, pp. 14-15). Nonpoint pollution originates from agricultural activities, poultry and cattle feedlots, abandoned mine runoff, construction, silviculture, failing septic tanks, and contaminated runoff from urban areas (Deutsch et al. 1990, entire; USFWS 2000, pp. 14-15). These sources contribute pollution to streams via sediments, heavy metals, fertilizers, herbicides, pesticides, animal wastes, septic tank and gray water leakage, and oils and greases. Glyphosate (found in Roundup and other herbicides) that is widely used as an herbicide has been found in many waterways from agricultural run-off and exposure has been associated with endocrine and reproductive disorders in animals (Jerrell et al. 2020, entire; Medalie et al 2020, entire; Mesnage et al. 2015, entire). Water quality and native aquatic fauna decline as a result of this pollution, which causes nitrification, decreases in dissolved oxygen concentration, and increases in acidity and conductivity. These alterations likely have direct (e.g. decreased survival and/or reproduction) and indirect (e.g. loss, degradation, and fragmentation of habitat) effects. For aquatic species, submergent vegetation provides critical spawning habitat for adults, refugia from predators, and habitat for prey of all life stages (Jude and Pappas 1992, pp. 666-667), and degraded water quality and high algal biomass that result from pollutant inputs, cause loss of these critical submergent plant species (Chow-Fraser et al.1998, pp. 38-39).

A wide range of current activities and land uses can lead to sedimentation within streams that can include: agricultural practices, construction activities, stormwater runoff, unpaved roads, forestry activities, utility crossings, and mining. Fine sediments are not only input into streams during these activities, historical land use practices may have substantially altered hydrological and geological processes such that sediments continue to be input into streams for several decades after those activities cease (Harding et al. 1998, p. 14846). The negative effects of increased sedimentation are well understood for aquatic species (Burkhead et al. 1997, p. 411; Burkhead and Jelks 2001, p. 964). Sedimentation can alter food webs and stream productivity (Schofield et al. 2004, p. 907), force altered behaviors (Sweka and Hartman 2003, p. 346), and even have sub-lethal effects and mortality on individual aquatic organisms (Sutherland and Meyer 2007, p. 394; Wenger and Freeman 2007, p. 7).

Degradation of water quality from municipal and industrial effluents is recognized as a cause of decline in the ringed map turtle, a sympatric endangered species (Lindeman 1998, p. 137). Researchers also recorded lower numbers of ringed map turtles near gravel and sand mining operations (Shively 1999, p. 10). Native mussel and gastropod populations have likely already decreased due to sedimentation and other anthropogenic alterations (Jones at al. 2005, entire). Water quality may have impacted the Pearl River map turtle populations around Jackson, Mississippi (Selman 2020d, p. 194).

Water quality for the Pearl River map turtle is impacted by four processes that are further discussed below: channel and hydrology modifications and impoundments, agriculture, development (urbanization), and mining.

4.2.1 Channel and Hydrology Modifications and Impoundments

Dredging and channelization have led to loss of aquatic habitat in the Southeast (Warren Jr. et al. 1997, unpaginated). Dredging and channelization projects are extensive throughout the region for flood control, navigation, sand and gravel mining, and conversion of wetlands into croplands (Neves et al. 1997, unpaginated; Herrig and Shute 2002, pp. 542-543). Many rivers are continually dredged to maintain a channel for shipping traffic. Dredging and channelization modify and destroy habitat for aquatic species by destabilizing the substrate, increasing erosion and siltation, removing woody debris, decreasing habitat heterogeneity, and stirring up contaminants which settle onto the substrate (Williams et al. 1993, pp. 7-8; Buckner et al. 2002, entire; Bennett et al. 2008, pp. 467-468). Channelization can also lead to headcutting, which causes further erosion and sedimentation (Hartfield 1993, pp. 131-141). Dredging removes woody debris which provides cover and nest locations for many aquatic species (Bennett et al. 2008, pp. 467-468). Anthropogenic deadwood removal has been noted as a reason for decline in a microcephalic species, the ringed map turtle (Lindeman 1998, p. 137). Snags and logs are removed from some sites to facilitate boat navigation (Dundee and Rossman 1989, p. 187). Experiments with manual deposition of deadwood in stretches with less riparian forest have been suggested as potential habitat restoration measures (Lindeman 2019, p. 33).

Stream channelization, point-bar mining, and impoundments have been listed as potential threats in a report from before the Pascagoula map turtle and Pearl River map turtle were taxonomically separated (USFWS 2006, p. 2). Channel modification is recognized as a cause of decline in the ringed map turtle, a sympatric endangered species (Lindeman 1998, p. 137). Considerably low densities of Pearl River map turtles were observed in the lower reaches of the Pearl, where much channelization and flow diversion has occurred (Lindeman 2019, pp. 23-29).

Impoundment of rivers is a primary threat to aquatic species in the southeast (Folkerts 1997, p. 11; Buckner et al. 2002, entire). Dams modify habitat conditions and aquatic communities both upstream and downstream of an impoundment (Winston et al. 1991, pp. 103-104; Mulholland and Lenat 1992, pp. 193-231; Soballe et al. 1992, pp. 421-474). Upstream of dams, habitat is flooded and in-channel conditions change from flowing to still water, with increased depth, decreased levels of dissolved oxygen, and increased sedimentation. Sedimentation alters substrate conditions by filling in interstitial spaces between rocks which provide habitat for many species (Neves et al. 1997, unpaginated). Downstream of dams, flow regime fluctuates with resulting fluctuations in water temperature and dissolved oxygen levels, the substrate is scoured, and downstream tributaries are eroded (Schuster 1997, unpaginated; Buckner et al. 2002, unpaginated). Negative "tailwater" effects on habitat can extend many kilometers downstream (Neves et al. 1997, unpaginated). Dams fragment habitat for aquatic species by blocking corridors for migration and dispersal, resulting in population geographic and genetic isolation and heightened susceptibility to extinction (Neves et al. 1997, unpaginated). Dams also preclude the ability of aquatic organisms to escape from polluted waters and accidental spills (Buckner et al. 2002, unpaginated).

Damming of streams and springs is extensive throughout the southeast (Etnier 1997, unpaginated; Morse et al. 1997, unpaginated; Shute et al. 1997, unpaginated). Shute et al. (1997) report that "few Southeastern streams are spared from impoundment" (p. 458). Many streams have both small ponds in their headwaters and large reservoirs in their lower reaches. Small streams on private lands are regularly dammed to create ponds for cattle, irrigation, recreation, and fishing, with significant ecological effects due to the sheer abundance of these structures (Morse et al. 1997, unpaginated). Small headwater streams are increasingly being dammed in the southeast to supply water for municipalities (Buckner et al. 2002, unpaginated) and many southeastern springs have also been impounded (Etnier 1997, unpaginated). Dams are known to have caused the extirpation and extinction of many southeastern species, and existing and proposed dams pose an ongoing threat to many aquatic species (Folkerts 1997, p. 11; Neves et al. 1997, unpaginated; USFWS 2000, p. 15; Buckner et al. 2002, unpaginated).

The Ross Barnett Reservoir was constructed between 1960 and 1963 and provides a water supply for the City of Jackson and the associated area, as well as recreational opportunities on the 33,000 ac (13,355 ha) lake and the 17,000 ac (6,880 ha) surrounding it (Pearl River Valley Water

Management District 2020, entire). A total of 33.6 km (20.9 rm) of the Pearl River is submerged beneath the Ross Barnett Reservoir, which has 61.5 km (38.2 rm) of shoreline (Lindeman 2019, p. 19). Low population densities of Pearl River map turtles have been observed upstream of the Ross Barnett Reservoir, possibly due to recreational boating and extended recreational foot traffic or camping on sandbars by reservoir visitors (Selman and Jones 2017, pp. 32-34). Notable population declines also have been observed in the stretch of the Pearl River downstream of the Ross Barnett Reservoir (north of Lakeland Drive), but the exact reason for the decline is unknown (Selman 2020d, p. 194). Researchers have estimated that up to 170 individual Pearl River map turtles could be directly impacted by the One Lake Project (Selman 2020d, p. 192).

4.2.2 Agriculture

Agricultural practices such as traditional farming, feedlot operations, and associated land use practices can contribute pollutants to rivers. These practices degrade habitat by eroding stream banks, which results in alterations to stream hydrology and geomorphology. Nutrients, bacteria, pesticides, and other organic compounds are generally found in higher concentrations in areas affected by agricultural than forested areas. Contaminants associated with agriculture (e.g., fertilizers, pesticides, herbicides, and animal waste) can cause degradation of water quality and habitats through instream oxygen deficiencies, excess nutrification, and excessive algal growths, with a related alteration in aquatic community composition (Petersen et al. 1999, p. 6). Agricultural development can also reduce the amount of adjacent riparian forest available to produce deadwood; in another megacephalic map turtle species (Barbour's map turtle), abundance decreased in areas where adjacent riparian corridors had been disturbed by agriculture while the abundance of the red-eared slider (*Trachemys scripta*), a cosmopolitan species, increased (Sterrett et al. 2011, entire).

Agricultural practices such as use of glyphosate-based herbicides for weed control and animal waste for soil amendment are becoming common in many regions, and pose threats to biotic diversity in freshwater systems. Over the past two decades, these practices have corresponded with marked declines in populations of fish and mussel species in the Upper Conasauga River watershed in Georgia/Tennessee (Freeman et al. 2017, p. 419). A study in this watershed showed that nutrient enrichment of streams was widespread with nitrate and phosphorus exceeding levels associated with eutrophication, and hormone concentrations in sediments were often above those shown to cause endocrine disruption in fish, possibly reflecting widespread application of poultry litter and manure (Lasier et al. 2016, entire). Researchers postulate that species declines observed in the Conasauga watershed may be at least partially due to hormones, as well as excess nutrients and herbicide surfactants (Freeman et al. 2017, p. 429).

4.2.3 Development

Urbanization is a significant source of water quality degradation that can reduce the survival of aquatic organisms. Urban development can stress aquatic systems in a variety of ways, including increasing the frequency and magnitude of high flows in streams, increasing sedimentation and nutrient loads, increasing contamination and toxicity, decreasing the diversity of fish, aquatic insects, plants, and amphibians, and changing stream morphology and water chemistry (Coles et al. 2012, entire; CWP 2003; entire). Sources and risks of an acute or catastrophic contamination event, such as a leak from an underground storage tank or a hazardous materials spill on a highway or by train, increase as urbanization increases.

4.2.4 Mining

The rapid rise in urbanization and construction of large-scale infrastructure projects are driving increasing demands for construction materials such as sand and gravel. Rivers are a major source of sand and gravel because transport costs are low; river energy produces the gravel and sand, thus eliminating the cost of mining, grinding, and sorting rocks; and the material produced by rivers tends to consist of resilient minerals of angular shape that are preferred for construction (Koehnken et al. 2020, p. 363). Impacts of sand and gravel mining can be direct or indirect. Direct impacts include physical changes to the river system and the removal of gravel and floodplain habitats from the system. Indirect impacts include shifting of habitat types due to channel and sedimentation changes; changes in water quality, which change the chemical and physical conditions of the system; and hydraulic changes which can impact movement of species and habitat availability.

Gravel mining is a major industry in southeastern Louisiana, particularly along the Bogue Chitto River (Selman 2020c, p. 20). In-stream and unpermitted point bar mining was observed in the late

1990s, and were considered to be the biggest concern for *Graptemys* species in the Bogue Chitto River (Shively 1999, pp. 10-11). Gravel mining is perhaps still the greatest threat to the Pearl River system in southeastern Louisiana (Selman 2020c, p. 20). A recent comparison of aerial imagery from the mid-1908s and late 1990s with images from 2019 reveal increases in distribution and magnitude of gravel mines in the Bogue Chitto River system, and recent surveys have reported several areas where mining appears to have degraded water quality significantly (Selman 2020c, pp. 20-21, and p. 40).

4.3 Other Influences

Several other influences may be impacting the viability of the species, including the introduction and persistence of invasive species, collection for the pet trade, potential impacts of disease, and the lack of understanding of important conservation components (data deficiency).

4.3.1 Invasive Species

It is estimated that 42% of Federally Threatened or Endangered species are significantly impacted by nonnative nuisance species across the nation and nuisance species are significantly impeding recovery efforts for them in some way (NCANSMPC 2015, pp.8-9). There are many areas across the Southeastern United States where aquatic invasive species have invaded aquatic communities; are competing with native species for food, light, or breeding and nesting areas; and are impacting biodiversity. When an invasive species is introduced it may have many advantages over native species, such as easy adaptation to varying environments and a high tolerance of living conditions that allows it to thrive in its nonnative range. There may not be natural predators to keep the invasive species in check; therefore, it can potentially live longer and reproduce more often, further reducing the biodiversity in the system. The native species may become an easy food source for invasive species, or the invasive species may carry diseases that wipe out populations of native species or displace the native species by consuming the resources that are needed to survive. The degree to which invasive species effect the Pearl River map turtle has not been studied, but the diet of mature females may have been broader before the introduction of Asian Clams (*Corbicula* *fluminea*) and removal of invasive vegetation on sandbars has been suggested as nesting habitat management (Selman and Lindeman 2015, p. 794-795; Lindeman 2019, p. 33).

4.3.2 Collection

Due to the intricacy of the shell morphology, map turtles are popular in the pet trade (USFWS 2006, p. 2), both domestically and internationally. An analysis of online marketplace offerings in Hong Kong revealed that interest in turtles as pets is increasing, that many of the species offered for sale are from North America, and that there is a higher interest in rare species (Sung and Fong 2018, p. 221). The common map turtle (*Graptemys geographica*) is one of three key species in the international trade of wildlife, with individuals being sold both as pets and incorporated into Chinese aquaculture for consumption (Luiselli at al. 2016, p. 170). Exploitation of Pearl River map turtles for the pet trade domestically and in Asian markets has been documented, but the degree of impact is unclear, as it is unknown whether captive individuals were Pascagoula map turtles or Pearl River map turtles (Lindeman 1998, p. 137; Cheung and Dudgeon 2006, p. 756; USFWS 2006, p. 2; Selman and Qualls 2007, p. 32-34; Ennen et al. 2016, p. 094.6). In a recent report of surveys conducted in the lower Pearl River in Louisiana, Selman (2020, p. 23) stated his belief that wild turtles are being captured and collected in the Pearl River system, and similar to what has been observed in other states, they are likely destined for the high-end turtle pet trade in China and possibly other southeast Asian countries. Selman (2020, pp. 22-23) received information from three different local individuals, at three different locations, concerning turtle bycatch or harvesting in local Louisiana waterways occupied by Pearl River map turtles. These locations included the Pearl River south of Bogalusa (possible mortality resulting from bycatch in hoop nets), the West Pearl River Navigation Canal (turtles captured and sold, possibly for shipment to China), and the Bogue Chitto River (local comment that baby turtles were being captured and shipped to China). The specific species captured were not documented, however it is likely that at least some of these turtles were Pearl River map turtles.

U.S. Fish and Wildlife Service records information related to species exports in the Law Enforcement Management Information System (LEMIS). According to a LEMIS report from 2005 to 2019, there were over 300,000 turtles identified as *Graptemys* spp. or their parts exported from the United States into 29 countries (see APPENDIX B). The number of turtles recorded in each shipment ranged widely. Due to the similarity in appearance, species of *Graptemys* are difficult to differentiate. However, in 2005 when the highest number of *Graptemys* were exported (APPENDIX B), there were records of over 35,000 turtles (*Graptemys* spp.) in a single shipment to Spain and a total of 172,645 individual *Graptemys* exported to 24 different countries. There is some uncertainty in the sources of the exported turtles as they could have originated from captive stock or wild-caught. The status and life history of map turtles makes them particularly vulnerable to overharvesting and it has been suggested that the trade in wild-caught animals should be halted or severally reduced (Schlaepfer et al. 2005, p. 263).

Wanton shooting of turtles (documented for *Graptemys* sp.) may also impact populations (Buhlmann and Gibbons 1997, p. 222; Lindeman 1998, p. 137; USFWS 2006, p. 2), although it is not frequently reported, and thus difficult to study and/or quantify.

4.3.3 Disease

Ranaviruses are capable of infecting turtles. Aquatic turtles share habitat with susceptible fish and amphibian populations and as a result may be more at risk of infection than terrestrial turtles (Wirth *et al.* 2018, p. 6). Ranaviral infections are systemic, and there is often extensive damage to multiple organs during infection, especially the liver and spleen in reptiles (Wirth et al. 2018, p. 8). Susceptibility to disease may be increased by immunosuppressive effects to immune systems of aquatic turtles due to stressors such as pesticides, herbicides, and heavy metals that enter aquatic systems (Wirth *et al.* 2018, p. 13). Ranaviruses are likely underreported in turtles due to lack of awareness, few long-term studies on the pathology of disease in turtles, and minimal population monitoring (Wirth *et al.* 2018, p. 7). Few data are available for ranaviruses in *Graptemys* species; however, Mississippi map turtles (*G. pseudogeographica kohni)* have been infected under experimental conditions (Brenes *et al.* 2014, entire). Evidence of shell damage caused by disease has been documented in another megacephalic map turtle (*G. barbouri*), but the underlying disease was unknown (Lovich et al. 1996, p. 261). More recently, a new species of fungal pathogen (Order Onygenales) was isolated from shell lesions on freshwater aquatic turtles (Woodburn et al. 2019, entire).

4.3.4 Data Deficiency

Data deficiency limits our understanding of this species, and a 2006 article ranking North American turtle and tortoise species by number of publications listed *G. gibbonsi* (before the Pearl River map turtle was taxonomically split and therefore combining publications on both species) as only the 46th of 58 species (Lovich and Ennen 2013, p. 22), indicating that 78% of the turtle and tortoise species in North America have been studied more thoroughly. Selman and Jones attributed the data deficiency on the Pearl River map turtle to it being overshadowed by the listing status of and subsequent funding direction toward the sympatric ringed map turtle, as well as the prior taxonomy of the Pearl River map turtle encompassing a much larger range (2017, p. 27). A recent report acknowledged demographic bias issues with prior studies, summarized suggestions for future research, and acknowledged that variation in survey type and location make it difficult to tell whether apparent changes in abundance over time are the result of actual declines or the result of data collection bias (Lindeman 2019, pp. 25-32). Surveys have not been standardized for long-term monitoring of *G. pearlensis*. Much of what we know about *G. pearlensis* is based on opportunistic natural history observations, and many aspects of their biology have not been studied in detail, including fecundity, spatial ecology, microhabitat requirements, estimates of survivorship, diet via stable isotope analyses, temperature-dependent sex determination, salinity tolerance, disease, nest predation and flood mortality, demographic differences in habitat use (beyond basking structure), and the population-level impacts of anthropogenic activities and natural disasters.

4.4 Conservation Measures

4.4.1 Federal

The Clean Water Act of 1972 regulates dredge and fill activities that would adversely affect wetlands. Such activities are commonly associated with dry land projects for development, flood control, and land clearing, as well as for water dependent projects such as docks/marinas and maintenance of navigational channels. The U.S. Army Corps of Engineers (Corps) and the Environmental Protection Agency (EPA) share the responsibility for implementing the permitting program under Section 404 of the Clean Water Act. Permit review and issuance follows a process that encourages avoidance, minimizing and requiring mitigation for unavoidable impacts to the aquatic environment and habitats. This includes protecting the riverine habitat occupied by the Pearl River map turtle. These laws have resulted in some enhancement of water quality and habitat for aquatic life, particularly by reducing point-source pollutants.

The Endangered Species Act (Act) has as its primary purpose the conservation of endangered and threatened species and the ecosystems upon which they depend. Section 4 of the Act requires the Service to develop and implement recovery plans for the conservation of listed species in order to address the threats to their survival and recovery. Section 7 of the Act requires Federal agencies to evaluate their actions with respect to any listed species and their critical habitat, if designated. If a Federal action may affect a listed species or its critical habitat, the responsible Federal agency must enter into consultation with the Service and ensure that activities they authorize, fund, or carry out are not likely to jeopardize the continued existence of any listed species or destroy or adversely modify its critical habitat.

The Pearl River map turtle likely receives ancillary protection (i.e., water quality improvements, protection from proposed geomorphological changes) where it co- occurs with three other species listed under the Endangered Species Act of 1973 (Act), the inflated heelsplitter (*Potamilus inflatus)* (U.S. Fish and Wildlife Service 1990, entire), Gulf sturgeon (*Acipenser oxyrhynchus desotoi*) (U.S. Fish and Wildlife Service 1991, entire), and ringed map turtle (*G. oculifera*) (U.S. Fish and Wildlife Service 1986, entire). Conservation measures that could benefit the Pearl River map turtle include when projects affecting one of these listed species results in consultation that reduces or mitigates impacts. In addition, critical habitat is designated for the Gulf sturgeon (U.S. Fish and Wildlife Service 2003, entire) and includes areas of the Pearl River and Bogue Chitto River occupied by the Pearl River map turtle. Federal agency actions within the Pearl River map turtle's habitat that may require consultation due to affects to the above listed species include maintenance dredging for navigation in the lower Pearl River by the Corps and their issuance of section 404 Clean Water Act permits; construction and maintenance of gas and oil pipelines and power line rights- of-way by the Federal Energy Regulatory Commission; EPA pesticide registration; construction and maintenance of roads or highways by the Federal Highway Administration; and funding of various projects administered by the U.S. Department of Agriculture's Natural Resources Conservation Service and the Federal Emergency Management Agency.

In 2000, the State Wildlife Grants (SWG) Program was created through the Fiscal Year 2001 Interior Appropriates Act and provided funding to states "for the development and implementation of programs for the benefit of wildlife and their habitat, including species that are not hunted or fished." The SWG Program is administered by the Service's Division of Federal Aid and allocates federal funding for proactive nongame conservation measures nationwide. Congress stipulated that each state fish and wildlife agency that wished to participate in the SWG program develop a Wildlife Action Plan to guide the use of SWG funds (see discussion below regarding the plans developed by the Louisiana Department of Wildlife and Fisheries (LDWF) and Mississippi Department of Wildlife, Fisheries, and Parks (MDWFP).

The National Wildlife Refuge System Administration Act (NWRAA) represents organic legislation that set up the administration of a national network of lands and water for the conservation, management, and restoration of fish, wildlife, and plant resources and their habitats for the benefit of the American people. Conservation-minded management of public lands allows for: 1) natural processes to operate freely and thus changes to habitat occur due to current and future environmental conditions; 2) managing the use of resources and activities which minimizes impacts; 3) preservation and restoration to maintain habitats; and 4) reduction of the adverse physical impacts from human use. Amendment of the NWRAA in 1997 required the refuge system to ensure that the biological integrity, diversity, and environmental health of refuges be maintained. The Pearl River map turtle occurs on the Bogue Chitto National Wildlife Refuge which occupies Pearl River County in Mississippi and St. Tammany and Washington Parishes in Louisiana. A Comprehensive Conservation Plan (CCP) has been developed to provide the framework of fish and wildlife management on the refuge (U.S. Fish and Wildlife Service 2011, entire). Within the CCP, specific actions are described to protect the ringed map turtle that will also benefit the Pearl River map turtle. Actions include ongoing habitat management to provide downed woody debris for basking turtles and to maintain 330ft (100.6m) buffers along all named streams during forest habitat improvement and harvest to protect water quality in streams (U.S. Fish and Wildlife Service 2011, p. 21, 73, 89, 179).

The National Forests in Mississippi have adopted, and in most cases exceeded, the best management practices (BMPs) (see discussion below of state BMPs) established by the state of Mississippi (U.S. Forest Service 2014, p. 66). These include practices such as establishing streamside buffer zones, restricting vegetation management in riparian zones, and employing erosion control measures. The Bienville National Forest has no known records for the Pearl River map turtle, but does contain tributaries that flow into the Pearl and Strong Rivers thus these practices may provide some protective measures for habitat occupied by the species downstream.

The Sikes Act Improvement Act (1997) led to Department of Defense guidance regarding development of Integrated Natural Resources Management Plans (INRMP) for promoting environmental conservation on military installations. The U.S. Navy operates the Stennis Western Maneuver Area (Stennis WMA) located along the western edge of the NASA Stennis Space Center and incorporated into the Stennis Space Center Buffer Zone. The Stennis WMA encompasses a 4-mile reach of the East Pearl River and a smaller eastern tributary named Mikes River (Buhlman 2014, p. 4) in Hancock and Pearl River Counties, Mississippi. These river reaches are used by the Navy's Construction Battalion Center for riverboat warfare training. The western bank of the East Pearl River denotes the boundary of the Navy property and is managed as the Pearl River Wildlife Management Area by the state of Louisiana (see below under State/Louisiana). There are records of the Pearl River map turtle from Stennis WMA (Buhlman 2014, pp. 11-12, 31-32). The U.S. Navy has developed an INRMP for the Stennis WMA (U.S. Navy 2011, entire). Measures within the INRMP expected to protect listed species, and also provide a level of protection for the Pearl River map turtle, include erosion and storm water control, floodplain management, invasive plant species management, and the use of an ecosystem approach to general fish and wildlife management (U.S. Navy 2011, pp. 4-4 through 4-20). The work summarized in the Buhlman report (2014, entire) was conducted as part of rare, threatened and endangered species surveys implemented in compliance with the Act and described in the INRMP (U.S. Navy 2011, p. 9).

4.4.2 Convention on International Trade in Endangered Species of Wild Fauna and Flora-Appendix III

All species of *Graptemys* are included on the Convention on International Trade in Endangered Species of Wild Fauna and Flora's (CITES) Appendix III (CITES 2019, p. 43). The Pearl River map turtle was added to the CITES Appendix III list in 2006 (USFWS 2005, entire). Appendix III is a list of species included at the request of a Party that already regulates trade in the species and needs the cooperator of other countries to prevent unsustainable of illegal exploitation. International trade in specimens of species listed in this Appendix is allowed only on presentation of the appropriate permits or certificates.

4.4.3 State/Louisiana

The Pearl River map turtle is globally ranked G2G3 (imperiled) and state-ranked S3 (rare and local throughout the state or found in a restricted region of the state, or because of other factors making it vulnerable to extirpation) in Louisiana (Holcomb et al. 2015, p. 624). Protections under state law for collecting the Pearl River map turtle are limited to licensing restrictions for turtles. In Louisiana, a recreational basic fishing license is required but allows unlimited take of most species of turtles, including the Pearl River map turtle; exceptions are that no turtle eggs or nesting turtles may be taken (LDWF 2020a, p. 50-51). A recreational gear license would also be required for operating specified trap types (see Louisiana's regulations for details on trap types), for instance, five or less hoop nets; greater than five hoop nets requires a Commercial Fisherman License. The Pearl River map turtle has no state status under Louisiana regulations or law (LDWF 2020b, entire).

Louisiana Scenic Rivers Act (1988) was established as a regulatory program which is administered by LDWF through a system of regulations and permits. Certain actions that may negatively affect the Pearl River map turtle are either prohibited or require a permit on rivers included on the natural and scenic river list. Prohibited actions include channelization, channel realignment, clearing and snagging, impoundments, and commercial clearcutting within 100ft (30.5m) of the river low water mark (Louisiana Department of Agriculture and Forestry (LDAF) undated, p. 45). Permits are required for river crossing structures, bulkheads, land development adjacent to the river, and water withdrawals (LDAF undated, p. 45). Rivers with the natural and scenic river designation which are occupied by the Pearl River map turtle include the Bogue Chitto River, Holmes Bayou, and West

Pearl River in St. Tammany Parish; and Pushepatapa Creek in Washington Parish (LDAF undated, p. 48).

Additional protected areas of Pearl River map turtle habitat in Louisiana include the Pearl River Wildlife Management Area located in St. Tammany Parish and Bogue Chitto State Park located on the Bogue Chitto River in Washington Parish. A master plan for management of Wildlife Management Areas and State Refuges has been developed for Louisiana which describes the role of these lands in improving wildlife populations and their habitat including identifying and prioritizing issues threatening wildlife resources (LDWF and The Conservation Fund 2014, entire). Bogue Chitto State Park is managed by the Louisiana Department of Culture, Recreation, and Tourism to preserve and enhance Louisiana's heritage and natural landscape; provide cultural and recreational resources; and promote the use of these resources by the public (State of Louisiana 2019, p. 3).

The Louisiana State Comprehensive Wildlife Action Plan (Holcomb et al. 2015, entire) was developed as a roadmap for nongame conservation in Louisiana. The primary focus of the plan is the recovery of Species of Greatest Conservation Need (SGCN), those wildlife species in need of conservation action within Louisiana which include the Pearl River map turtle. Specific actions identified for the Pearl River map turtle include conducting ecological studies of the turtle's reproduction, nest success, and recruitment as well as developing general population estimates via mark and recapture studies (Holcomb et al. 2015, p. 69). Recent Pearl River map turtle survey work in Louisiana was conducting using funding from the SWG program (Selman 2020c, entire).

4.4.4 State/Mississippi

The Pearl River map turtle is globally ranked G2G3 (imperiled) and S2 (imperiled because of rarity or because of some factor making it very vulnerable to extinction) in Mississippi, (Mississippi Museum of Natural Science 2015, p. 38), but is not listed on the Mississippi state list of protected species (Mississippi Natural Heritage Program 2015, entire). Protections under state law are limited to licensing restrictions for take for personal use of nongame species in need of management (which includes native species of turtles). A Mississippi resident is required to obtain one of three licenses for capture and possession of Pearl River map turtles (Mississippi Commission on Wildlife, Fisheries, and Parks, Mississippi Department of Wildlife, Fisheries, and Parks 2016, pp. 3-5). The three licenses available for this purpose are a Sportsman License, an All Game Hunting/Freshwater Fishing License, and a Small Game Hunting/Freshwater Fishing License. A Nonresident would require a Nonresident All Game Hunting License (Mississippi Department of Wildlife, Fisheries, and Parks 2020, entire). Restrictions on take for personal use include that no more than four turtles of any species or subspecies may be possessed or taken within a single year, and that no turtles may be taken between April 1st and June 30th except by permit from the Mississippi Department of Wildlife, Fisheries, and Parks (Mississippi Commission on Wildlife, Fisheries, and Parks, Mississippi Department of Wildlife, Fisheries, and Parks 2016, pp. 3-5).

The Mississippi Comprehensive Wildlife Action Plan (MMNS 2015, entire) was developed to provide a guide for effective and efficient long-term conservation of biodiversity in Mississippi. As in Louisiana, the primary focus of the plan is on the recovery of species designated as SGCN which includes the Pearl River map turtle. Specific actions identified for the Pearl River map turtle in Mississippi include planning and conducting status surveys for the species (MMNS 2015, p. 686).

Lands managed for wildlife by the state of Mississippi, which may provide habitat protections for the Pearl River map turtle, include the Old River Wildlife Management Area, Pearl River County and Pearl River Wildlife Management Area, Madison County. In addition, a ringed map turtle sanctuary was designated in 1990 by the Pearl River Valley Water Supply District (District) north of the Ross Barnett Reservoir, Madison County. One of the goals of management on Wildlife Management Areas in Mississippi is to improve wildlife populations and their habitat (MDWFP 2020, entire). The District sanctuary is approximately 19.3 rkm (12 rm) north from Ratliff Ferry to Lowhead Dam on the Pearl River (U.S. Fish and Wildlife Service 2010, p. 4). Within the sanctuary, the District is required to maintain informational signs to facilitate public awareness of the sanctuary and of the importance of the area to the species; conduct channel maintenance by methods which do not hinder the propagation of the species; and record a notation on the deed of the property comprising the sanctuary area that will in perpetuity notify transferees that the sanctuary must be maintained in accordance with the stated provisions (U.S. Fish and Wildlife Service 2010, p. 4).

4.4.5 Voluntary

Most of the land adjacent to the Pearl and Bogue Chitto Rivers in Louisiana and Mississippi is privately owned and much of it is managed for timber. Both states have developed voluntary BMPs for forestry activities conducted in their respective states. In addition, the forest industry has a number of forest certification programs, such as the Sustainable Forestry Initiative, which require participating landowners to meet or exceed state forestry BMPs. Silvicultural practices implemented with BMPs reduce negative impacts to aquatic species through reductions in nonpoint source pollution. Although nonpoint source pollution is a localized threat to the Pearl River map turtle, it is less prevalent in areas where certified BMPs are utilized. In Louisiana, BMPs include streamside management zones (SMZ) of 50ft (15.24m) measured from the top of the streambank, for streams of less than 20ft (6.1m) under estimated normal flow, to a width of 100ft (30.5m) for streams more than 20ft (6.1m) wide (LDAF undated, p. 15). BMP guidance includes maintaining adequate forest canopy cover for normal water and shade conditions as well as an appropriate amount of residual cover to minimize soil erosion (LDAF undated, p. 14). An overall rate of 97.4 percent of 204 forestry operations surveyed by the LDAF in 2018 complied with the state's voluntary guidelines; compliance with guidelines in SMZs was 98.6 percent (LDAF 2018, entire). The state of Mississippi has voluntary BMPs developed by the Mississippi Forestry Commission (MFC) (MFC 2008, entire). These BMPs include SMZs with the purpose of maintaining bank stability and enhancing wildlife habitat by leaving 50 percent crown cover during timber cuts (MFC 2008, p. 6). The width of SMZs is based on slope, with a minimum SMZ width of 30ft (9.14m) extending to 60ft (18.3m) at sites with over 40 percent slope (MFC 2008, p. 6). The most recent monitoring survey of 174 Mississippi forestry sites indicated that 95 percent of applicable sites were implemented in accordance with the 2008 guidelines (MFC 2019, p. 6).

CHAPTER 5 – CURRENT CONDITION

As the population is a biologically meaningful unit in an analysis of resilience, which is then scaled up to redundancy and representation at the species scale, appropriately defining and delineating populations is a crucial step to assess species viability. Below we discuss the challenges of delineating populations for the Pearl River map turtle and our approach. After delineating resilience units (i.e. populations), we then assessed the resilience of each unit as described in the following sections by synthesizing the best available information about observations and other important metrics thought to be important for viability of the Pearl River map turtle. Resilience of these units was used to assess current redundancy and representation for the species.

5.1 Delineating Resilience Units

Home range sizes and movements for Pearl River map turtle are not well known, so delineating biological populations is not feasible. Thus, we delineated what we term "resilience units" for the species to assess resilience. These units are not meant to represent "populations" in a biological sense; they may represent multiple or portions of groups of demographically-linked interbreeding individuals. As data are not available to delineate biological populations at this time, these units were intended to subdivide the species range in a way that facilitates assessing and reporting the variation in current and future resilience across the range.

Pearl River map turtle resilience units were delineated using HUC8 (8-digit hydrologic unit code) hydrologic units, taken from the USGS Water Boundary Dataset (USGS 2019a, unpaginated). HUC8 hydrologic units correspond to watersheds, with units denoted by fewer digits (e.g., HUC6) corresponding to larger units (basin), and those with more digits (e.g., HUC10 or HUC12) corresponding to smaller units or subwatersheds. Hydrologic units of smaller sizes (more digits) are nested within units of larger sizes (fewer digits). We used HUC 8 watersheds because that level best represented the distribution of the turtle in the mainstem and tributaries.

For the purpose of this SSA, we delineated five resilience units of Pearl River map turtles based on HUC8 watersheds and in accordance with guidance from species experts. These units are: Upper

Pearl, Middle Pearl-Silver, Middle Pearl-Strong, Bogue Chitto, and Lower Pearl (Figure 5.1). Historically, the majority of the range of the species was likely connected in a single interbreeding biological population, but we used the five analysis units in the SSA to most accurately describe trends in resiliency, forecast future resiliency, and capture differences in stressors among units.

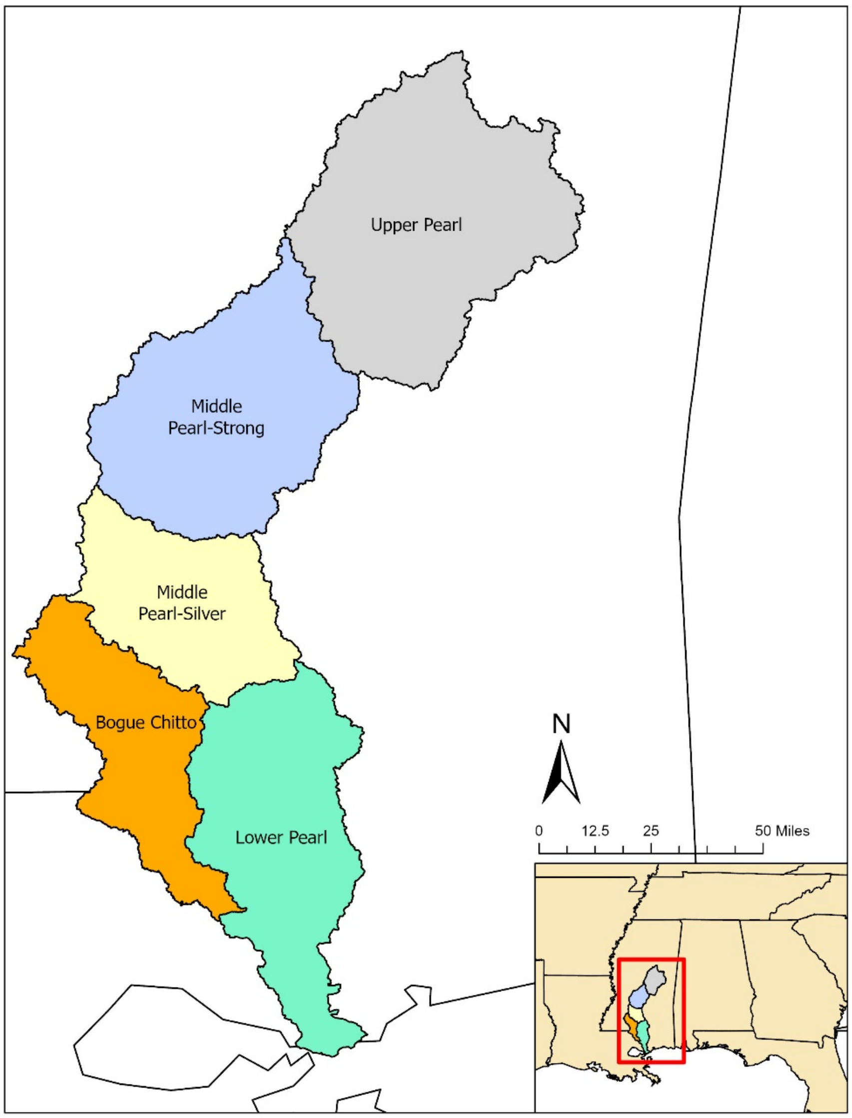

Figure 5.1-Pearl River map turtle resilience units delineated using HUC8 watersheds.

5.2 Population Resiliency

To assess current resiliency of Pearl River map turtle populations, we assessed the condition of two population factors within each analysis unit: 1) species presence using occupied tributaries and 2) density and abundance, and four habitat factors: 1) water quality, 2) forested riparian cover, 3) protected land, and 4) presence of channelization/reservoirs/gravel mining. These influences on population resiliency will hereafter be referred to as population and habitat factors, or collectively as resiliency factors.

5.2.1 Population Factor: *Presence using Occupied Tributaries*

For a given population to be resilient, the species must be present. Furthermore, although Pearl River map turtle relative abundance is typically much higher within mainstem reaches, presence of the species within tributary systems can contribute to resiliency by increasing the number of occupied miles of stream within a given unit, and also by providing refugia from catastrophic events, such as spill events or flooding. Insulation against threats in tributaries is important to the conservation status of the species, because the relatively tributary-poor Pearl River drainage (e.g. compared to the Pascagoula basin) has considerably more plans for projects that would degrade habitat for the species, including several proposed reservoirs (Lindeman 2013, pp. 202-203).

Forty-nine percent (49%) of the total range occupied by the Pearl River map turtle is in the mainstem Pearl and West Pearl rivers, with the remaining 51% of the occupied range found in various tributary systems (Lindeman 2019, p. 19). Tributary populations have been shown to be less densely populated compared to mainstem populations, although some tributaries (e.g. Bogue Chitto River) contain relatively large populations of Pearl River map turtles, including some that have only recently been discovered. For example, the species was first found in Yockanookany Creek in 1994, and extensive upstream range extensions have now been reported in several tributary systems, including Yockanookany Creek, Strong River, and Bogue Chitto River (Lindeman 1998, p. 139; Lindeman 2013, p. 298; Lindeman 2019, p. 50); these three tributaries together are inhabited by nearly one-third of the entire species, with nearly half occurring in stretches not known to be part of the species' range prior to the 21st century (Lindeman 2019, p. 23).

In order to assess occupied tributaries, we used survey data collected from 2005-2020. These data were collected by several different observers through a variety of survey types, including bridge surveys, basking surveys, and live trapping. Surveys were not repeated or standardized over time, thus we cannot assess trends in occupancy; rather, we consider a drainage occupied if it has been seen within the 2005-2020 time period. We used 2005 as the cutoff based on the species' biology and expert input. Females typically reach sexual maturity after eight years, so 15 years captures approximately two generations. Also, species experts noted that most surveys conducted for the species have occurred after 2005. When assessing occupancy of tributaries, we considered both occupied tributaries and tributaries that were surveyed, but no Pearl River map turtles were found (i.e. inferred absence; Figure 5.2). We established thresholds for occupied tributaries by applying the following rule set:

- Very Low: no currently occupied tributaries
- Low: between 1-24% of surveyed tributaries are currently occupied
- Moderate: between 25-49% of surveyed tributaries are currently occupied
- High: 50% or more of surveyed tributaries are currently occupied

Using this rule set, we found that one unit was determined to be ranked very low (Middle Pearl Silver); three ranked moderate (Upper Pearl, Bogue Chitto, and Lower Pearl); and one ranked high (Middle Pearl Strong) (Table 5.1). The Middle Pearl Silver unit has four surveyed tributaries, and there have been no detections within any of those tributaries, leading to the very low rank. In the Lower Pearl, although only 43% of surveyed tributaries were found to be occupied, this unit had by far, the most occupied tributaries (7), thus the moderate rank is likely more a function of survey effort. Half of the tributaries surveyed within the Middle Pearl Strong unit were found to be occupied, giving it a high rank.

Table 5.1-Assessment of tributaries found to be occupied for the Pearl River map turtle. Rank is based off of the % of occupied tributaries surveyed: very low (no occupied tributaries); low (1-24% occupied tributaries); moderate (25-49% occupied tributaries); high (50% or more occupied tributaries).

Unit	**# Surveyed Tributaries**	**# Occupied Tributaries**	**% Occupied Tributaries**	**Rank**
Upper Pearl	7	3	43%	Moderate
Middle Pearl Strong	6	3	50%	High
Middle Pearl Silver	4	0	0	Very Low
Bogue Chitto	5	2	40%	Moderate
Lower Pearl	15	7	43%	Moderate

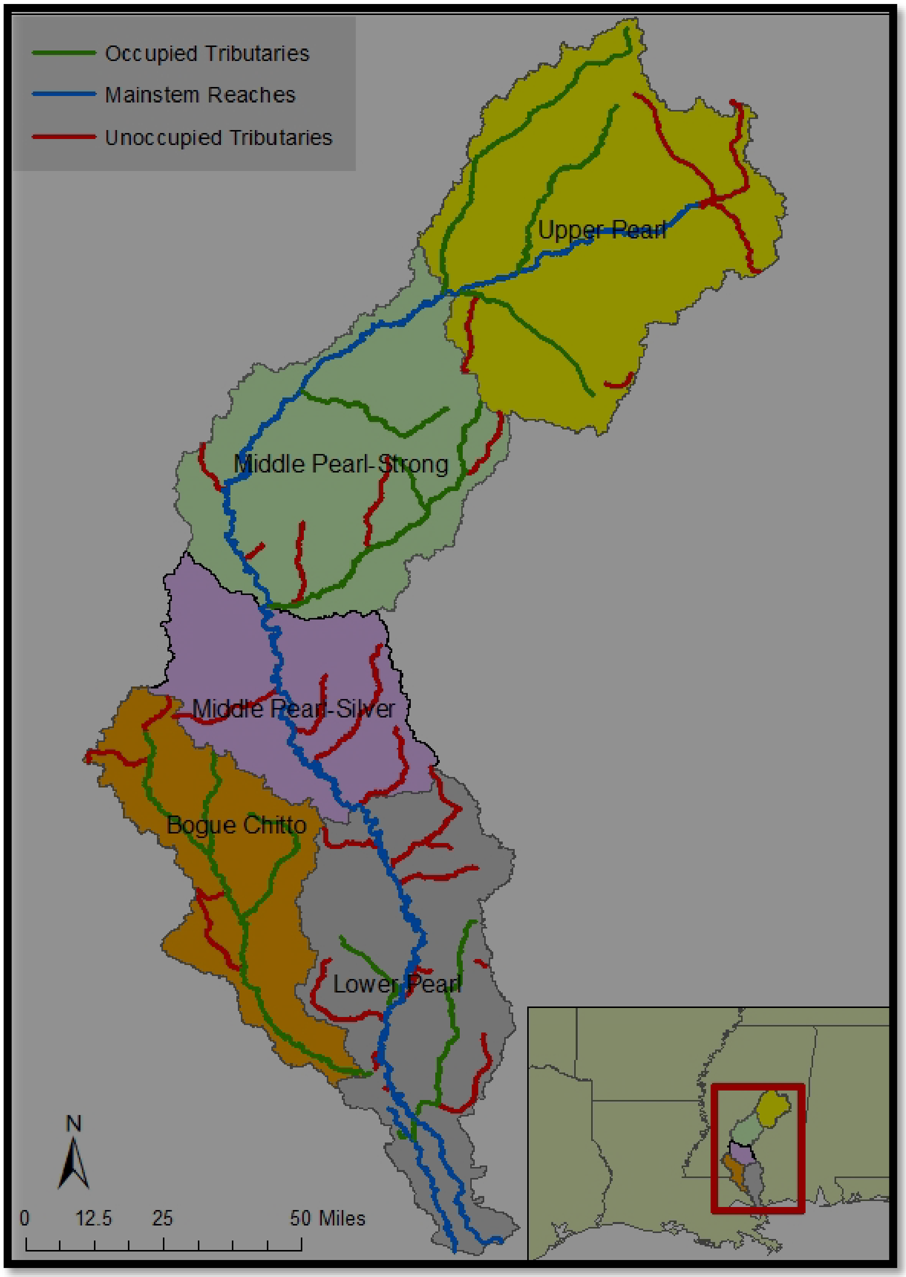

Figure 5.2-Surveyed streams across all Pearl River map turtle units. Occupied tributaries (green), unoccupied tributaries (red), and mainstem reaches (blue) shown.

*5.2.2 **Population Factor:*** *Density and Abundance*

The influence of stochastic variation in demographic (reproductive and mortality) rates is much higher for small populations than large ones. Stochastic variation in demographic rates causes small populations to fluctuate randomly in size. In general, the smaller the population, the greater the probability that fluctuations will lead to extinction. There are also genetic concerns with small populations, including reduced availability of compatible mates, genetic drift, and low genetic diversity or inbreeding depression. Small populations of Pearl River map turtles inherently have low resilience, leaving them particularly vulnerable to stochastic events.

Because we do not have historical surveys to use as a baseline to compare current abundance estimates, we cannot make inferences as to whether populations are decreasing, increasing, or stable through time. However, a recent study combined data from point counts, basking density surveys, and trapping from 2006–2018 to estimate density and abundance for stream segments throughout the range of the Pearl River map turtle (Lindeman 2019, p. 11-12). The global population was estimated to be 21,841 individuals, with 61% occurring on mainstem reaches, 34% occurring in four large tributaries, and the remaining 5% spread amongst other smaller tributaries (Lindeman, 2019 p. 21). Generally, abundance of the species declined with the size of the river reach surveyed, where tributaries generally had lower numbers of turtles compared to mainstem reaches (Lindeman 2019, p. 13). For example, basking density was found to be 2.2 times higher on mainstem reaches as on tributary reaches and 2.1 times higher on large tributaries than on small tributaries (Lindeman 2019, p. 15).

Based on basking density surveys and on results of point counts, each river drainage was divided into river reaches that were categorized as high, moderate, low, and very low density. Lindeman (2019, p. 20) assigned these categories based on typical basking densities of 10, 3, 1, and 0.7 individuals/rkm, respectively, and these categories comprised 3%, 68%, 16%, and 14%, respectively, of the total range of the Pearl River map turtle.

We used the density categories from Lindeman (2019, p. 47) to assess density of mainstem reaches and large tributaries as a population factor for this SSA. As discussed in the next section, we then combined the density score and occupied tributary score for each unit to come up with a composite population factor.

Table 5.2-Density of mainstem reaches and large tributaries within the analysis units (in bold) of Pearl River map turtles based on Lindeman (2019, p. 47). *a 19.5 rkm (12.1 rm) stretch was assessed as very low by Lindeman, however most of the occupied tributary was classified as moderate. **a 33.4 rkm (20.7 rm) stretch of mainstem reach was assessed as high by Lindeman, however the majority of the mainstem within this unit was classified as moderate.

Unit/Stream	Density Class
Bogue Chitto	**Moderate**
Bogue Chitto (large trib)	*Moderate/Very Low**
Lower Pearl	**Low**
East Pearl (large trib)	Low
Pearl River (mainstem)	*Moderate*
West Pearl (mainstem)	*Low*
Middle Pearl Silver	**Moderate**
Pearl River (mainstem)	*Moderate*
Middle Pearl Strong	**Moderate**
Pearl River (mainstem)	*Moderate*
Strong River (large trib)	*Moderate*
Upper Pearl	**Moderate**
Pearl River (mainstem)	*Moderate/High***
Yockanookany Creek	*Moderate*

5.2.3 ***Population Factor:*** *Composite Score*

To determine a composite score for population factors within individual units, we combined the results of the assessment of occupied tributaries (Table 5.1) and density classes of mainstream reaches and large tributaries (Table 5.2). Classifications were averaged together for each population factor as if very low, low, moderate, and high were equal to values of 1, 2, 3, and 4 respectively. If averaging the two factors resulted in a value ending in .5, the score was weighted more heavily towards the density factor. For example, if the occupied tributary's rank was assessed as low, and density was assessed as moderate, the composite score would be moderate due to heavier weighting given to density. Composite population factor scores were then categorized on a very low (1), low (2), moderate (3), and high (4) scale.

Table 5.3-Population factor composite score for each unit based on the combination of the occupied tributaries and density population factors. Density was weighted slightly higher in the assessment.

Unit	Occupied Tribs	Density	Composite Score
Bogue Chitto	Moderate	Moderate	Moderate
Lower Pearl	Moderate	Low	Low
Middle Pearl Silver	Very Low	Moderate	Low
Middle Pearl Strong	High	Moderate	Moderate
Upper Pearl	Moderate	Moderate	Moderate

5.2.4 Habitat Factor: *Water Quality*

As mentioned previously, water quality is an important component of Pearl River map turtle resilience because it affects how well they survive and reproduce. In the absence of site-specific water quality measurements taken at occurrence locations within each unit, we used data available at the resiliency unit scale from land cover data that were compiled from the 2016 National Land Cover Dataset (NLCD) Version 1, accessed via the Multi-Resolution Land Characteristics (MRLC) consortium online to characterize nonpoint source pollution (i.e., development and agriculture) (MRLC 2016, unpaginated).

Land use can be an indicator of overall watershed health and provide insight into water quality. Agricultural land use within riparian zones has been shown to directly impact biotic integrity when assessed within the intermediate-sized zones (i.e., 200ft (61m) buffer) surrounding the streams in the region (Diamond et al. 2002, p. 1150). Urbanization has also been shown to impair stream quality by impacting riparian health (Diamond et al. 2002, p. 1150). We assessed watershed health by combining several metrics within each resilience unit: percent urbanization and agriculture land use at the watershed level, as well as riparian effects, which included urbanization and agriculture within close proximity to the stream 200ft (61m)) buffer from the center of the waterbody). Many riparian BMPs stipulate maintaining a natural buffer of 100ft (30m) to protect water quality, thus the buffer chosen for our analysis captures the area adjacent to the stream that is believed to be most important to water quality (EPA 2005, p. 9).

5.2.4.1 Watershed Health

Increased agricultural land use within a watershed has the potential to increase nutrient and other pollutant-loading to stream systems. In addition to other impacts on aquatic habitat structure and quality, urban cover increases runoff volume into streams, likely increasing loads of sediments, nutrients, metals, pesticides, and other nonpoint source pollutants (CWP 2003, entire). Watershed health within populations boundaries and riparian buffers were calculated using urban and agricultural land use information. Land cover data were compiled from the 2016 NLCD land use land cover data set. We combined the low, medium, and high intensity development into a single developed land class. We combined hay/pasture and cultivated crops land use classes into a single agriculture land class. To calculate percentages of development and agriculture across the unit, we simply divided the corresponding land class by the total acreage of the unit. To calculate percentages of development and agriculture within riparian buffers, we divided the corresponding land class by the total acreage of the buffer around each occupied stream.

To establish current water quality levels within a unit, we created thresholds of low, moderate, and high threats to Pearl River map turtles. By creating these levels, we enable an assessment of the projected changes in the levels of these threats in future scenarios, as well as subsequent predictions about changes in resilience. The scaling of urban watershed impacts was derived from the Impervious Cover Model (ICM) and studies on amphibians and other taxa (Schueler 1994, entire) which is widely used in planning and zoning. An updated model includes ranges of impervious cover likely impacting stream quality (Schueler et al. 2009, p. 313) and indicates good stream quality is <5-10% impervious cover, fair quality (i.e., impacted) ranges from 5-25% impervious cover, and poor quality occurs at >20-25% impervious cover within the watershed. Several other studies have found impacts of urbanization on biotic health occur at 8-12% impervious cover (Wang et al. 2001 p. 259), although results from a recent study in the Etowah (Wenger et al. 2008, pp.1260-1261) indicate some species could become rare at impervious cover as low as 2%.

5.2.4.2 Riparian Health

Riparian impairment, either through urbanization or agriculture use, can amplify negative effects of nonpoint source pollution within the watershed as well as impact stream quality independent of land use within the watershed. Impacts from impervious cover can be mitigated through riparian forest cover and good riparian health (Roy et al 2005, p. 2318; Walsh et al. 2005, entire); however, several studies have indicated benefit of the riparian cover diminishes when impervious cover (i.e. urban cover) exceeds ~10% within the watershed (Booth and Jackson 1997, p. 1084; Goetz *et al*. 2003, p. 205). Diamond et al. (2002, entire) assessed the relationship between human land uses (urban and agriculture) and fauna in the Clinch and Powell River watersheds in Tennessee and Virginia. They found that when urban areas and major highways approached 12.2% cover within 200ft (61m) of the stream, the stream was more likely to be classified as impaired within the Clinch River, Powell River, and Copper Creek while unimpaired sections of those streams averaged 5.6% urban cover (Diamond et al. 2002, p. 1151). We calculated percent cover of urban land use within 200ft (61m) of each stream in each population and classified percentages to a low(<6%)-moderate(6-12%)-high(>12%) threat scale (Table 5.4).

Like the effects of urban use in riparian zones, agricultural impacts can directly decrease riparian vegetation cover and health. Agricultural practices within the riparian zones can further impact water quality and aquatic organisms via increased exposure to chemical fertilizers, pesticides, livestock waste, and sedimentation which has been implicated in amphibian malformation, susceptibility to disease, and declines in population numbers, reproductive success, and biodiversity (Beja and Alcazar 2003, entire; Montag *et al*. 2019, entire; Burkholder *et al*. 2007, pp. 309-310). There is little information regarding the threshold for agriculture land use within a riparian area that will begin to have an impact on stream water quality. Therefore, we used the thresholds for urban land use to inform thresholds for agricultural land cover. However, because the relationship between area of agricultural land and water quality is less certain than the relationship between urban area and water quality, we reduced the number of classifications used to assess agricultural land use threats (Table 5.5). A threshold of 10%, rather than the 5% threshold used for urban development, to distinguish between low and moderate levels of threats is reasonable because it is in line with suggested values from the literature (i.e. 8-12% threshold; Wang *et al*. 2001 p. 259;

Schueler et al. 2009, p. 313), and agriculture is typically not associated with high amounts of impervious cover, thus % agriculture of <10% is unlikely to significantly impact infiltration capacity, and thus water quality.

Table 5.4. Metrics used to categorize impacts of urbanization within units.

		% Urban in unit			
		0-5%	5 10%	10 20%	>20%
Urban Cover in Riparian Areas	Low (0-6%)	Low	Low	Moderate	High
	Moderate (6-12%)	Low	Moderate	Moderate	High
	High (>12%)	High	High	High	High

Table 5.5. Metrics used to categorize impacts of agriculture within units.

		% Agriculture in unit		
		0-10%	10 20%	>20%
Ag Cover in Riparian Areas	Low (0-10%)	Low	Moderate	High
	Moderate (10-20%)	Low	Moderate	High
	High (>20%)	High	High	High

5.2.4.3 Land Use Composite Score

In our analysis, overall watershed health within a population is considered to be influenced by a combination of direct impacts by urbanization and agriculture. To generate a single composite score for watershed health for each unit, all agriculture and urban composite water quality scores were combined. Classifications were averaged together for each composite watershed score as if low, moderate, and high threats were equal to values of 1, 2, and 3, respectively. If averaging the two factors resulted in a value ending in .5, the overall water quality score was rounded down (rather than typical mathematical convention of rounding up) to be conservative (i.e. to avoid underestimating threats derived from land use). Composite population land use scores were then categorized on a low (1)-moderate (2)-high (3) threat scale.

5.2.4.4 Results

Results for land use across the watersheds and within riparian areas, along with their overall composite classification scores, are summarized in Appendix A and Tables 5.6-5.10. Land use composite scores for all 5 units were moderate. In fact, the only stream that was assessed as having a relatively high degree of threat based on land use, was the Lower Pearl, driven primarily by a high degree of development within the riparian buffer (33%). In general, development is low throughout the Pearl River basin, although there is significant development across the Middle Pearl Strong Unit (12% development), where the city of Jackson is located. Agriculture is generally high across the Pearl River basin, where levels of agriculture within the units ranged from 12-23%, with the Bogue Chitto Unit having the highest levels of agriculture.

Table 5.6-Composite land use score for streams within the Bogue Chitto unit, and for the entire unit, based on development and agriculture levels. Streams in bold are mainstem reaches; all other streams are considered tributaries.

Stream	**Developed**	**Agriculture**	**Composite Land Use Score**
Bogue Chitto	Low	High	Moderate
Magees	Low	High	Moderate
Topisaw	Low	High	Moderate
Bogue Chitto Unit			**Moderate**

Table 5.7-Composite land use score for streams within the Lower Pearl unit, and for the entire unit, based on development and agriculture levels. Streams in bold are mainstem reaches; all other streams are considered tributaries.

Stream	Developed	Agriculture	Composite Land Use Score
East Pearl	Low	Moderate	Moderate
Hobolochitto	Low	Moderate	Moderate
Holmes Bayou	Low	Moderate	Moderate
Lower Pearl	High	Moderate	Low
Navigation Canal	Low	Moderate	Moderate
Pushepatapa	Low	Moderate	Moderate
West Hobolochitto	Low	Moderate	Moderate
West Pearl	Low	Moderate	Moderate
Lower Pearl Unit			**Moderate**

Table 5.8- Composite land use score for streams within the Middle Pearl Silver unit, and for the entire unit, based on development and agriculture levels. Streams in bold are mainstem reaches; all other streams are considered tributaries.

Stream	Developed	Agriculture	Composite Land Use Score
Middle Pearl Silver	**Low**	**Moderate**	**Moderate**

Table 5.9-Composite land use score for streams within the Middle Pearl Strong unit, and for the entire unit, based on development and agriculture levels. Streams in bold are mainstem reaches; all other streams are considered tributaries.

Stream	Developed	Agriculture	Composite Land Use Score
Middle Pearl Strong	Moderate	Moderate	Moderate
Pelahatchie	Moderate	Moderate	Moderate
Purvis	Moderate	Moderate	Moderate
Strong	Moderate	Moderate	Moderate
Middle Pearl Strong Unit			**Moderate**

Table 5.10-Composite land use score for streams within the Upper Pearl unit, and for the entire unit, based on development and agriculture levels. Streams in bold are mainstem reaches; all other streams are considered tributaries.

Stream	Developed	Agriculture	Composite Land Use Score
Lobutcha	Low	Moderate	Moderate
Tuscolometa	Low	Moderate	Moderate
Upper Pearl	Low	Moderate	Moderate
Yockanookany	Low	Moderate	Moderate
Upper Pearl Unit			**Moderate**

5.2.5 Habitat Factor: Channelization/Reservoirs/Gravel Mining

Channel modification is recognized as a cause of decline in the similar and sympatric species, the ringed map turtle (*G. oculifera*), which is federally threatened (Lindeman 1998, p. 137). Stream channelization, point-bar mining, and impoundment have been listed as potential threats in a report from before the Pascagoula map turtle and Pearl River map turtle were taxonomically separated (USFWS 2006, p. 2). Gravel mining has been identified as perhaps the greatest threat to the Pearl River system in southeastern Louisiana (Selman 2020c, p. 20). Gravel mining can degrade water quality, increase erosion, and ultimately impact movement and habitat quality for aquatic species such as the Pearl River map turtle (Koehnken et al. 2020, p. 363). We assume that substantial channelization, the presence of a major reservoir, or evidence of gravel mining operations has a negative impact on resiliency, and include these as a resilience factor. Below we describe several areas where alteration of streams has likely led to declines in Pearl River map turtles, and assess the condition category for each unit (Table 5.11).

Considerably low densities of Pearl River map turtles were observed in the Lower Pearl unit, where much channelization and flow diversion has occurred (Lindeman 2019, p. 23-29). Low densities of Pearl River map turtles in the West and East Pearl rivers, have been attributed to flow alteration due to the construction of the Pearl River Navigation Canal, which also has very low densities, suggesting that substantial loss of population in the lower reaches of the Pearl River drainage has occurred historically due to river engineering (Lindeman 2019, p. 27). Significantly lower basking densities of Pearl River map turtles have been reported in the West Pearl (0.1/rkm) compared to the upper Pearl (1.8/rkm) (Dickerson and Reine 1996, Table 4, unpaginated; Selman 2020c, pp. 17-18). Because of these stream alterations, we assessed the Lower Pearl unit as low (i.e. high degree of threats) for this factor.

Within the Middle Pearl Strong unit, 33.6 rkm (20.9 rm) of the middle Pearl River is inundated by the Ross Barnett Reservoir. The Ross Barnett Reservoir has greatly reduced habitat suitability of 5% of the mainstem Pearl River by altering the lotic (flowing water) habitat preferred by Pearl River map turtles to lentic (lake) habitat. In addition, declines in population densities have been observed upstream and downstream of the Ross Barnett Reservoir. Declines in population densities upstream of the reservoir are possibly due to recreational boating and extended recreational foot traffic or camping on sandbars by reservoir visitors (Selman and Jones 2017, p. 32-34). Between the late 1980s and early 2010s, notable population declines have been observed in the stretch of the Pearl River downstream of the Ross Barnett Reservoir (Selman 2020d, p. 194). It is unknown why the population declined, but altered hydrology of this reach may have had an impact. Plans for new reservoirs on the Pearl River both upstream and downstream of Jackson have been or are being considered (Lindeman 2013, pp. 202-203). Researchers have estimated that up to 170 individual Pearl River map turtles could be directly impacted, and up to 360 indirectly impacted, both upstream and downstream, by the One Lake Project, one of several proposed impoundments (Selman 2020d, p. 192). Near Jackson, Mississippi, river channelization has also impacted the species habitat negatively, and Pearl River map turtles are almost non-existent in a highly channelized stretch of the Pearl River (Selman 2020d, p. 194). However, upstream and downstream of this section, the species occurs in low numbers (Selman 2020d, pp. 192-194). Due to the presence of the Ross Barnett Reservoir, and the river channelization that has occurred in and around Jackson, we assessed the Middle Pearl Strong unit as low for this factor.

In the Upper Pearl unit, channelization has occurred along Tuscolameta Creek and the upper Yockanookany River. In 1924, the Tuscolameta Creek received a 39 km (24-mi) channelization and Yockanookany River received a 58 km (36-mi) canal, which was completed in 1928 (Speer et al. 1964, p. 8). In the Yockanookany, low water stages in 1960 were six feet (1.82m) higher than those of 1939, as the channel silted significantly during that period (Speer et al. 1964, pp. 26-27). In some areas of the Yockanookany, water continues to flow in the river’s old natural channel

(Speer et al. 1964, pp. 26-27). Although stream alteration has occurred within these streams, there has yet to be any reported evidence of Pearl River map turtle decline, thus we assessed this factor as moderate for the Upper Pearl unit.

In-stream and unpermitted point bar mining in the Bogue Chitto unit was a concern in the late 1990s (Shively 1999), and although these activities no longer occur, gravel mining operations within floodplains does occur (Selman 2020c, pp. 20-21). Recent surveys have reported several areas where mining appears to have degraded water quality significantly (Selman 2020c, pp. 20-21). There is also a concern that historical in-stream and point bar mining can have deleterious legacy effects that could be negatively impacting the species (Selman 2020c, p. 21). For these reasons, we assessed this factor as low for the Bogue Chitto unit.

Table 5.11-Categorization of habitat factor "Channelization/Reservoirs" for Pearl River map turtle resilience units.

Population	**Channelization/Reservoir/Mining (Y/N)**	**Condition Category**
Bogue Chitto	Yes	Low
Lower Pearl	Yes	Low
Middle Pearl Silver	No	High
Middle Pearl Strong	Yes	Low
Upper Pearl	Yes	Moderate

5.2.6 Habitat Factor: *Protected Land*

A protected area is a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (IUCN 2008, unpaginated). Protected areas are a generally accepted, although not always uncontroversial, mechanism for halting the global decline of biodiversity. Some examples of the positive effects that protected areas can have on freshwater biodiversity have been reported, such as increased local abundance or size classes of some fish species (Suski and Cooke, 2007, entire).

From an indirect standpoint, the presence of protected lands will function to minimize human disturbance in an area, which may benefit freshwater environments at multiple levels. First, enforcement of restrictions in protected areas can serve to minimize boat traffic that has been shown to have deleterious impacts to other *Graptemys* species (Selman et al. 2013, entire; Heppard and Buchholz 2019; entire). Also, conflict between various users of freshwater resources has been documented (Jones 2006, p. 208), and will likely continue to escalate as human demands on freshwater resources continue to increase. The presence of protected areas may help ameliorate some of these conflicts by segregating user groups into defined areas (Suski and Cooke 2007, p. 2024). Finally, the more land within a unit that is under some sort of protection (e.g. easement, state and federal ownership), the less likely land will be developed. Because development can have negative impacts to aquatic fauna, as discussed previously, the more protected land in a unit, the more resilient that unit is assumed to be.

Numerous refuges and conservation areas have been established along the Pearl River that have positively influenced riparian forest along the river or forest land cover in the basin. Riparian conservation areas include Nanih Waiya WMA (Neshoba County), Mississippi Band of Choctaw Indian Reservation (Neshoba County), Pearl River WMA (Madison County), Fannye Cook Natural Area (Rankin County), Old River WMA (Pearl River County), Bogue Chitto NWR (St. Tammany and Washington Parishes), and Pearl River WMA (St. Tammany Parish). Bienville National Forest contributes positively to increased forest cover in headwater streams that drain into the Pearl River, especially the Strong River. The most extensive habitat preservation on the Pearl River is the Bogue Chitto National Wildlife Refuge along the upper West and East Pearl and lower Bogue Chitto Rivers, which is contiguous with the Pearl River WMA, which protects the area between the West and East Pearl Rivers downstream to the Gulf of Mexico.

To assess the contribution of protected areas to the resilience of Pearl River map turtle analysis units, we calculated the percentage of the HUC8 that was in protected status. We used the Protected Areas Database of the U.S. version 2.0 (PAD-US 2.0), released in 2019 (USGS 2019b, unpaginated). The database is a national inventory of lands held by cities, counties, special park and open-space districts, state parks and preserves as well as federally administered lands, including national parks and forests, national wildlife refuges, public lands and more. We calculated the total area of land within each unit that was considered protected in the PAD-US 2.0, and divided by the total area of the HUC8. We then created categories of low, moderate, and high based off of the percentage of protected land within a unit as follows: low (0-10% protected), moderate (10-20% protected), and high (>20% protected). These categories were based off of natural breaks in the data from both the Pearl and Pascagoula basins, and serve as a relative measure of protected lands between all of the units; there are no data that we are aware of that suggests biologically significant breaks between the low, moderate, and high classes.

The results of the analysis of protected lands show that the Pearl River basin in general has relatively small amounts of land in protected status (Table 5.12). Four of the units have a low condition (i.e. <10% of land protected), and one unit has a moderate condition (10-20% of land protected). The Middle Pearl Strong Unit has by far the greatest amount of land in protection with 147,597 acres in protection (11.67%), with all other units having less than 6% of land in protected status.

Table 5.12-Summary of the resilience unit area, percentage of protected lands, along with the condition category, for each Pearl River map turtle resilience unit.

Resilience Unit	Total Acres/Hectares	% areas protected	Condition
Bogue Chitto	773546 ac/ 313042 ha	0	Low
Lower Pearl	1165616 ac/ 471708 ha	4.44%	Low
Middle Pearl Silver	779923 ac/ 315623 ha	0.16%	Low
Middle Pearl Strong	1265209 ac/ 512011 ha	11.67%	Moderate
Upper Pearl	1576500 ac/ 637986 ha	5.36%	Low

5.2.7 Habitat Factor: *Forested Riparian Cover*

Correlations of Pearl River map turtle density with deadwood density have been shown to be positive, and high basking densities have yet to be associated with low deadwood densities (Lindeman 1999, pp. 35-38). Abundance of basking substrates has shown to be an important habitat component driving *Graptemys* abundance in Kansas and Pennsylvania (Pluto and Bellis 1986, pp.

26-30; Fuselier and Edds 1994, entire), and radiotelemetry work with yellow-blotched map turtles (*G. flavimaculata*) has indicated the importance of deadwood to habitat selection on the lower Pascagoula River (Jones 1996, pp. 383, 376, 379-380). Anthropogenic deadwood removal, mainly through dredging, has been noted as a reason for decline in the sympatric microcephalic species, the ringed map turtle (*G. oculifera*) (Lindeman 1998, p. 137). Experiments with manual deposition of deadwood in stretches with less riparian forest have been recommended as potential habitat restoration measures (Lindeman 2019, p. 33).

An intact riparian habitat provides numerous benefits to map turtles including the stabilization of stream banks, and the reduction of erosional processes and channel sedimentation. Under normal erosional processes, riparian forests also provide material for in-stream deposition of deadwood, and deadwood is known to be important basking sites for thermoregulation and also foraging sites for prey items (Lindeman 1999, entire). To assess the contribution of riparian forests to the resilience of Pearl River map turtle units, we calculated the percentage of forest within a 200ft (61m) riparian buffer using the 2016 NLCD land use land cover data. We further describe forests to include four land use classes: deciduous forest, evergreen forest, mixed forest, and woody wetlands. We categorized the percent forested cover into three classes: <60% forested cover (low), 60-80% forested cover (moderate), and >80% forested cover (high). These classes were based on natural breaks in the data, and serve as relative comparisons of forested cover between the resilience units.

An assessment of forested cover (Table 5.13 A-E) resulted in 3 units in high condition (Lower Pearl, Middle Pearl Strong, and Upper Pearl), and 2 units in moderate condition (Bogue Chitto and Middle Pearl Silver). Forested cover within riparian buffers ranged from 60-98% across the 5 resilience units. Forested cover was highest in the Upper Pearl, where cover ranged from 90-96% across the occupied streams within the unit, and lowest in the Middle Pearl Silver, where forested cover was 60% across the single occupied river segment. The Bogue Chitto unit was assessed as moderate for forested cover, primarily due to the Bogue Chitto and Topisaw having relatively low cover compared to other streams across the range.

Table 5.13-Summary of forested cover with riparian areas and the associated condition class for each of the 5 resilience units for the Pearl River map turtle. A) Bogue Chitto B) Lower Pearl C) Lower Pearl Silver D) Lower Pearl Strong and E) Upper Pearl.

A) Bogue Chitto Unit

Stream	Total Acres Riparian	% Forested Riparian	Condition
Bogue Chitto	4962.29 ac/ 2008.17 ha	72.37	Moderate
Magees	1821.63 ac/737.19 ha	89.83	High
Topisaw	1305.68 ac/528.39 ha	75.68	Moderate
TOTAL	**8089.60 ac/3273.74 ha**	**76.83**	**Moderate**

B) Lower Pearl Unit

Stream	Total Acres Riparian	% Forested Riparian	Condition
East Pearl	860.22 ac/348.12 ha	85.37	High
Hobolochitto	291.78 ac/118.08 ha	98.32	High
Holmes Bayou	134.99 ac/54.63 ha	95.06	High
Lower Pearl	2539.97 ac/1027.89 ha	61.21	Moderate
Navigation Canal	28.91 ac/11.70 ha	84.62	High
Pushepatapa	1210.72 ac/489.96 ac	96.62	High
West Hobolochitto	2452.79 ac/992.61 ha	97.33	High
West Pearl	526.19 ac/212.94 ha	60.36	Moderate
TOTAL	**8045.57 ac/3255.93 ha**	**82.05**	**High**

C) Middle Pearl Silver Unit

Stream	Total Acres Riparian	% Forested Riparian	Condition
Middle Pearl Silver	**1844.54 ac/746.46 ha**	**60.42**	**Moderate**

D) Middle Pearl Strong Unit

Stream	Total Acres Riparian	% Forested Riparian	Condition
Middle Pearl Strong	2606.91 ac/1054.98 ha	70.99	Low
Pelahatchie	1876.34 ac/759.33 ha	91.10	Moderate
Purvis	825.31 ac/333.99 ha	82.73	Low
Strong	4237.06 ac/1714.68 ha	92.39	Moderate
TOTAL	**9545.62 ac/ 3862.98 ha**	**85.46**	**High**

E) Upper Pearl Unit

Stream	Total Acres Riparian	% Forested Riparian	Condition
Lobutcha	2917.59 ac/ 1180.71 ha	95.54	High
Tuscolometa	1604.35 ac/ 649.26 ha	94.01	High
Upper Pearl	3363.72 ac/1361.25 ha	89.82	High
Yockanookany	3650.38 ac/1477.26 ha	92.17	High
TOTAL	**11536.05 ac/4668.47 ha**	**92.59**	High

5.2.8 Habitat Factor Composite Score

To determine a composite score for habitat factors, we combined the results of the water quality, channelization/reservoirs, protected lands, and deadwood abundance assessments. Classifications were averaged together for each habitat factor as if low, moderate, and high were equal to values of 1, 2, and 3 respectively. If averaging the four factors resulted in a value ending in .5, the overall habitat factor composite score was rounded down (rather than typical mathematical convention of rounding up) to be conservative (i.e. to avoid underestimating threats derived from our habitat factor surrogates). Composite habitat scores were then categorized on a low (1), moderate (2), and high (3) scale.

Table 5.14-Habitat factor composite scores for all Pearl River map turtle units as a function of 4 habitat factors (water quality, channelization/reservoirs, protected land, and deadwood abundance).

Unit	**Water Quality**	**Channelization/ reservoirs**	**Protected Land**	**Deadwood**	**Composite Score**
Bogue Chitto	Moderate	Low	Low	Moderate	Low
Lower Pearl	Moderate	Low	Low	High	Low
Middle Pearl Silver	Moderate	High	Low	Moderate	Moderate
Middle Pearl Strong	Moderate	Low	Moderate	High	Moderate
Upper Pearl	Moderate	Moderate	Low	High	Moderate

5.2.9 Current Resilience

We assessed each unit's current resilience as a function of both population and habitat factors. To do this, we determined a composite resilience score by averaging the composite population factor and composite habitat factor for each unit as if low, moderate, and high were equal to values of 1, 2, and 3, respectively. If averaging resulted in a composite resilience score ending in .5, the score was weighted more heavily towards the population factor, as presence and density of turtles is a more important metric associated with resilience of populations. For example, if the composite habitat factor for a given unit was assessed as low, and the composite population score was assessed as moderate, the composite resilience score would be moderate due to heavier weighting being given to the population factor. Composite resilience scores were then categorized on a low (1), moderate (2), and high (3) scale.

Current resilience results are as follows: two populations have low resilience (Bogue Chitto and Lower Pearl) and three populations have moderate resilience (Middle Pearl Silver, Middle Pearl Strong, and Upper Pearl) (Table 5.15). The Lower Pearl seems particularly vulnerable, as both the population and habitat composite scores were low. The Lower Pearl has significant channelization issues, low amounts of protected land, and a low density of individual turtles, all of which are driving the low resilience of this unit. Although the Middle Pearl Silver unit scored moderate for composite habitat score, the low composite population score (mainly a function of there being no occupied tributaries) is what is driving the low resilience of this unit. When looking at the three units with moderate resilience, the Middle Pearl Strong and Bogue Chitto units appear to be vulnerable to further decreases in resiliency. For the Bogue Chitto unit, low amounts of protected land, and substantial mining activity make this unit vulnerable. For the Middle Pearl Strong, development in the Jackson area and the presence of the Ross Barnett reservoir, make this unit vulnerable. If development increases substantially in this unit, or if proposed reservoir projects move forward, it is likely there would be population level impacts that would drop the resilience to low.

Table 5.15-Current resiliency of Pearl River map turtle units based off of composite habitat and population factors.

Unit	Composite Habitat Score	Composite Population Score	Current Resilience
Bogue Chitto	Low	Moderate	Moderate
Lower Pearl	Low	Low	Low
Middle Pearl Silver	Moderate	Low	Low
Middle Pearl Strong	Moderate	Moderate	Moderate
Upper Pearl	Moderate	Moderate	Moderate

5.2.10 Current Representation

Representation refers to the breadth of genetic and environmental diversity within and among populations, which influences the ability of a species to adapt to changing environmental conditions over time. Differences in life history traits, habitat features, and/or genetics across a species range often aid in the delineation of representative units, which are used to assess species representation.

Between 2005 and 2018, researchers genotyped 124 Pearl River map turtles from 15 sites across the Pearl River basin (Pearson et al. 2020, pp. 6-7). No distinct genetic structure was found throughout the Pearl River system. A single genetic population was recovered, and there was no evidence of isolation by distance (Pearson et al. 2020, pp. 11-12). For this reason, we consider the entire range of the Pearl River map turtle to be a single representative unit.

It has been suggested that the Strong River, located in the Pearl River Strong unit, may have some unique habitat features that could confer unique adaptative pressure (Lindeman pers. comm. 2020, p. 4). Perhaps most notably, the Strong River has some very rocky stretches that are unlike anything else in the drainage and could conceivably have a population with a distinct diet or life history, though no studies to date have addressed this question (Lindeman pers. comm. 2020, p. 4). Although we do not consider the Strong River to be a separate representative unit, we consider the Strong River to be a potentially significant stream from a habitat diversity perspective.

The Strong River is a large tributary and occupies an estimated 87.4 rkm (54.3 rm) range, with an estimated 1,749 individuals, accounting for 8% of the global population estimate (Lindeman 2019, p. 47). Lindeman (2019, p. 47) assessed density as moderate, which is relatively high for a tributary. The Strong River does not appear to have a high degree of threats, habitat factors are relatively high, and population factors appear stable. For this reason, the Strong River is currently contributing to the overall representation of the species.

5.2.11 Current Redundancy

Redundancy refers to the ability of a species to withstand catastrophic events and is measured by the amount and distribution of resilient populations across the species range. Catastrophic events that could severely impact or extirpate entire Pearl River map turtle units include chemical spills, changes in upstream land use that alters stream characteristics and water quality downstream, dam construction with a reservoir drowning lotic river habitat, and potential effects of climate change such as rising temperatures and sea level rise. The Middle Pearl Silver unit is perhaps the most vulnerable to a catastrophic spill, as there are no known occupied tributaries at this time. Extant units of the species are distributed relatively widely, and several of those units are classified as moderate or better resilience, thus it is highly unlikely that a catastrophic event would impact the entire species' range. Because of all of this, the Pearl River map turtle exhibits a moderate-high degree of redundancy, and that level of redundancy has stayed relatively stable over time.

5.2.12 Current Conditions Summary

We assessed current resilience as a function of two population factors (occupied tributaries and density) and four habitat factors (water quality, protected areas, deadwood abundance, and reservoirs/channelization). Based on these factors, there are two populations with low resilience (Lower Pearl and Middle Pearl-Silver) and three populations with moderate resilience (Upper Pearl, Middle Pearl Strong, and Bogue Chitto; Table 5.15; Figure 5.3); no units were assessed as highly resilient. Because three of the five units are classified as moderate resilience, and those units are distributed relatively widely, the Pearl River map turtle exhibits a moderate-high degree of redundancy (i.e. it is unlikely that a catastrophic event would impact the entire range of the species). We did not assess multiple representative units for the species due to recent research indicating no significant genetic structuring. Species experts indicated the Strong River might be unique from a habitat perspective (i.e. distinctly rocky stretches), so it is notable that the Strong River does not appear to have a high degree of threats, habitat factors are relatively high, and population factors appear stable. If the Strong River is indeed conferring different adaptive pressures due to the unique habitat type found there, the Strong River does appear to be currently contributing to representation for the species.

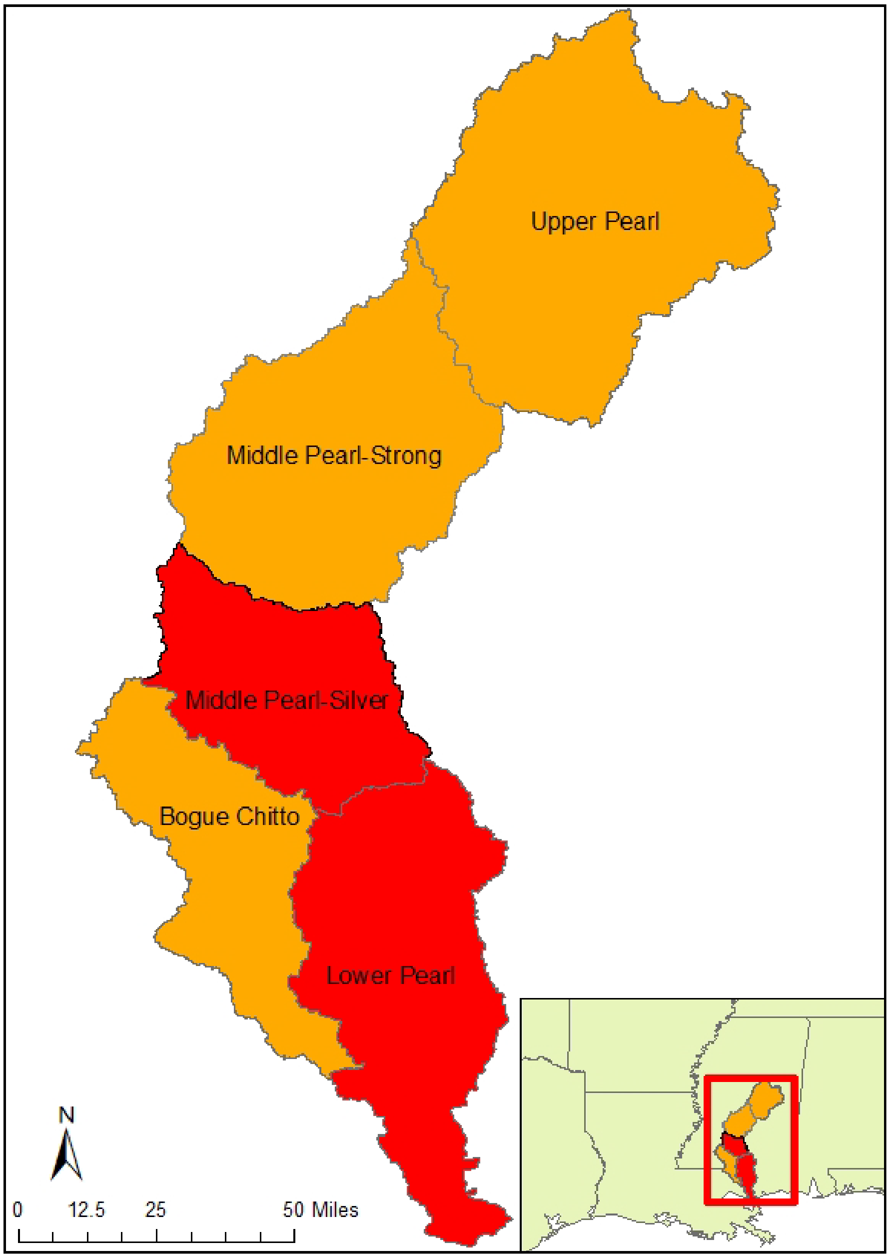

Figure 5.3-Resilience of the five units of Pearl River map turtles: low (red), moderate (orange), high (green).

CHAPTER 6 – FUTURE CONDITIONS AND VIABILITY

We have considered what the Pearl River map turtle needs for viability and the current condition of those needs (Chapters 3 and 5), and we reviewed the factors that are driving the current and future conditions of the species (Chapter 4). We now consider what the species' future condition is likely to be. We apply our future forecasts to the concepts of resiliency, representation, and redundancy to describe the future viability of the Pearl River map turtle.

6.1 Introduction

To assess future condition of Pearl River map turtle units, we projected the primary current threats of land use (agriculture and development), potential future water engineering projects, and sea level rise into the future under six plausible scenarios. The six scenarios capture the range of uncertainty in the changing human population footprint on the landscape, current emission models, implementation of water engineering projects, and how the Pearl River map turtle will respond to these changing conditions.

All six scenarios were projected out to two different time steps: 2040 (20 years) and 2070 (50 years). These time frames are based on input from species experts, generation time for the species, and the fact that beyond 50 years, the ability to predict patterns of urbanization and agriculture, and how these land uses will interact with the species and its habitat diminishes.

6.2 Future Resilience Factors

6.2.1 Land Use and Water Quality

We considered projected land-use changes in regards to agricultural and developed land in assessing future resilience of each unit for the Pearl River map turtle. We consider these land use classes as surrogates for potential changes in water quality, a primary risk factor for the species. We used data available at the resiliency unit scale from the USGS Forecasting Scenarios of Land-use

Change (FORE-SCE) modelling framework (USGS 2017, unpaginated) to characterize nonpoint source pollution (i.e., development and agriculture). The FORE-SCE model provides spatially explicit historic, current, and future projections of land use and land cover. The projections were originally created as part of the "LandCarbon" project, an effort to understand biological carbon sequestration potential in the United States. However, the projections are being used for a wide variety of purposes, including analyses of the effects of landscape change on biodiversity, water quality, and regional weather and climate. The 1992 to 2005 period is considered the historical baseline, with datasets such as the National Land Cover Database, USGS Land Cover Trends, and U.S. Department of Agriculture's Census of Agriculture used to guide the recreation of historical land cover. The future projection time frame is available at annual time steps up to the year 2100. Four scenarios were modeled, corresponding to four major scenario storylines from the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES). The global IPCC SRES (A1B, A2, B1, and B2 scenarios) were downscaled to ecoregions in the conterminous United States, with the USGS Forecasting Scenarios of land use (FORE-SCE) model used to produce landscape projections consistent with the IPCC SRES. The land-use scenarios focused on socioeconomic impacts on anthropogenic land use (demographics, energy use, agricultural economics, and other socioeconomic considerations; Figure 6.1.

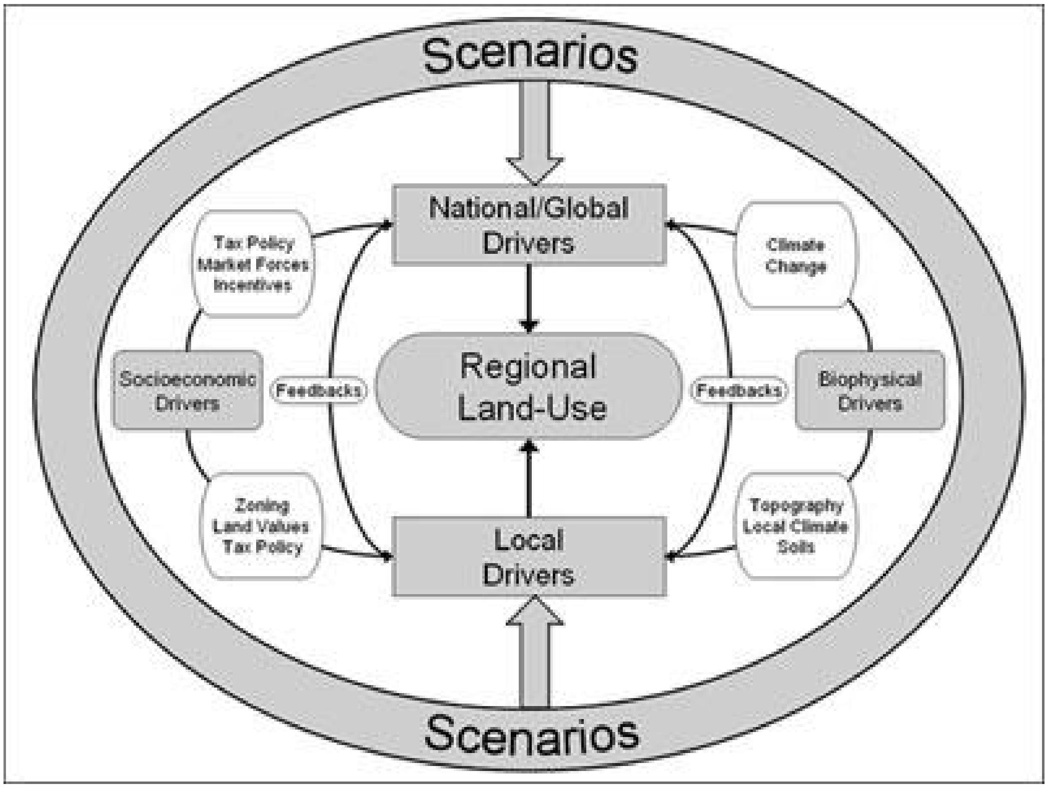

Figure 6.1- Forecasting scenarios of land-use change modeling framework to provide spatially explicit projections of future land-use and land-cover change (USGS 2017, unpaginated).

6.2.2 *Sea Level Rise*

Sea level rise (SLR) impacts future resilience of Pearl River map turtles by influencing the area occupied and suitable habitat available through increased salinity of the freshwater system. Although some species of *Graptemys* appear to handle some salinity increases, there is some evidence that the group is largely intolerant of brackish and saltwater environments (Selman and Qualls 2008 pp. 228-229; Lindeman 2013, pp. 396-397). To estimate loss/degradation of habitat due to inundation from sea level rise, we used NOAA's shapefiles available at their online sea level rise viewer (NOAA 2020, unpaginated). Projected sea level rise scenarios from NOAA provide a range of inundation levels from low to extreme. We chose NOAA's intermediate-high and extreme scenarios, which correspond to the representative concentration pathways (RCP) of RCP6 and RCP8.5 emission scenarios, the most likely RCP scenarios (IPCC 2013, p. 20). Local scenarios are available at a location near Mobile Bay, providing estimates of sea level rise at decadal time steps out to the year 2100. We found the average sea rise level estimate for the intermediate high and extreme NOAA scenarios from this station, and used the estimate (rounded to the nearest foot, because shapefiles are only available at 1-foot increments) to project estimated habitat loss at years 2040 and 2070. If SLR estimates overlap with known occupied portions of the river system, we assume that area is no longer suitable, or occupiable, thus resilience would decrease.

Table 6.1-Estimated inundation from sea level rise, in and around the Pearl River basin, according to NOAA's Sea Level Rise Viewer, Dauphin Island local scenarios. (https://coast.noaa.gov/slr/#/layer/sce/0/-9960877.61950826/3551130.846077425/9/satellite/11/0.8/2070/interHigh/midAccretion).

Year	Intermediate-High SLR	Extreme SLR
2040	1 feet (0.30 meters)	2 feet (0.61 meters)
2070	3 feet (0.91 meters)	5 feet (1.52 meters)

6.2.3 Future Water Engineering Projects and Mining

Stream channelization, point-bar mining, and impoundment have been listed as potential threats in a report written before the Pascagoula map turtle and Pearl River map turtle were taxonomically separated (USFWS 2006, p. 2). Between the late 1980s and early 2010s, notable population declines were observed in the stretch of the Pearl River downstream of the Ross Barnett Reservoir (Selman 2020d, p. 194). It is unknown why the population declined, but altered hydrology of this reach may have had an impact.

Plans for new reservoirs on the Pearl River both upstream and downstream of Jackson, Mississippi have been or are being considered as areas of interest (Lindeman 2013, pp. 202-203). Of particular note is the proposed One Lake project. The project proposes a new dam and commercial development area 14.5 km (9 mi) south of the current Ross Barnett Reservoir Dam near Interstate 20. The goal of the One Lake project is to dredge the Pearl River in order to widen, deepen and straighten an additional 16.1 km (10 mi) of waterway for flood control protection and commercial development opportunities.

The One Lake project is still being debated, and there is uncertainty on whether the project will move forward. Because of this uncertainty, we have created 2 scenarios based around the proposed One Lake project: one in which the project goes online, and another assuming the project does not occur within the next 50 years. Because of the likely negative impacts of the proposed One Lake project, we assume a decrease in resilience of the Middle Pearl Strong unit if the project moves forward.

6.3 Models and Scenarios

In order to assess future viability for the Pearl River map turtle, we project two land use and two sea level rise scenarios out to the years 2040 (20 years) and 2070 (50 years). The two land use scenarios are based on two SRES emission scenarios embedded within the FORE-SCE model (A2 and B1) as described in the previous section. The two sea level rise scenarios are based on NOAAs intermediate-high (RCP 6.0) and extreme (RCP 8.5) local scenarios, as described in the previous section. We also include two scenarios regarding the proposed One Lake project, as described in the previous section. This results in eight plausible scenarios at two time steps (2040 and 2070), with the A2-Extreme-One Lake project scenarios representing the highest threat scenario, the B1-Intermediate High-no One Lake project scenario the lowest threat scenario, and the other four scenarios representing moderate threat scenarios (Table 6.2 A-B).

Because data for population factors (occupied tributaries and density) are not comparable through time or space, we do not assess these factors in our future condition analysis. Additionally, we assume the amount of protected land within each unit stays the same within our projection timeframes, although it is possible that additional land could be converted to a protected status or lands could degrade over time. Rather than attempting to categorize future resilience as was done in the current conditions analysis, we indicate a magnitude and direction of anticipated change in resilience of Pearl River map turtle units.

Table 6.2-Scenarios used to model future condition for Pearl River map turtle. Scenarios were built around three factors: land use (emission scenarios A2 and B1), sea level rise (emission scenarios Intermediate High [IH] and Extreme [EX]), and water engineering projects (One Lake Project Yes/No). Scenarios were projected to two time steps: 2040 (A) and 2070 (B).

A) 2040

One Lake Project (Yes)			
		Sea Level Rise	
		Intermediate High	Extreme
Land Use	A2	A2-IH-OneLake 2040	A2-EX-OneLake 2040
	B1	B1-IH-OneLake 2040	B1-EX-OneLake 2040

One Lake Project (No)			
		Sea Level Rise	
		Intermediate High	Extreme
Land Use	A2	A2-IH-NoProject 2040	A2-EX-NoProject 2040
	B1	B1-IH-NoProject 2040	B1-EX-NoProject 2040

B) 2070

One Lake Project (Yes)			
		Sea Level Rise	
		Intermediate High	Extreme
Land Use	A2	A2-IH-OneLake 2070	A2-EX-OneLake 2070
	B1	B1-IH-OneLake 2070	B1-EX-OneLake 2070

One Lake Project (No)			
		Sea Level Rise	
		Intermediate High	Extreme
Land Use	A2	A2-IH-NoProject 2070	A2-EX-NoProject 2070
	B1	B1-IH-NoProject 2070	B1-EX-NoProject 2070

6.4 Results

6.4.1 Land Use and Water Quality

6.4.1.1 Bogue Chitto Unit

Table 6.3 summarizes the results of the future land use projections by climate scenario (A2 and B1) and year (2040 and 2070) for the Bogue Chitto Unit. Development remains low across the entire unit under all scenarios. Agriculture is projected to be high across the entire unit under all scenarios, as was assessed in current conditions, except under the B1 scenario in 2070, where rates of agriculture drop to moderate. Overall, land use was assessed as an overall moderate condition class within the Bogue Chitto Unit across both climate scenarios and future time steps, resulting in no change from current condition.

Table 6.3-Composite land use score for streams within the Bogue Chitto unit, and for the entire unit, based on development and agriculture levels. Climate scenarios (A2 and B1) are separated by projection time steps (2040 and 2070). Streams in bold are mainstem reaches; all other streams are considered tributaries.

2040

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Bogue Chitto	Low	High	Moderate	Low	High	Moderate
Magees	Low	High	Moderate	Low	High	Moderate
Topisaw	Low	High	Moderate	Low	High	Moderate
Bogue Chitto Unit	**Low**	**High**	**Moderate**	**Low**	**High**	**Moderate**

2070

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Bogue Chitto	Low	High	Moderate	Low	Moderate	Moderate
Magees	Low	High	Moderate	Low	Moderate	Moderate
Topisaw	Low	High	Moderate	Low	Moderate	Moderate
Bogue Chitto Unit	**Low**	**High**	**Moderate**	**Low**	**Moderate**	**Moderate**

6.4.1.2 Lower Pearl Unit

The Lower Pearl Unit is the most southern unit that connects to the Gulf of Mexico. This unit will endure effects of sea level rise due to its proximity to the coast. Development is also expected to increase in this unit. Table 6.4 summarizes the results of the future land use projections by climate scenario (A2 and B1) and year (2040 and 2070) for the Lower Pearl Unit. In 2040, under the A2 scenario, development remains low across much of the unit, although the Hobolochitto and West

Pearl are projected to have high amounts of development; agriculture is projected to be high across the entire unit. In 2070, under the A2 scenario, development is anticipated to increase substantially across the entire unit, with East Pearl, Navigation Canal, Hobolochitto and West Pearl, all projected to have high levels of development, and the rest of the unit projected to have moderate development; agriculture is projected to be high across the entire unit. The B1 scenarios, at both 2040 and 2070, predict low levels of development, and moderate levels of agriculture across the entire unit. Based on these predictions, this unit appears relatively stable under the B1 scenarios, but is anticipated to decrease in water quality under the A2 scenarios due to increases in both development and agriculture.

Table 6.4- Composite land use score for streams within the Lower Pearl unit, and for the entire unit, based on development and agriculture levels. Climate scenarios (A2 and B1) are separated by projection time steps (2040 and 2070). Streams in bold are mainstem reaches; all other streams are considered tributaries.

2040

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
East Pearl	Low	High	Moderate	Low	Moderate	Moderate
Hobolochitto	High	High	Low	Low	Moderate	Moderate
Holmes Bayou	Low	High	Moderate	Low	Moderate	Moderate
Lower Pearl	Low	High	Moderate	Low	Moderate	Moderate
Navigation Canal	Low	High	Moderate	Low	Moderate	Moderate
Pushepatapa	Low	High	Moderate	Low	Moderate	Moderate
West Hobolochitto	Low	High	Moderate	Low	Moderate	Moderate
West Pearl	High	High	Low	Low	Moderate	Moderate

Lower Pearl Unit	**Low**	**High**	**Moderate**	**Low**	**Moderate**	**Moderate**

2070

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
East Pearl	High	High	Low	Low	Moderate	Moderate
Hobolochitto	High	High	Low	Low	Moderate	Moderate
Holmes Bayou	Moderate	High	Low	Low	Moderate	Moderate
Lower Pearl	Moderate	High	Low	Low	Moderate	Moderate
Navigation Canal	High	High	Low	Low	Moderate	Moderate
Pushepatapa	Moderate	High	Low	Low	Moderate	Moderate
West Hobolochitto	Moderate	High	Low	Low	Moderate	Moderate
West Pearl	High	High	Low	Low	Moderate	Moderate
Lower Pearl Unit	**High**	**High**	Low	**Low**	**Moderate**	**Moderate**

6.4.1.3 Middle Pearl Silver Unit

Table 6.5 summarizes the results of the future land use projections by climate scenario (A2 and B1) and year (2040 and 2070) for the Middle Pearl Silver unit. Development remains low across the entire unit under all scenarios. Agriculture increases from moderate to high across the entire unit under the A2 scenarios at both time steps. Agriculture stays moderate in 2040 under the B1 scenario, and actually decreases by 2070. Because of this decrease in agriculture across the unit under the B1 scenario, the overall condition class for land use increases from moderate to high in 2070.

Table 6.5- Composite land use score for streams within the Middle Pearl Silver unit, and for the entire unit, based on development and agriculture levels. Climate scenarios (A2 and B1) are separated by projection time steps (2040 and 2070). Streams in bold are mainstem reaches; all other streams are considered tributaries.

2040

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Middle Pearl Silver	**Low**	**High**	**Moderate**	**Low**	**Moderate**	**Moderate**

2070

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Middle Pearl Silver	**Low**	**High**	**Moderate**	**Low**	**Low**	**High**

6.4.1.4 Middle Pearl Strong Unit

Table 6.6 summarizes the results of the future land use projections by climate scenario (A2 and B1) and year (2040 and 2070) for the Middle Pearl Strong Unit. In 2040, development remains moderate across the entire unit under both climate scenarios; agriculture increases from moderate to high across the entire unit under both climate scenarios. In 2070, under the A2 climate scenario, the mainstem Pearl River, is projected to have high levels of development and agriculture, driving the overall score to low; the rest of the unit maintains a moderate overall score. In 2070, under the B1 climate scenario, development and agriculture remains moderate, the same as current condition.

Table 6.6- Composite land use score for streams within the Middle Pearl Strong unit, and for the entire unit, based on development and agriculture levels. Climate scenarios (A2 and B1) are separated by projection time steps (2040 and 2070). Streams in bold are mainstem reaches; all other streams are considered tributaries.

2040

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Middle Pearl Strong	Moderate	High	Moderate	Moderate	High	Moderate
Pelahatchie	Moderate	High	Moderate	Moderate	High	Moderate
Purvis	Moderate	High	Moderate	Moderate	High	Moderate
Strong	Moderate	High	Moderate	Moderate	High	Moderate
Middle Pearl Strong Unit	**Moderate**	**High**	**Moderate**	**Moderate**	**High**	**Moderate**

2070

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Middle Pearl Strong	High	High	Low	Moderate	Moderate	Moderate
Pelahatchie	Moderate	High	Moderate	Moderate	Moderate	Moderate
Purvis	Moderate	High	Moderate	Moderate	Moderate	Moderate
Strong	Moderate	High	Moderate	Moderate	Moderate	Moderate
Middle Pearl Strong Unit		**High**	**Moderate**	**Moderate**	**Moderate**	**Moderate**

6.4.1.5 Upper Pearl Unit

Table 6.7 summarizes the results of the future land use projections by climate scenario (A2 and B1) and year (2040 and 2070) for the Upper Pearl Unit. In 2040, development remains low across the entire unit under both climate scenarios; agriculture increases from moderate to high across the entire unit under both climate scenarios. In 2070, under the A2 climate scenario, development is projected to be low, and agriculture is projected to be high; for the B1 climate scenario, development is projected to be low, and agriculture is projected to be moderate, the same as current condition.

Table 6.7- Composite land use score for streams within the Upper Pearl unit, and for the entire unit, based on development and agriculture levels. Climate scenarios (A2 and B1) are separated by projection time steps (2040 and 2070). Streams in bold are mainstem reaches; all other streams are considered tributaries.

2040

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Lobutcha	Low	High	Moderate	Low	High	Moderate
Tuscolometa	Low	High	Moderate	Low	High	Moderate
Upper Pearl	Low	High	Moderate	Low	High	Moderate
Yockanookany	Low	High	Moderate	Low	High	Moderate
Upper Pearl Unit	Low	High	Moderate	Low	High	Moderate

2070

Stream	A2 Developed	A2 Agriculture	A2 Overall	B1 Developed	B1 Agriculture	B1 Overall
Lobutcha	Low	High	Moderate	Low	Moderate	Moderate
Tuscolometa	Low	High	Moderate	Low	Moderate	Moderate
Upper Pearl	Low	High	Moderate	Low	Moderate	Moderate
Yockanookany	Low	High	Moderate	Low	Moderate	Moderate
Upper Pearl Unit	Low	High	Moderate	Low	Moderate	Moderate

6.4.2 Riparian Forested Cover/Deadwood Abundance Results

Table 6.8 and 6.9 summarize the percentage of riparian area in forested cover currently, and under both climate scenarios at years 2040 and 2070. Although forested cover remains generally high across all of the units, there are some streams that see substantial decreases. For example, in the Lower Pearl unit, the West Pearl is projected to have low (<60%) cover under all climate scenarios. Furthermore, the East Pearl is projected to drop from high (>80%) to low under the A2 2070 scenario, and moderate (60-80%) under the rest of the scenarios. In the Middle Pearl Strong, substantial decreases in forested cover are anticipated in the Strong River and Purvis Creek. In the Upper Pearl Unit, forested cover remains high across the entire unit under all scenarios, thus deadwood availability does not appear to be a limiting factor in this unit. Based on the thresholds established in the current condition chapter, below is a summary of overall forested cover condition classes for the 27 occupied streams:

- Current Condition: 14 high; 6 moderate
- A2 2040: 14 high; 4 moderate; 2 low. East Pearl and Strong drop from high to moderate; Purvis creek drops from high to low; West Pearl drops from moderate to low; all other streams are projected to have a high ranking for forested riparian cover.
- A2 2070: 10 high; 6 moderate; 4 low. Magees, Hobolochitto, and Strong drop from high to moderate; Pearl Navigation Canal, East Pearl and Purvis drop from high tall other streams are projected to have a high ranking for forested riparian cover; all other streams are projected to have a high ranking for forested riparian cover.
- B1 2040: 17 high; 2 moderate; 1 low. East Pearl and Purvis drop from high to moderate; West Pearl drops from moderate to low; all other streams are projected to have a high ranking for forested riparian cover.
- B1 2070: 16 high; 3 moderate; 1 low. East Pearl and Purvis drop from high to moderate; West Pearl drops from moderate to low; Topisaw remains moderate; all other streams are projected to have a high ranking for forested riparian cover.

Table 6.8-Percentage of forested cover within riparian areas for occupied within resilience units, currently, and under 2 land use/climate scenarios (A2 and B1) at 2 time steps (2040 and 2070).

		Current	**A2 2040**	**A2 2070**	**B1 2040**	**B1 2070**
Unit	Stream	Forested %	Forested %	Forested %	Forested %	Forested %
Bogue Chitto	Bogue Chitto	72.37	85.29	82.05	86.99	88.47
	Magees	89.83	80.52	67.01	84.64	87.90
	Topisaw	75.68	71.64	66.08	80.22	77.37
Lower Pearl	East Pearl	85.37	66.48	41.70	69.53	69.51
	Hobolochitto	98.32	81.80	79.36	99.26	100.01
	Holmes Bayou	95.06	95.22	92.18	95.22	95.22
	Lower Pearl	61.21	94.17	92.41	95.44	95.87
	Navigation Canal	84.62	94.08	37.08	93.92	93.92
	Pushepatapa	96.62	88.83	89.05	87.53	96.52
	West Hobolochitto	97.33	90.35	82.26	92.81	94.24
	West Pearl	60.36	46.05	35.44	58.63	58.09
Middle Pearl Silver	Middle Pearl Silver	60.42	82.94	77.17	89.18	88.91
Middle Pearl Strong	Middle Pearl Strong	70.99	79.96	70.89	87.29	86.39
	Pelahatchie	91.10	86.86	80.85	91.61	90.75
	Purvis	82.73	58.89	46.43	73.01	75.91
	Strong	92.39	75.68	68.46	82.53	82.84
Upper Pearl	Lobutcha	95.54	90.81	86.35	93.18	93.43
	Tuscolometa	94.01	93.89	91.10	98.70	96.07
	Upper Pearl	89.82	95.43	94.89	97.02	97.74
	Yockanookany	92.17	89.18	85.87	93.32	91.44

Table 6.9-Overall composite scores for the habitat factor forested cover, within riparian areas for resilience units, currently, and under 2 land use/climate scenarios (A2 and B1) at 2 time steps (2040 and 2070).

		Current	**A2 2040**	**A2 2070**	**B1 2040**	**B1 2070**
Unit	Stream	Forested %	Forested %	Forested %	Forested %	Forested %
Bogue Chitto	Bogue Chitto	Moderate	High	High	High	High
	Magees	High	High	Moderate	High	High
	Topisaw	Moderate	Moderate	Moderate	High	Moderate
Lower Pearl	East Pearl	High	Moderate	Low	Moderate	Moderate
	Hobolochitto	High	High	Moderate	High	High
	Holmes Bayou	High	High	High	High	High
	Lower Pearl	Moderate	High	High	High	High
	Navigation Canal	High	High	Low	High	High
	Pushepatapa	High	High	High	High	High
	West Hobolochitto	High	High	High	High	High
	West Pearl	Moderate	Low	Low	Low	Low
Middle Pearl Silver	Middle Pearl Silver	Moderate	High	Moderate	High	High
Middle Pearl Strong	Middle Pearl Strong	Moderate	Moderate	Moderate	High	High
	Pelahatchie	High	High	High	High	High
	Purvis	High	Low	Low	Moderate	Moderate
	Strong	High	Moderate	Moderate	High	High
Upper Pearl	Lobutcha	High	High	High	High	High
	Tuscolometa	High	High	High	High	High
	Upper Pearl	High	High	High	High	High
	Yockanookany	High	High	High	High	High

6.4.3 Future Water Engineering: One Lake Project

If the One Lake project is implemented, it will alter the hydrologic regime of this stretch of the Pearl River substantially, converting habitat from lotic to lentic, which will have negative consequences on the Pearl River map turtle (Selman 2020d, p. 194). The conversion of riverine habitat to a lake setting will reduce water velocity, limiting the input of deadwood, a critical component of Pearl River map turtle habitat (Lindeman 1999, p. 40; Selman 2020d, p. 194). This project could impact instream riverine habitat (i.e., altered flows may alter prey resources) and adjacent nesting habitat (i.e., reduced flooding during spring, could prevent sandbar scouring and narrow nesting habitat) (Selman 2020d, p. 194). Researchers have estimated that up to 170 individual Pearl River map turtles could be directly impacted, and up to 360 indirectly impacted, both upstream and downstream, by the One Lake Project (Selman 2020d, p. 194). For all scenarios in which we assume the One Lake Project comes on line, we predict a substantial drop in resilience for the mainstem Pearl River within the Middle Pearl Strong.

6.4.4 Sea Level Rise

We used NOAA estimates of sea level rise inundation under the intermediate-high and extreme emissions scenarios for the years 2040 and 2070. Estimated inundation levels for the intermediate-high scenario are one foot (0.30m) in 2040, and three feet (0.91m) in 2070. For the extreme scenario, estimated inundation levels for the intermediate-high scenario are two feet (0.61m) in 2040, and five feet (1.52m) in 2070 and may be exacerbated by salt water intrusion due to storm surge from increased storm frequency. As anticipated, only the southern coastal unit, the Lower Pearl, is predicted to be impacted by sea level rise. Portions of the Lower Pearl unit are projected to be inundated by sea level rise, such as the mainstem West and East Pearl rivers, and these impacts are seen as soon as 2040 under both climate scenarios. Table 6.10 shows the estimated amount of known occupied habitat anticipated to be inundated for one (0.30m), two (0.61m), three (0.91m), and five feet (1.52m) of sea level rise. Between 13.7-17.4 rkm (6.3-10.8 rm) of occupied habitat in the East Pearl River is projected to be inundated under the various sea level rise scenarios (Figures 6.2-6.5). This area in the East Pearl River, which is directly adjacent to Stennis Space Center, is an area where there have been many detections of Pearl River Map turtles. Although we would expect individual turtles to move in response to inundation, the fact remains that suitable habitat will be lost in an area that supports a fairly robust population of the species. Only under five feet (1.52m) of sea level rise is there projected to be any impacts in the West Pearl River, where one isolated detection is projected to be inundated.

Table 6.10-estimated number of known occupied river miles for Pearl River map turtles projected to be inundated at 1ft (0.30m), 2ft (0.61m), 3ft (0.91m), and 5ft (1.52m) of sea level rise.

Sea Level Rise (feet/meters)	River Miles/Kilometers Inundated
1ft (0.3m)	6.3rmi/10.1rkm (East Pearl)
2ft (0.6m)	8.5rmi/13.68rkm (East Pearl)
3ft (0.9m)	9.9rmi/15.9rkm (East Pearl)
5ft (1.5m)	10.8rmi/17.4rkm (East Pearl); small portion of West Pearl

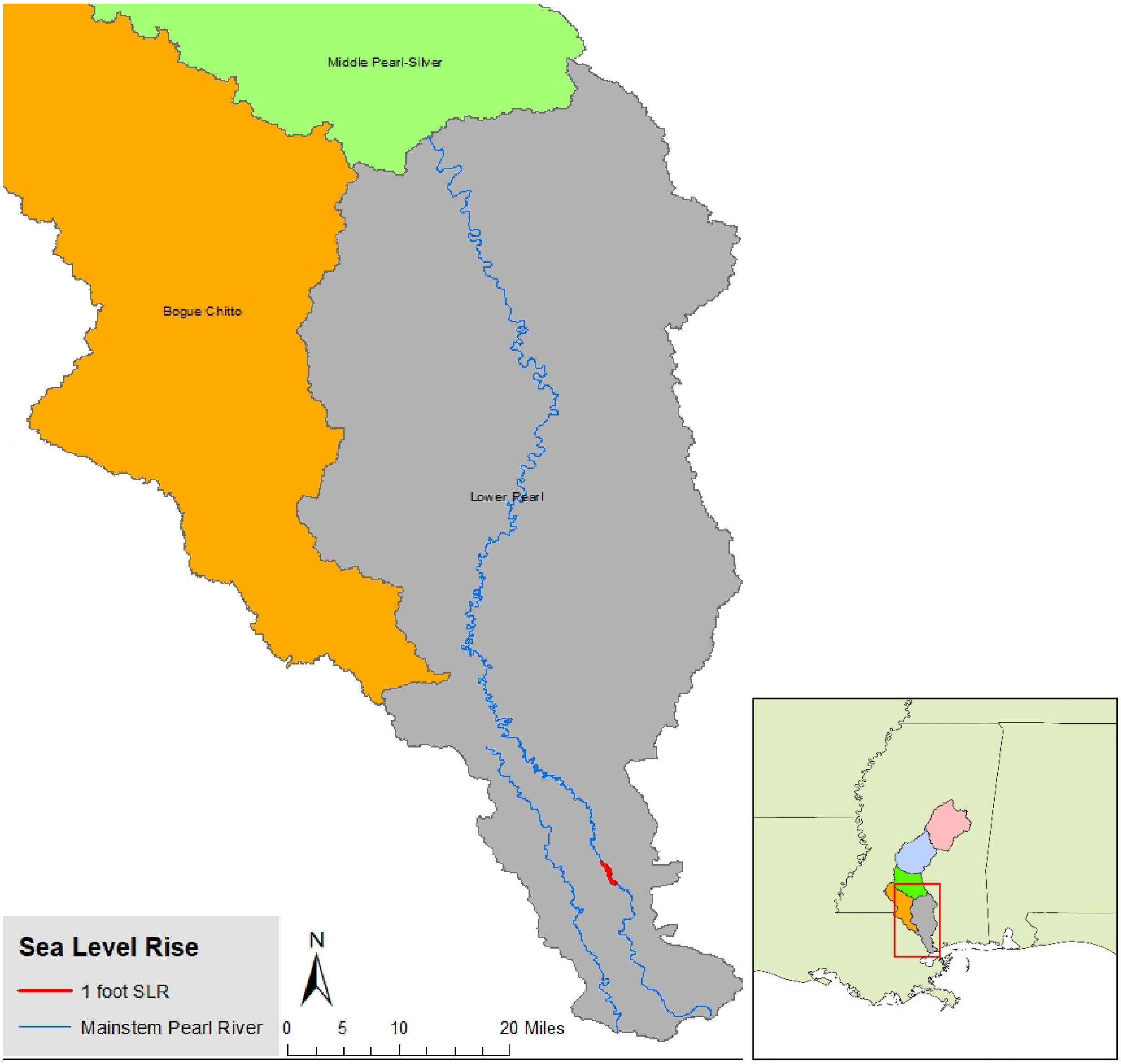

Figure 6.2-Occupied portion of the range of the Pearl River map turtle, anticipated to be inundated by 1 foot of sea level rise (red).

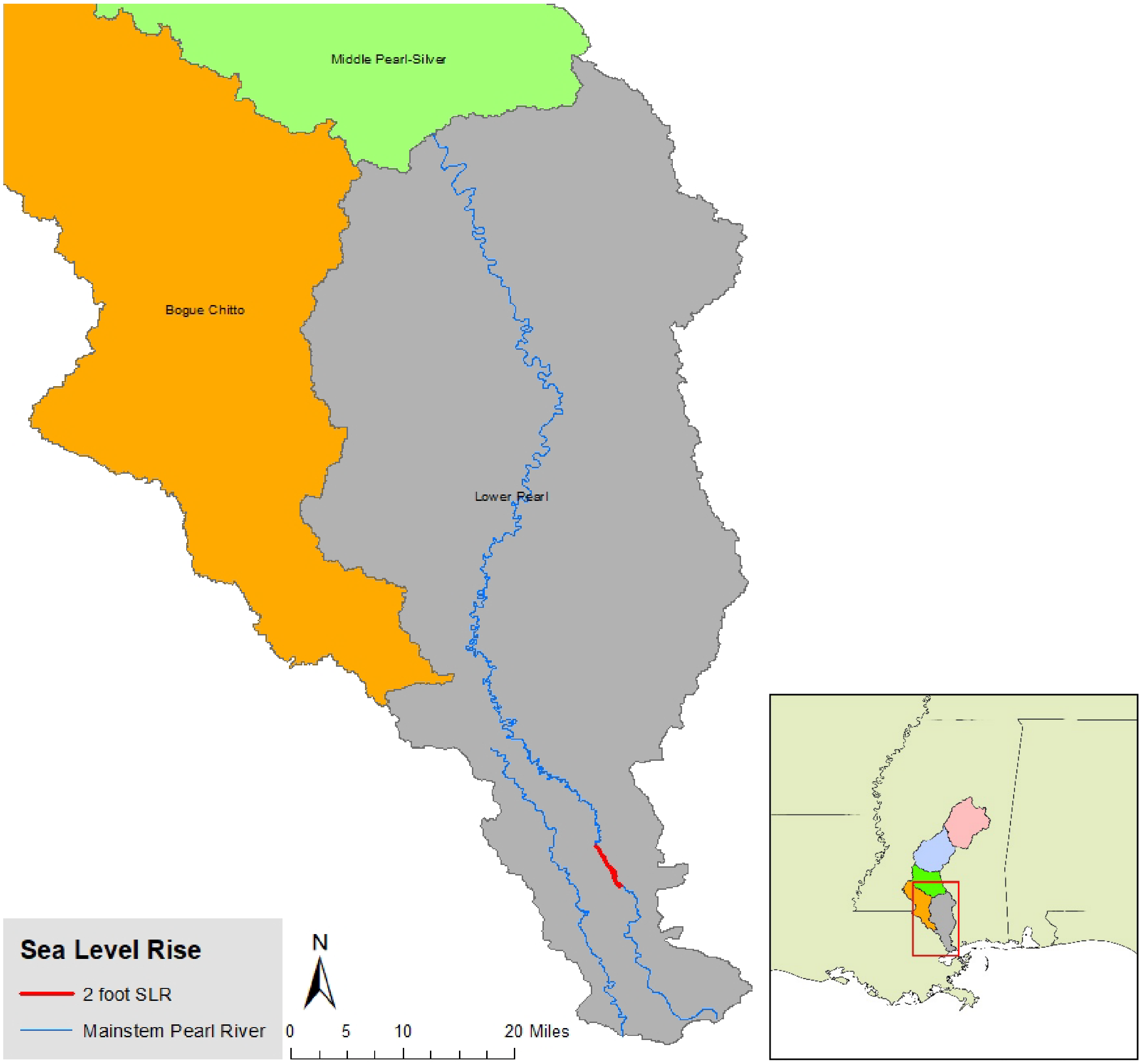

Figure 6.3-Occupied portion of the range of the Pearl River map turtle, anticipated to be inundated by 2 feet (0.61m) of sea level rise (red).

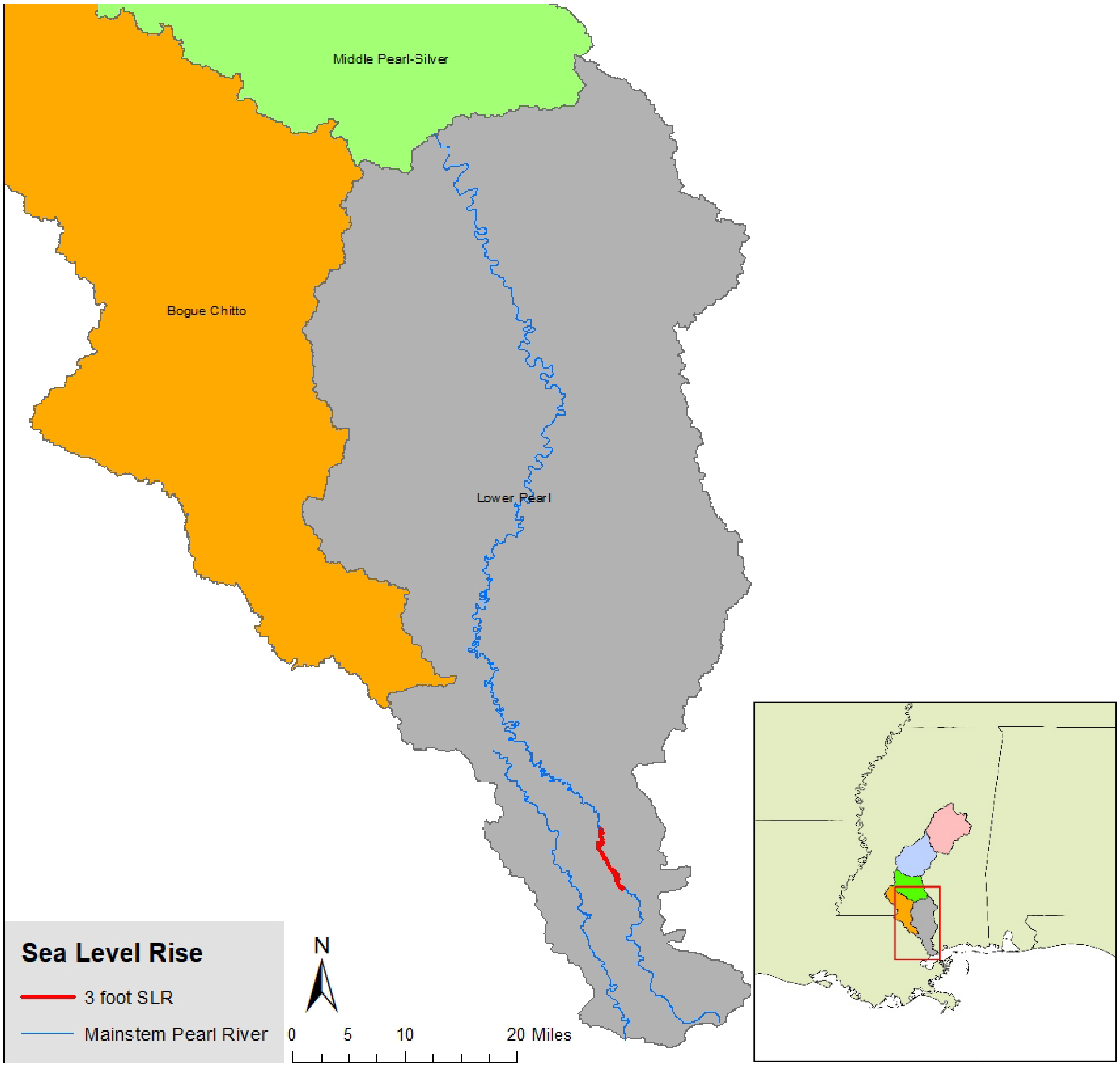

Figure 6.4-Occupied portion of the range of the Pearl River map turtle, anticipated to be inundated by 3 feet (0.91m) of sea level rise (red).

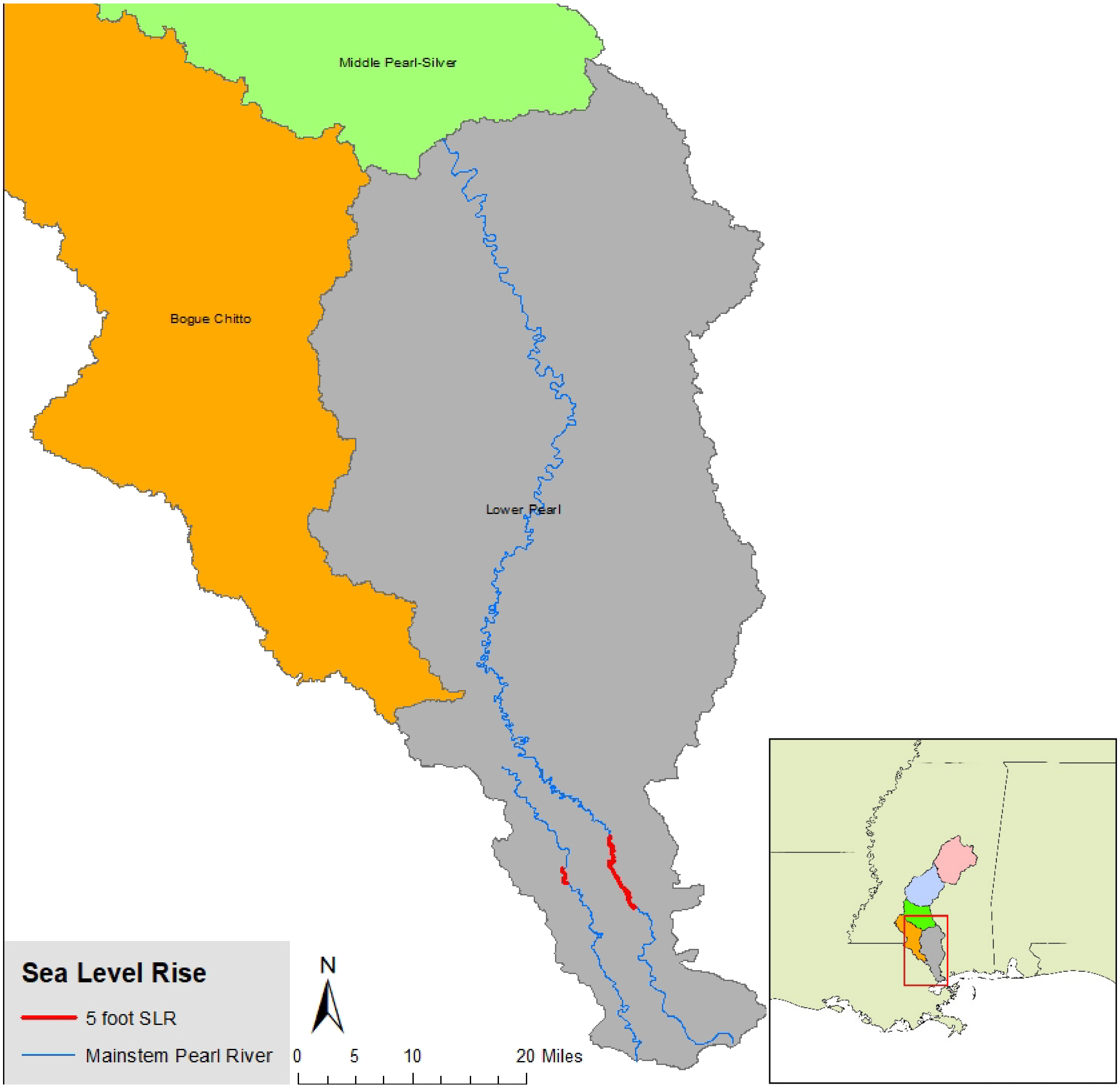

Figure 6.5-Occupied portion of the range of the Pearl River map turtle, anticipated to be inundated by 5 feet (1.52m) of sea level rise (red).

6.5 Summary of Future Conditions and Viability based on Resiliency, Representation, and Redundancy

6.5.1 Future Resiliency

To assess future resiliency of Pearl River map turtle resilience units, we assessed the potential impacts of sea level rise and the predicted change in the condition of three habitat factors: land use/water quality, forested riparian cover, and construction of major reservoirs. We assess future resiliency at the years 2040 and 2070, under two climate/land use scenarios (A2 and B1), two sea level rise scenarios (intermediate high and extreme), and two water engineering project scenarios (One Lake project: yes/no). Below, and in Table 6.11, we summarize the results of the future scenarios.

- ***Bogue Chitto***- Under all scenarios, development remains low across the Bogue Chitto unit. Agriculture is high across the entire unit in all scenarios, except for the B1 scenarios in the year 2070, where agriculture is moderate. Forested cover is relatively high across the unit under all scenarios; thus, deadwood does not appear to be a limiting factor. There are no predicted sea level rise or water engineering project impacts. It is likely that the unit maintains a moderate resilience, though there is uncertainty in future mining activity, which has the potential of significantly decreasing resiliency.
- ***Lower Pearl***- *S*ea level rise impacts this unit under all scenarios, although the impacts of inundation are localized to the southern portion of the unit, mainly in the East Pearl River. Under the A2 scenarios, there are a few streams that are impacted by high levels of development, although most of the unit has low levels of development; under the B1 scenarios, development is low across the entire unit. Agriculture is predicted to be high across the unit under the A2 scenarios, and moderate across the unit under the B1 scenarios. There are no predicted water engineering projects, and forested cover is anticipated to be relatively high. Current resilience for this unit is low, and resilience is anticipated to decrease across all scenarios, with the A2 scenarios with extreme sea level rise will see the most substantial decreases.

- ***Middle Pearl Silver***- Development remains low across the unit under all scenarios at both time steps. Agriculture increases to high under the A2 scenarios and stays moderate under the B1 scenarios. There are no predicted sea level rise or water engineering project impacts. Forested cover is relatively high across the unit under all scenarios, and is actually predicted to increase under B1 scenarios, thus deadwood does not appear to be a limiting factor. Current resilience for this unit is low, and it is likely there will not be any decreases in resilience in the future based on the factors assessed.
- ***Middle Pearl Strong***- Development is substantial in a few areas within this unit, particularly around Jackson, Mississippi. Agriculture is predicted to be high across the unit under all scenarios. If the One Lake project goes online, there is predicted to be a substantial decrease in resilience within and adjacent to the project area. A few streams are predicted to lose a substantial amount of forested cover. No sea level rise impacts are predicted in this unit. The Middle Pearl Strong unit is perhaps the most vulnerable unit, as development, agriculture, and water engineering projects are all potential stressors to the species.
- ***Upper Pearl***- Development remains low across the entire unit under all scenarios. Agriculture is high across the entire unit in all scenarios, except for the B1 scenarios in the year 2070, where agriculture is moderate. Forested cover is relatively high across the unit under all scenarios; thus, deadwood does not appear to be a limiting factor. There are no predicted sea level rise or water engineering project impacts. The Upper Pearl Unit is likely to remain at a moderate level of resilience, and is likely a stronghold for the species, as threats are projected to be low, and this unit has the highest amount of protected land compared to all other units.

Table 6.11-Summary of projected habitat factors and resilience for resilience units at A) Year 2040 Land Use/Climate Scenario A2 + 1ft (0.30m) Sea Level Rise, B) Year 2040 Land Use/Climate Scenario A2 + 2ft (0.61m) Sea Level Rise, C) Year 2040 Land Use/Climate Scenario B1 + 1ft (0.30m) Sea Level Rise, D) Year 2040 Land Use/Climate Scenario B1 + 2ft (0.61m) Sea Level Rise, E) Year 2070 Land Use/Climate Scenario A2 + 3ft (0.91m) Sea Level Rise, F) Year 2070 Land Use/Climate Scenario A2 + 5ft (1.52m) Sea Level Rise, G)Year 2070 Land Use/Climate Scenario B1 + 3ft (0.91m) Sea Level Rise, and H) Year 2070 Land Use/Climate Scenario B1 + 5ft (1.52m) Sea Level Rise.

A) Year 2040 Land Use/Climate Scenario A2 + 1ft (0.30m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development remains low across much of the unit, although the Hoblochitto and West Pearl are projected to have high amounts of development; agriculture is projected to be high across the entire unit; forested cover remains stable; SLR impacts approximately 6 river miles of known occupied habitat in the southern extent of the East Pearl.	Likely a decrease in resilience due to loss of habitat in the southern portion of the unit due to SLR, high levels of agriculture across the unit, and high development in two streams.
Middle Pearl Silver	Development remains low; agriculture increases from moderate to high; forested cover increases; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (low) due to increase in agriculture, although this could be offset by increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development remains moderate; agriculture increases to high; 2 streams (Strong and Purvis) decrease substantially in forested cover; no SLR or reservoir projects predicted.	Likely decrease to current resiliency (moderate) due to increased agriculture and decreased forested cover within unit.
Middle Pearl Strong (One Lake Project)	Development remains moderate; agriculture increases to high; 2 streams (Strong and Purvis) decrease substantially in forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals	Likely to substantially decrease resiliency. Although the unit is highly likely to remain extant, there will likely be substantial loss of habitat and degradation of remaining habitat due to One Lake project and increased levels of development and agriculture.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

B) Year 2040 Land Use/Climate Scenario A2 + 2ft (0.61m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development remains low across much of the unit, although the Hoblochitto and West Pearl are projected to have high amounts of development; agriculture is projected to be high across the entire unit; forested cover remains stable; SLR impacts approximately 8 river miles of known occupied habitat in the southern extent of the East Pearl.	Likely a significant decrease in resilience due to loss of habitat in the southern portion of the unit due to SLR, high levels of agriculture across the unit, and high development in two streams.
Middle Pearl Silver	Development remains low; agriculture increases from moderate to high; forested cover increases; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (low) due to increase in agriculture, although this could be offset by increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development remains moderate; agriculture increases to high; 2 streams (Strong and Purvis) decrease substantially in forested cover; no SLR or reservoir projects predicted.	Likely decrease to current resiliency (moderate) due to increased agriculture and decreased forested cover within unit.
Middle Pearl Strong (One Lake Project)	Development remains moderate; agriculture increases to high; 2 streams (Strong and Purvis) decrease substantially in forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals	Likely to substantially decrease resiliency. Although the unit is highly likely to remain extant, there will likely be substantial loss of habitat and degradation of remaining habitat due to One Lake project and increased levels of development and agriculture.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

C) Year 2040 Land Use/Climate Scenario B1 + 1ft (0.30m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Low levels of development, and moderate levels of agriculture across the entire unit; forested cover remains stable; SLR impacts approximately 6 river miles of known occupied habitat in the southern extent of the East Pearl.	Decrease in resilience in the southern portion of the unit because of loss of habitat due to SLR. The rest of the unit is likely to maintain current resilience (low).
Middle Pearl Silver	Development remains low; agriculture stays moderate; forested cover increases; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development remains moderate; agriculture increases to high; stable forested cover; no SLR or reservoir projects predicted.	Likely a slight decrease to current resiliency (moderate) due to increased agriculture.
Middle Pearl Strong (One Lake Project)	Development remains moderate; agriculture increases to high; stable forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals.	Likely to substantially decrease resiliency. Although the unit is highly likely to remain extant, there will likely be substantial loss of habitat and degradation of remaining habitat due to One Lake project and increased levels of development and agriculture.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

D) Year 2040 Land Use/Climate Scenario B1 + 2ft (0.61m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Low levels of development, and moderate levels of agriculture across the entire unit; forested cover remains stable; SLR impacts approximately 8 river miles of known occupied habitat in the southern extent of the East Pearl.	Decrease in resilience in the southern portion of the unit because of loss of habitat due to SLR. The rest of the unit is likely to maintain current resilience (low).
Middle Pearl Silver	Development remains low; agriculture stays moderate; forested cover increases; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development remains moderate; agriculture increases to high; stable forested cover; no SLR or reservoir projects predicted.	Likely a slight decrease to current resiliency (moderate) due to increased agriculture.
Middle Pearl Strong (One Lake Project)	Development remains moderate; agriculture increases to high; stable forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals.	Likely to substantially decrease resiliency. Although the unit is highly likely to remain extant, there will likely be substantial loss of habitat and degradation of remaining habitat due to One Lake project and increased levels of development and agriculture.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

E) Year 2070 Land Use/Climate Scenario A2 + 3ft (0.91m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development and agriculture increase significantly across the entire unit. Forested cover decreases substantially in the East Pearl, Hoblochitto, and Pearl Navigation Canal. SLR impacts approximately 9 river miles of known occupied habitat in the southern extent of the East Pearl.	Likely a significant decrease in resilience due to loss of habitat in the southern portion of the unit due to SLR, high levels of agriculture across the unit, significant increases in development, and substantial decreases in forested cover in 3 streams.
Middle Pearl Silver	Development remains low; agriculture increases from moderate to high; stable forested cover; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (low) due to increase in agriculture.
Middle Pearl Strong (No One Lake Project)	Development is high in the mainstem, and moderate in all other streams; agriculture increases to high; forested cover decreases across most of the unit; no SLR or reservoir projects predicted.	Likely a significant decrease to current resilience, potentially from moderate to low due to high levels of development and agriculture and decreases in forested cover.
Middle Pearl Strong (One Lake Project)	Development is high in the mainstem, and moderate in all other streams; agriculture increases to high; forested cover decreases across most of the unit; no SLR predicted; One Lake project results in loss of habitat and individuals.	Likely a significant decrease to current resilience, likely from moderate to low due to high levels of development and agriculture and decreases in forested cover. Compounding these issues is the One Lake Project, which will result in significant loss and degradation of habitat.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

F) Year 2070 Land Use/Climate Scenario A2 + 5ft (1.52m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development and agriculture increase significantly across the entire unit. Forested cover decreases substantially in the East Pearl, Hoblochitto, and Pearl Navigation Canal. SLR impacts approximately 10 river miles of known occupied habitat in the southern extent of the East Pearl and a small portion of occupied habitat in the West Pearl just north of I-10.	Likely a significant decrease in resilience due to loss of habitat in the southern portion of the unit due to SLR, high levels of agriculture across the unit, significant increases in development, and substantial decreases in forested cover in 3 streams.
Middle Pearl Silver	Development remains low; agriculture increases from moderate to high; stable forested cover; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (low) due to increase in agriculture.
Middle Pearl Strong (No One Lake Project)	Development is high in the mainstem, and moderate in all other streams; agriculture increases to high; forested cover decreases across most of the unit; no SLR or reservoir projects predicted.	Likely a significant decrease to current resilience, potentially from moderate to low due to high levels of development and agriculture, and decreases in forested cover.
Middle Pearl Strong (One Lake Project)	Development is high in the mainstem, and moderate in all other streams; agriculture increases to high; forested cover decreases across most of the unit; no SLR predicted. One Lake project results in loss of habitat and individuals.	Likely a significant decrease to current resilience, likely from moderate to low due to high levels of development and agriculture, and decreases in forested cover. Compounding these issues is the One Lake Project, which will result in significant loss and degradation of habitat.
Upper Pearl	Development remains low; agriculture increases to high; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely a slight decrease to current resilience (moderate) due to increase in agriculture, although this could be offset by high levels of forested cover.

G) Year 2070 Land Use/Climate Scenario B1 + 3ft (0.91m) Sea Level Rise, and H) Year 2070 Land Use/Climate Scenario B1 + 5ft (1.52m) Sea Level Rise

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture remains high; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development is low across the entire unit. Agriculture remains moderate across the entire unit. Forested cover is high. SLR impacts approximately 9 river miles of known occupied habitat in the southern extent of the East Pearl.	Likely a slight decrease to resilience due to SLR impacts in the southern portion of the unit. Land use and forested cover remain stable.
Middle Pearl Silver	Development remains low; agriculture stays moderate; forested cover increases; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development and agriculture stay at moderate levels; stable forested cover; no SLR or reservoir projects predicted.	Resilience is likely to stay moderate; the same as current condition.
Middle Pearl Strong (One Lake Project)	Development and agriculture stay at moderate levels; stable forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals.	Resilience will decrease in the areas in and adjacent to the One Lake Project area due to significant loss and degradation of habitat. The rest of the unit will likely remain stable due to no significant increases in other threats.
Upper Pearl	Development remains low; agriculture remains moderate; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to increases in forested cover.

H) Year 2070 Land Use/Climate Scenario B1 + 5ft (1.52m) Sea Level Rise.

Unit	Future Habitat Factors	Future Resilience
Bogue Chitto	Development remains low; agriculture decreases to moderate; stable forested cover; no SLR or reservoir projects predicted.	Likely to maintain moderate resilience, although uncertainty remains regarding future mining activity.
Lower Pearl	Development is low across the entire unit. Agriculture remains moderate across the entire unit. Forested cover is high. SLR impacts approximately 10 river miles of known occupied habitat in the southern extent of the East Pearl and a small portion of occupied habitat in the West Pearl just north of I-10.	Likely a slight decrease to resilience due to SLR impacts in the southern portion of the unit. Land use and forested cover remain stable.
Middle Pearl Silver	Development remains low; agriculture decreases from moderate to low; forested cover increases; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to decreased agriculture and increases in forested cover.
Middle Pearl Strong (No One Lake Project)	Development and agriculture stay at moderate levels; stable forested cover; no SLR or reservoir projects predicted.	Resilience is likely to stay moderate; the same as current condition.
Middle Pearl Strong (One Lake Project)	Development and agriculture stay at moderate levels; stable forested cover; no SLR predicted. One Lake project results in loss of habitat and individuals.	Resilience will decrease in the areas in and adjacent to the One Lake Project area due to significant loss and degradation of habitat. The rest of the unit will likely remain stable due to no significant increases in other threats.
Upper Pearl	Development remains low; agriculture remains moderate; forested cover is very high across the entire unit; no SLR or reservoir projects predicted.	Likely to maintain current resilience (low), or potentially increase slightly due to increases in forested cover.

6.5.2 Future Representation

Representation refers to the breadth of genetic and environmental diversity within and among populations, which influences the ability of a species to adapt to changing environmental conditions over time. Differences in life history traits, habitat features, and/or genetics across a species range often aid in the delineation of representative units, which are used to assess species representation.

We consider the entire range of the Pearl River map turtle to be one representative unit based on a recent study that found no distinct genetic structure was present across the species range (Pearson et al. 2020, pp. 11-12). However, the Strong River, located in the Pearl River Strong unit, may have some unique habitat features that could confer unique adaptive pressure (Lindeman pers. comm. 2020, p. 4), most notably, the Strong River has some very rocky stretches that are unlike anything else in the drainage, water quality appears to be better than most of the rest of the drainage, and fish diversity is high in these rocky areas (Selman pers. comm. 2020a, p. 7). Although we do not consider the Strong River to be a separate representative unit, we consider the Strong River to be a potentially significant stream because of the unique characteristics of the habitat.

When looking at projections of threats within the Strong River, a few general trends can be seen. First, for land use, development is projected to remain low. In the A2 climate scenarios, agriculture increases from moderate to high; in the B1 climate scenarios, agriculture stays moderate. Also, forested cover within the riparian zone of the Strong River remains relatively high (68-83%), although it does drop across all climate scenarios from current condition (92%). Sea level rise does not impact this river in any of our scenarios, as the Strong River is far enough inland to avoid the effects of inundation. Finally, the One Lake project is not anticipated to directly impact the Strong River due to the location of the project (i.e. mainstem Pearl River). Given all of this information, although the resilience of the Strong River might decrease slightly due to land use projections, it is likely the Strong River will support a moderate density of individual turtles, and thus contribute to representation through maintenance of potential genetic diversity based on unique habitat features. It is noteworthy that a recent genetics study has revealed that genetic diversity is lower in Pearl River map turtles compared to Pascagoula map turtles (Pearson et al. 2020, pp. 11-12). Declining populations generally have reduced genetic diversity, which can potentially elevate the risk of extinction by reducing a species' ability and potential to adapt to environmental changes (Spielman et al. 2004, entire). Future studies could help to elucidate whether levels of genetic diversity seen in Pearl River map turtles are low enough to suggest potential genetic bottlenecks, thus clarifying the species level of representation.

6.5.3 Future Redundancy

Redundancy refers to the ability of a species to withstand catastrophic events and is measured by the amount and distribution of resilient populations across the species range. Catastrophic events that could severely impact or extirpate entire Pearl River map turtle units include chemical spills, changes in upstream land use that alters stream characteristics and water quality downstream, dam construction with a reservoir drowning lotic river habitat, and potential effects of climate change such as rising temperatures and sea level rise. Although we do not project any of the units to be extirpated in any scenarios, we do anticipate resilience to drop significantly in several units across many scenarios. For example, the Middle Pearl Strong unit will likely lose a substantial amount of habitat and individuals under all scenarios in which the One Lake project is initiated. Also, the Lower Pearl unit will be impacted by sea level rise under all scenarios, and this is compounded by projected increases in both development and agriculture. All other units are anticipated to remain relatively stable. Because extant units of the species are predicted to be distributed relatively widely, it is highly unlikely that a catastrophic event would impact the entire species' range, thus the Pearl River map turtle is predicted to exhibit a moderate degree of redundancy in the future under all scenarios.

6.5.4 Future Conditions Summary

We assessed future resilience as a function of four habitat/threat factors (land use, sea level rise, riparian forested cover, and future water engineering projects). Based on these factors, there are two populations predicted to substantially decrease in resilience across most of the future scenarios: Lower Pearl and Middle Pearl Strong. The Lower Pearl unit faces a myriad of future threats, including impacts from seal level rise in the southern portion of the unit, substantial increases in development and agriculture within the A2 climate scenarios, and loss of forested cover within the A2 climate scenarios. For the Middle Pearl Strong, the magnitude of decrease is most closely tied to the One Lake project. If the One Lake Project moves forward within the next 50 years, areas in and around the project site are anticipated to be substantially negatively impacted. The Bogue Chitto, Middle Pearl Silver, and Upper Pearl units are anticipated to maintain their current resilience, or only slightly decrease, as the main threats assessed are not anticipated to increase markedly in the future. One caveat for the Bogue Chitto: there is uncertainty in levels and patterns of mining in the future, and this has been identified as a current threat in some portions of the unit.

Regarding redundancy, all five units are predicted to remain extant under all scenarios, and those units are distributed relatively widely, thus it is unlikely that a catastrophic event would impact the entire range of the species. We did not assess multiple representative units for the species due to recent research indicating no significant genetic structuring. Species experts indicated the Strong River might be unique from a habitat perspective (i.e. distinctly rocky stretches), so it is notable that the Strong River is not predicted to significantly increase any primary threats. If the Strong River is indeed conferring different adaptive pressures due to the unique habitat type found there, the Strong River is predicted to continue contributing to representation for the species.

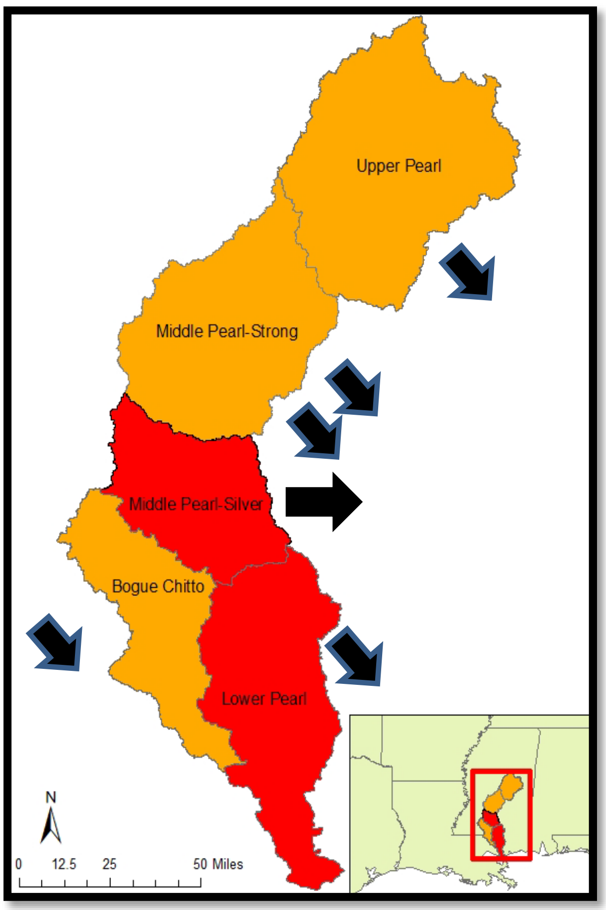

Figure 6.6- Future resiliency of Pearl River map turtle analysis units. The arrows indicate direction of change from current to future conditions. The color of the units indicates the current condition; red=low, orange=moderate, and green= high.

LITERATURE CITED

Agha, M., J.R. Ennen, D.S. Bower, A.J. Nowakowski, S.C. Sweat, and B.D. Todd. 2018. Salinity tolerances and use of saline environments by freshwater turtles: implications of sea level rise. Biological Reviews 93:1634-1648.

Agha, M., Y.R. Yanagitsuru, N.A. Fangue, A.J. Nowakowski, L.V. Kojima, J.J. Cech Jr, M.K. Riley, D.E. Cocherell, and B.D. Todd. 2019. Physiological consequences of rising water salinity for a declining freshwater turtle. Conservation Physiology 7:1-10.

Alder, J. & Hostetler, S. W., 2013. U.S. Geological Survey, National Climate Change Viewer. [Online] Available at: https://www2.usgs.gov/climate_landuse/clu_rd/nccv.asp <Accessed June 2020>

Anderson, P.K. 1958. The Photic Responses and Water-Approach Behavior of Hatchling Turtles. Copeia 1958:211-215.

Backlund, P., A. Janetos, D. Schimel, J. Hatfield, K. Boote, P. Fay, L. Hahn, C. Izaurralde, B.A. Kimball, T. Mader, J. Morgan, D. Ort, W. Polley, A. Thomson, D. Wolfe, M. Ryan, S. Archer, R. Birdsey, C. Dahm, L. Heath, J. Hicke, D. Hollinger, T. Huxman, G. Okin, R. Oren, J. Randerson, W. Schlesinger, D. Lettenmaier, D. Major, L. Poff, S. Running, L. Hansen, D. Inouye, B.P. Kelly, L Meyerson, B. Peterson, R. Shaw. 2008. The effects of climate change on agriculture, land resources, water resources, and biodiversity. A Report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research. U.S. Environmental Protection Agency, Washington, D.C. 362 pp.

Baur, G. 1893. Two new species of North American Testudinata. The American Naturalist 27:675−677.

Beever EA, O'Leary J, Mengelt C, West JM, Julius S, Green N, Magness D, Petes L, Stein B, Nicotra AB, Hellmann JJ. 2016. Improving conservation outcomes with a new paradigm for understanding species' fundamental and realized adaptive capacity. Conservation Letters 9:131-137.

Beja, P. and R. Alcazar. 2003. Conservation of Mediterranean temporary ponds under agricultural intensification: an evaluation using amphibians. Biological Conservation 114:317-326.

Bennett, M.G., B.R. Kuhajda, J.H. Howell. 2008. Status of the frecklebelly madtom, *Notorus munitus*, (Siluriformes: Ictaluridae): a review of data from field surveys, museum records and the literature. Southeastern Naturalist 7:459-474.

Booth, D.B. and C.R. Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association 33:1077-1090.

Box, J.B., and J. Mossa. 1999. Sediment, land use, and freshwater mussels: prospects and problems. Journal of the North American Benthological Society 18:99-117.

Brenes, R., D.L. Miller, T.B. Waltzek, R.P. Wilkes, J.L. Tucker, J.C. Chaney, R.H. Hardman, M.D. Brand, R.R. Huether, and M.J. Gray. 2014. Susceptibility of fish and turtles to three ranaviruses isolated from different ectothermic vertebrate classes. Journal of Aquatic Animal Health 26:118-126.

Buckner, M.M., W. Smith, J.A. Takats. 2002. Tennessee, Cumberland, and Mobile River Basins at Risk. A Biological Assessment and Vision for the World Wildlife Fund's Southeast Rivers and Streams Project. Nashville, TN. 52 pp. Available online: www.worldwildlife.org/what/wherewework/sers/WWFBinaryitem2738.pdf <Last accessed June 2020>

Buhlmann, K.A. 2014. Assessment of ringed map turtle (*Graptemys oculifera*) and Pearl map turtle (*Graptemys pearlensis*) abundance and habitat on the U.S. Navy's Stennis Western Maneuver Area East Pearl and Mike's Rivers, Mississippi, with recommendations for habitat enhancement. Unpublished report submitted to Stennis Space Center, Stennis Western Maneuver Area, Mississippi and Naval Facilities Engineering Command, Jacksonville, FL. 34 pp.

Buhlmann, K. and G. Coffman. 2001. Fire Ant Predation of Turtle Nests and Implications for the Strategy of Delayed Emergence. Journal of the Elisha Mitchell Scientific Society 117:94-100.

Buhlmann, K.A. and J.W. Gibbons. 1997. Imperiled aquatic reptiles of the southeastern United States: Historical review and current conservation status. Pgs. 201-223 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA.

Bull, J.J. 1985. Sex ratio and nest temperature in turtles: comparing field and laboratory data. Ecology 66: 1115-1122.

Burkhead N.M., Walsh S.J., Freeman B.J., Williams J.D. 1997. Status and restoration of the Etowah River, an imperiled southern Appalachian ecosystem. Pg. 375-444 In: G.W. Benz and D.E. Collins (eds). Aquatic fauna in peril: the southeastern perspective. Decatur, Georgia: Lenz Design and Communications. https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_16/index.html <Last accessed April 14, 2019>

Burkhead, N. M., and H. L. Jelks. 2001. Effects of suspended sediment on the reproductive success of the tricolor shiner, a crevice-spawning minnow. Transactions of the American Fisheries Society 130: 959-968.

Burkholder, J., B. Libra, P. Weyer, S. Heathcote, D. Kolpin, P.S. Thorne, and M. Wichman. 2007. Impacts of waste from concentrated animal feeding operations on water quality. Environmental Health Perspectives 115:308-312.

Cagle, F.R. 1952. The status of the turtles *Graptemys pulchra* Baur and *Graptemys barbouri* Carr and Marchand, with notes of their natural history. Copeia 1952:223-234.

Carr, J.L., and M. A. Messinger. 2002. *Graptemys gibbonsi* (Pascagoula Map Turtle). Predation. Herpetological Review 33:201-202.

Center for Biological Diversity. 2010. Petition to list 404 aquatic, riparian and wetland species from the southeastern United States as threatened or endangered under the endangered species act. http://www.fws.gov/cookeville/pdfs/sepetitionfinal.pdf. 1145 pp.

Center for Watershed Protection (CWP). 2003. Impacts of Impervious Cover on Aquatic Systems. Watershed Protection Research Monograph No. 1. Ellicott City, MD. 158 pp.

Cheung, S.M. and D. Dudgeon. 2006. Quantifying the Asian turtle crisis: market surveys in southern China, 2000-2003. Aquatic Conservation: Marine and Freshwater Ecosystems 16:751-770.

Chow-Fraser P., V.L. Lougheed, B. Crosbie, V. LeThiec, L. Simser, and J. Lord. 1998. Longterm response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of L. Ontario. Wetlands Ecology and Management 6:19–42.

Coles, .J.F., G. McMahon, A. H. Bell, L. R. Brown, F. A. Fitzpatrick, B. C. Eikenberry, M. D. Woodside, T. F. Cuffney, W. L. Bryant, K. Cappiella, L. Fraley-McNeal, and W.P. Stack. 2012. Effects of urban development on stream ecosystems in nine metropolitan study areas across the United States. Circular 1373. U.S. Geological Survey. 152 pp.

Congdon, J.D., D.W. Tinkle, G.L. Breitenbach, and R.C. VanLoben Sels. 1993. Delayed sexual maturity and demographics of Blanding's turtles (*Emydoidea blandingii*): Implications for conservation and management of long-lived organisms. Conservation Biology 7:826-833.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). 2019. Appendices I, II and III, valid from November 26, 2019.

Craig, M.J. 1992. Radio-telemetry and tagging study of movement patterns, activity cycles, and habitat utilization in Cagle's map turtle. Unpublished M.Sc. thesis, West Texas State University, Canyon. 82 pp.

Crandall K.A., Bininda-Emonds O.R., Mace G.M., and R.K Wayne. 2000. Considering evolutionary processes in conservation biology. Trends in Ecology & Evolution 15:290-295.

DeCatanzaro, R., and P. Chow-Fraser. 2010. Relationship of road density and marsh condition to turtle assemblage characteristics in the Laurentian Great Lakes. Journal of Great Lakes Research 36:357–365.

Deutsch, W.G., W.C. Seesock, E.C. Webber, and D.R. Bayne. 1990. The impact of poultry rearing operations on water quality and biological communities of second order streams in Cullman and Winston counties, Alabama, 1988-89. Auburn University, Department of Fisheries and Allied Aquacultures, Auburn, AL. 62 pp.

DeWan, A., N. Dubois, K. Theoharides, and J. Boshoven. 2010. Understanding the impacts of climate change on fish and wildlife in North Carolina. Defenders of Wildlife Report, Washington D.C.

Diamond, J.M., D.W. Bressler, and U.B. Serveiss. 2002. Assessing relationships between human land uses and the decline of native mussels, fish, and macroinvertebrates in the Clinch and Powell river watershed, USA. Environmental Toxicology and Chemistry 21:1147-1155.

Dickerson D.D. and K.J. Reine. 1996. Habitat assessment and relative abundance estimates for the ringed sawback turtle (*Graptemys oculifera*) in dredging sites of the West Pearl River navigation project (Louisiana). Unpublished report. U.S. Army Engineer District, Vicksburg, Mississippi. 21 pp. + tables and figures.

Dieter, C.D., L.A. Dixon, S.L. Ronningen, and T. Ronningen. 2014. Survey of turtles nesting on the Missouri River on the South Dakota-Nebraska border. Great Plains Research 24:111-118.

Dundee, H.A., and D.A. Rossman. 1989. The amphibians and reptiles of Louisiana. Louisiana State University Press, Baton Rouge, LA. pp. 186-187.

Eisemberg, C.C., R.A.M. Balestra, S. Famelli, F.F. Pereira, V.C.D. Bernardes, and R.C. Vogt. 2016. Vulnerability of Giant South American Turtle (*Podocnemis expansa*) nesting habitat to climate-change-induced alterations to fluvial cycles. Tropical Conservation Science October-December 2016:1-12.

Emanuel, K. 2005. Increasing destructiveness of tropical cyclones of the past 30 years. Nature 436:686-688.

Ennen, J.R., J.E. Lovich, and R.D. Birkhead. 2012. *Graptemys pearlensis*. Ennen, Lovich, Kreiser, Selman, and Qualls. Pearl River Map Turtle. Catalogue of American Amphibians and Reptiles:889.881-889.884.

Ennen, J.R., J.E. Lovich, and R.L. Jones. 2016. *Graptemys pearlensis* Ennen, Lovich, Selman, and Qualls 2010 – Pearl River Map Turtle. *In:* Rhodin, A.G.J., Pritchard, P.C.J., van Dijk, P.P., Saumure, R.A., Buhlmann, K.A., Iverson, J.B., and Mittermeier, R.A. (Eds.). Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group, Chelonian Research Monographs 5(9):094.1-8.

Ennen, J.R., J.E. Lovich, B.R. Kreiser, W. Selman, and C.P. Qualls. 2010. Genetic and morphological variation between populations of the Pascagoula map turtle (*Graptemys gibbonsi*) in the Pearl and Pascagoula rivers with description of a new species. Chelonian Conservation and Biology 9:98–113.

Eskew, E., S. Price, and M. Dorcas. 2010. Survival and recruitment of semi-aquatic turtles in an urbanized region. Urban Ecosystems 13:365–374.

Etnier, D.A. 1997. Jeopardized southeastern freshwater fishes. p. 87-104 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_4/index.html <Last Accessed June 2020>

Folkerts, G.W. 1997. State and fate of the world's aquatic fauna. Pgs. 1-16 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA.

Freeman, M.C., M.M. Hagler, P.M. Bumpers, K. Wheeler, S.J. Wenger, and B.J. Freeman. 2017. Long-term monitoring data provide evidence of declining species richness in a river valued for biodiversity conservation. Journal of Fish and Wildlife Management 8:418-435.

French, S. S., Fokidis, H. B., and Moore, M. C. 2008. Variation in stress and innate immunity in the tree lizard (*Urosaurus ornatus*) across an urban–rural gradient. Journal of Comparative Physiology. B,Biochemical, Systemic, and Environmental Physiology 178:997–1005.

Fuselier, L. and D. Edds. 1994. Habitat partitioning among three sympatric species of map turtles, Genus *Graptemys*. Journal of Herpetology 28:154-158.

Geller, G. A. 2012. Notes on the nesting ecology of Ouachita Map Turtles (*Graptemys ouachitensis*) at two Wisconsin sites using trail camera monitoring. Chelonian Conservation and Biology 11:206-213.

Gibbons, J.W., and J.E. Lovich. 1990. Sexual dimorphism in turtles with emphasis on the slider turtle (*Trachemys scripta*). Herpetological Monographs 4:1–29.

Goetz, S.J., R.K. Wright, A.J. Smith, E. Zinecker, and E. Schaub. 2003. IKONOS imagery for resource management: Tree cover, impervious surfaces, and riparian buffer analyses in the mid-Atlantic region. Remote Sensing of Environment 88:195-208.

Goldenberg, S. B., C. W. Landsea, A. M. Mestas-Nuñez, and W. M. Gray. 2001. The recent increase in Atlantic hurricane activity: Causes and implications. Science 293:474–479.

Greening H., P.H. Doering, and C. Corbett. 2006. Hurricane impacts on coastal ecosystems. Estuaries and Coasts 29:877-879.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman, and E.B.D. Jones III. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Sciences:14843-14847.

Hartfield, P. 1993. Headcuts and their effects on freshwater mussels. Pgs. 131-141 In: K.S. Cummings, A.C. Buchanan, and L. M. Koch (eds.). Conservation and Management of Freshwater Mussels. Proceedings of a UMRCC Symposium, St. Louis, MO. Upper Mississippi River Conservation Committee, Rock Island, IL.

Heppard, J.M. and R. Buchholz. 2019. Impact of human disturbance on the thermoregulatory behaviour of the endangered ringed sawback turtle (*Graptemys oculifera*). Aquatic Conservation: Marine and Freshwater Ecosystems 29:990‑1001.

Herrig, J. and P. Shute. 2002. Chapter 23: aquatic animals and their habitats. Southern Region, USDA Forest Service and Tennessee Valley Authority. Pgs. 45 In: D.N.Wear, J.G. Greis (eds.). Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 635 pp. Available at: www.srs.fs.usda.gov/sustain/report/pdf/chapter_23e.pdf <Last accessed July 2020>

Holcomb, Samuel R., Amity A. Bass, Christopher S. Reid, Michael A. Seymour, Nicole F. Lorenz, Beau B. Gregory, Sairah M. Javed, and Kyle F. Balkum. 2015. Louisiana Wildlife Action Plan. Louisiana Department of Wildlife and Fisheries. Baton Rouge, LA.

Horne, B.D., R.J. Brauman, M.J.C. Moore, and R.A. Seigel. 2003. Reproductive and nesting ecology of the yellow-blotched map turtle, *Graptemys flavimaculata*: Implications for Conservation and Management. Copeia, 2003:729–738.

International Panel on Climate Change (IPCC). 2007. Climate change 2007, synthesis report. In Contributions of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (eds Core Writing Team, Pachauri R. K., Reisinger A.), pp. 104. Geneva, Switzerland: IPCC.

International Panel on Climate Change (IPCC). 2013. Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY.

International Union for Conservation of Nature (IUCN). 2008. Protected area definition. Saved as pdf. Available online: https://www.iucn.org/theme/protected-areas/about <Last accessed April 8, 2021>

Iverson, J.B., P.A. Meylan, and M.E. Seidel. 2017. Testudines – Turtles, Pg. 86 In: B. I. Crother (ed.), Scientific and Standard English Names of Amphibians and Reptiles of North America North of Mexico, with Comments Regarding Confidence in Our Understanding pp. 1–102. SSAR Herpetological Circular 43.

Jones, R.L. 1996. Home Range and Seasonal Movements of the Turtle *Graptemys flavimaculata*. Journal of Herpetology 30:376-385.

Jones, R.L. 2006. Reproduction and nesting of the endangered ringed map turtle, *Graptemys oculifera*, in Mississippi. Chelonian Conservation and Biology 5:195-209.

Jones, R. L., W.T. Slack, and P.D. Hartfield. 2005. The Freshwater Mussels (Mollusca: Bivalvia: Unionidae) of Mississippi. Southeastern Naturalist 4:77-92.

Jude D.J., and J. Pappas. 1992. Fish utilization of Great Lakes coastal wetlands. J Great Lakes Res 18:651–672.

Keellings, D., and J. Engstrom. 2019. The future of drought in the Southeastern U.S.: projections from downscaled CMIPD models. Water 11:1-9.

Koehnken, L., M.S. Rintoul, M. Goichot, D. Tickner, A-C. Loftus, and M.C. Acreman. 2020. Impacts of riverine sand mining on freshwater ecosystems: A review of the scientific evidence and guidance for future research. River Research and Applications; 2020:1-9.

Lamb, T., C. Lydeard, R.B. Walker, and J.W. Gibbons. 1994. Molecular systematics of map turtles (*Graptemys*): A comparison of mitochondrial restriction site versus sequence data. Systematic Biology 43:543-559.

Lasier P.J., M.L. Urich, S.M. Hassan, W.N. Jacobs, R.B. Bringolf, and K.M. Owens. 2016. Changing agricultural practices: potential consequences to aquatic organisms. Environmental Monitoring and Assessment 188:672.

Lindeman, P.V. 1998. Of deadwood and map turtles (*Graptemys*): An analysis of species status for five species in three river drainages using replicated spotting-scope counts of basking turtles. Linnaeus Fund research report. Chelonian Conservation and Biology 3:137−141.

Lindeman, P.V. 1999. Surveys of basking map turtles *Graptemys* spp. in three river drainages and the importance of deadwood abundance. Biological Conservation 88:33−42.

Lindeman, P.V. 2000. Evolution of relative width of the head and alveolar surfaces in map turtles (Testudines: Emydidae *Graptemys*). Biological Journal of the Linnean Society 69:549-576.

Lindeman, P.V. 2013. The Map Turtle and Sawback Atlas: Ecology, Evolution, Distribution, and Conservation. University of Oklahoma Press, Norman, OK. 460 pp.

Lindeman, P.V. 2019. Population status of the megacephalic map turtles *Graptemys pearlensis* and *Graptemys gibbonsi*. Unpublished Final Report submitted to U.S. Fish and Wildlife Service, Jackson, MS. 54 pp.

Lindeman, P.V. 2020. Species status assessment expert elicitation questions. Pearl River map turtle. Provided on April 20, 2020 to the U.S. Fish and Wildlife Service, Jackson, MS. 5 pp.

Louisiana, State of. 2019. House of Representatives. FY 19-20 HB 105 Review, Culture, Recreation, and Tourism, House Committee on Appropriations. Powerpoint presentation, April 3, 2019. 26 pp.

Louisiana Department of Agriculture and Forestry (LDAF). Undated. Recommended forestry best management practices for Louisiana. Viewed: August 10, 2020. https://www.ldaf.state.la.us/forestry/management/best-management-practices-and-statistics/

LDAF. 2018. 2018 BMP survey results, Mike Strain, Commissioner, Baton Rouge, LA. Letterhead cover letter + excel datasheet. 2 pp.

Louisiana Department of Wildlife and Fisheries (LDWF). 2020a. Louisiana 2020 fishing regulations. Baton Rouge, LA. 65 pp.

LDWF. 2020b. Rare species and natural communities by parish. Rare animals Track List (pdf). 7pp. Website viewed 8/12/2020: https://www.wlf.louisiana.gov/page/rare-species-and-natural-communities-by-parish

LDWF and The Conservation Fund. 2014. Master Plan for Wildlife Management Areas and Refuges. Baton Rouge, LA. 84 pp.

Lovich, J.E. 2009. *Graptemys gibbonsi* Lovich and McCoy, 1992. Pascagoula map turtle. Pgs. 303-306. In: C.H. Ernest and J.E. Lovich (eds.). Turtles of the United States and Canada. The Johns Hopkins University Press, Baltimore, MD.

Lovich, J.E., and J.R. Ennen. 2013. A quantitative analysis of the state of knowledge of turtles of the United States and Canada. Amphibia-Reptilia 34:11–23.

Lovich, J.E., S.W. Gotte, C.H. Ernst, J.C. Harshburger, A.F. Laemmerzahl, and J.W. Gibbons. 1996. Prevalence and histopathology of shell disease in turtles from Blackshear, Georgia. Journal of Wildlife Diseases 32:259-265.

Lovich, J.E., and C.J. McCoy. 1992. Review of the *Graptemys pulchra* group (Reptilia: Testudines: Emydidae), with descriptions of two new species. Annals of the Carnegie Museum 61:293−315.

Lovich, J.E., M. Quillman, B. Zitt, A. Schroeder, D.E., Green, C. Yackulic, P. Gibbons, and E. Goode. 2017. The effects of drought and fire in the extirpation of an abundant semi-aquatic turtle from a lacustrine environment in the southwestern USA. Knowledge and Management of Aquatic Ecosystems 418:1-11.

Lovich, J.E., W. Selman, and C.J. McCoy. 2009. *Graptemys gibbonsi* Lovich and McCoy 1992. Pascagoula Map Turtle, Pearl River Map Turtle, Gibbon's Map Turtle. Chelonian Research Monographs 5:029.1-029.8.

Luiselli, L., A. Starita, G.M. Carpaneto, G.H. Segniagbeto, and G. Amori. 2016. A short review of the international trade of wild tortoises and freshwater turtles across the world and throughout two decades. Chelonian Conservation and Biology 15:167-172.

Mallin, M.A., and C. A. Corbett. 2006. How Hurricane Attributes Determine the Extent of Environmental Effects: Multiple Hurricanes and Different Coastal Systems. Estuaries and Coasts 29:1046–1061.

Mangal M.and C. Tier. 1993. A simple direct method for finding persistence times of populations and application to conservation problems. Proceedings of the National Academy of Sciences of the USA 90:1083-1086.

Marion, K. R. 1986. Alabama map turtle *Graptemys pulchra* Baur. Pgs. 50-52 In: R.H. Mount (ed.). Vertebrate animals of Alabama in need of special conservation. Alabama Agricultural Experiment Station, Auburn University, AL.

McCoy, C.J. and R.C. Vogt. 1979. Distribution and population status of the ringed sawback (*Graptemys oculifera*), blotched sawback (*Graptemys flavimaculata*), and black-knobbed sawback (Graptemys nigrinoda) in Alabama and Mississippi.

Medalie L, N.T. Baker, M.E. Shoda, W.W. Stone, M.T. Meyer, E.G Stets and M. Wilson. 2020. Influence of land use and region on glyphosate and aminomethylphosphonic acid in streams in the USA. Science of the Total Environment 707:136008. [open access] https://doi.org/10.1016/j.scitotenv.2019.136008

Mesnage R, N. Defarge, ,J. Spiroux de Vendômois, and G.E. Séralini. 2015. Potential toxic effects of glyphosate and its commercial formulations below regulatory limits. Food and Chemical Toxicology 84:133-153.

Mississippi Commission on Wildlife, Fisheries, and Parks, Mississippi Department of Wildlife, Fisheries, and Parks. 2016. Title 40: Wildlife, Fisheries, and Parks; Part 5: Museum of

Natural Science; Part 5, Chapter 2: Regulations regarding non-game and endangered species; Rule 2.3 Regulations regarding non-game wildlife in need of management. Source: MISS. CODE ANN. §§ 49-1-29, 49-4-4, 49-1-41, and 49-5-107. 14 pp.

Mississippi Department of Wildlife, Fisheries, and Parks. 2020. Wildlife and Hunting. Mississippi Wildlife Management Areas – Something for everyone? Website viewed 8/12/2020: https://www.mdwfp.com/wildlife-hunting/wma/mississippi-wildlife-management-areas-something-for-everyone/

Mississippi Forestry Commission (MFC). 2008. Mississippi's Best Management Practices Fourth Edition, MFC Publication # 107. Jackson, Mississippi. 52 pp.

MFC. 2019. Mississippi's voluntary silvicultural best management practices (BMPs) implementation monitoring program. 2019 BMP implementation survey for Mississippi. MFC, Jackson, Mississippi. 25 pp.

Mississippi Museum of Natural Science. 2015. Mississippi State Wildlife Action Plan. Mississippi Department of Wildlife, Fisheries, and Parks, Mississippi Museum of Natural Science, Jackson, Mississippi. 692 pp.

Mississippi Natural Heritage Program. 2015. Listed species of Mississippi. Museum of Natural Science, Mississippi Department of Wildlife, Fisheries, and Parks, Jackson, Mississippi. 3pp.

Montag, L.F.A., K.O. Winemiller, F.W. Keppeler, H. Leão, N.L. Benone, N.R. Torres, B.S. Prudente, T.O. Begot, L.M. Bower, D.E. Saenz, E.O. Lopez-Delgado, Y. Quintana, D.J. Hoeinghaus, and L. Juen. 2019. Land cover, riparian zones and instream habitat influence stream fish assemblages in the eastern Amazon. Ecology of Freshwater Fish 28:1-13.

Moore, M. V., M. L. Pace, J. R. Mather, P. S. Murdoch, R. W. Howarth, C. L. Folt, C. Y. Chen, H. F. Hemond, P. A. Flebbe, and C. T. Driscoll. 1997. Potential effects of climate change on freshwater ecosystems of the New England/Mid-Atlantic Coastal Region, Climate Research. 14:161-173.

Morse, J.C., B.P. Stark, W. P. McCafferty, and K.J. Tennessen. 1997. Southern Appalachian and other Southeastern streams at risk: Implications for mayflies, dragonflies and damselflies, stoneflies, and caddisflies. Pgs. 17-42 In: Benz, G.W. and D.E. Collins (editors). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA.

https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_2/index.html<Last Accessed June 2020>

Mulholland, P.J. and D.R. Lenat. 1992. Streams of the southeastern piedmont, Atlantic drainage. In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY, p. 193-231.

Multi-Resolution Land Characteristics Consortium (MRLC). 2016. National Land Cover Dataset. https://www.mrlc.gov/data/nlcd-2016-land-cover-conus <Last Accessed July 2020>

Neves, R.J., A.E. Bogan, J.D. Williams, S.A. Ahlstedt, P.W. Hartfield. 1997. Status of aquatic mollusks in the southeastern United States: a downward spiral of diversity. Pgs. 43-85 In: Benz, G.W. and D.E. Collins (editors). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. Available online: http://www.sherpaguides.com/southeast/aquatic_fauna/chapter_3/ <Last accessed July 2020>

Nicotra, A.B., Beever E.A., Robertson A.L., Hofmann G.E., and J. O'Leary. 2015. Assessing the components of adaptive capacity to improve conservation and management efforts under global change. Conservation Biology 29:1268-1278.

National Oceanic and Atmospheric Administration (NOAA). 2020. Sea Level Rise Viewer. Available at https://coast.noaa.gov/slr/ (Accessed: August 17, 2020).

North Carolina Aquatic Nuisance Species Management Plan Committee (NCANSMPC). 2015. North Carolina Aquatic Nuisance Species Management Plan. Raleigh, NC. 96 pp.

Overpeck, J. T., B. L. Otto-Bliesner, G. H. Miller, D. R. Muhs, R. B. Alley, and J. T. Kiehl. 2006. Paleoclimatic evidence for future ice-sheet instability and rapid sea-level rise. Science 311:1747–1750.

Patrick, D. A., and Gibbs, J. P. 2010. Population structure and movements of freshwater turtles across a road-density gradient. Landscape Ecology 25:791–801.

Pearl River Water Management Supply District. 2020. Barnett Reservoir. Webpage viewed on September 15, 2020: https://www.therez.ms.gov/Pages/About.aspx

Pearson, L., W. Selman, B. Kreiser, P.V. Lindeman, and R.L. Jones. 2020. Conservation genetics of two imperiled *Graptemys* species (*G. gibbonsi and G. pearlensis*) petitioned for federal listing. Unpublished Report 48 pp.

Petersen, J. C., J. C. Adamski, R. W. Bell, J. V. Davis, S. R. Femmer, D. A. Freiwald, and R. L. Joseph. 1999. Water quality in the Ozark Plateaus, Arkansas, Kansas, Missouri, and Oklahoma. U.S. Geological Survey Circular 1158.

Pluto, T.G. and E.D. Bellis. 1986. Utilization by the Turtle, *Graptemys geographica*, along a River. Journal of Herpetology 20:22-31.

Poff, N.L. 1992. Regional Hydrologic Response to Climate Change: An Ecological Perspective. In Global Climate Change and Freshwater Ecosystems. P. Firth and S.G. Fisher, eds. Springer-Verlag, New York, pp. 88-115.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime, BioScience. 47:769–784.

Power, M. E., A. Sun, G. Parker, W. E. Dietrich, and J. T. Wootton. 1995. Hydraulic food-chain models. BioScience. 456:159–167.

Praschag, P., F. Ihlow, M. Flecks, M. Vamberger, and U. Fritz. 2017. Diversity of North American map and sawback turtles (Testudines: Emydidae: *Graptemys)*. Zoologica Scripta 46:675-682.

Rahmstorf, S., Cazenave, A., Church, J.A., Hansen, J.E., Keeling, R.F., Parker, D.E., Somerville, R.C. 2007. Recent climate observations compared to projections. Science 316:709.

Redford, K.H., G. Amato, J. Baillie, P. Beldomenico, E.L. Bennett, N. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G.M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to successfully conserve a (vertebrate) species. BioScience. 61:39-48.

Resh, V. H., A. V. Brown, A. P. Covich, M. E. Gurtz, H. W. Li, G. W. Minshall, S. R. Reice, A. L. Sheldon, J. B. Wallace, and R. Wissmar. 1988. The role of disturbance in stream ecology. J. N. Am. Benthol. Soc. 7:433–455.

Rignot, E., and P. Kanagaratnam. 2006. Changes in the velocity structure of the Greenland ice sheet. Science 311:986 – 990.

Riley, S. P. D., Busteed, G. T., Kats, L. B., Vandergon, T. L., Lee, L. F. S.,Dagit, R. G., Kerby, J. L., Fisher, R. N., and Sauvajot, R. M. 2005. Effects of urbanization on the distribution and abundance of amphibians and invasive species in southern California streams. Conservation Biology 19:1894–1907.

Ringler, N.H. and J.D. Hall. 1975. Effects of harvesting on water temperature and dissolved oxygen in spawning beds. Transactions of the American Fisheries Society. 104:111–121.

Roy, A.H., M.C. Freeman, B.J. Freeman, S.J. Wenger, W.E. Ensign, and J.L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of North American Benthological Society 24:656-678.

Schlaepfer, M. A., Hoover, C. & Dodd, C. K. 2005. Challenges in evaluating the impact of the trade in amphibians and reptiles on wild populations. BioScience 55: 256.

Schofield, K. A., C. M. Pringle, and J. L. Meyer. 2004. Effects of increased bedload on algaland detrital-based stream food webs: experimental manipulation of sediment and macroconsumers. Limnology and Oceanography 49: 900-909.

Schueler, T.R. 1994. The importance of imperviousness. Watershed Protection Techniques 1(3):100-111.

Schueler, T.R., L. Fraley-McNeal, and K. Cappiella. 2009. Is impervious cover still important? Review of recent research. Journal of Hydrologic Engineering 14:309-315.

Schuster, G.A. 1997. Crustacean Resource Management in the Southeast. Pgs. 269-282 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_11/index.html <Last Accessed June 2020>

Selman, W. 2020a. Species status assessment expert elicitation questions. Pearl River map turtle. Provided on May 11, 2020 to the U.S. Fish and Wildlife Service, Jackson, MS. 8 pp.

Selman, W. 2020b. Comments provided May 27, 2020 on Draft Species Assessment Report for the Pearl River map turtle by Dr. Will Selman, Millsaps College, Jackson, Mississippi, to U.S. Fish and Wildlife Service, Jackson, MS. pg. 16

Selman, W. 2020c Distribution and abundance of three sympatric SGCN turtle species in the Pearl River System, Louisiana. Yearly Interim Report. A State Wildlife Grant Report to Louisiana Department of Wildlife and Fisheries, Baton Rouge, Louisiana. SWG 2000467758. 44 pp.

Selman, W. 2020d. River turtles and one dam lake: Two imperiled *Graptemys* species in the Pearl River and potential impacts of the proposed One Lake Project. Chelonian Conservation and Biology 19:186-196.

Selman, W. and R.L. Jones. 2017. Population structure, status, and conservation of two *Graptemys* species from the Pearl River, Mississippi. Journal of Herpetology 51:27-36.

Selman, W. and P.V. Lindeman. 2015. Life history and ecology of the Pascagoula map turtle (*Graptemys gibbonsi*). Herpetological Conservation and Biology 10:781-800.

Selman, W. and C. Qualls. 2007. Distribution, status, and conservation of the Pascagoula map turtle (*Graptemys gibbonsi*). Unpublished report submitted to U.S. Fish and Wildlife Service, Jackson, MS. 37 pp.

Selman, W. and C. Qualls. 2008. The impacts of Hurricane Katrina on a Population of yellow-blotched sawbacks (*Graptemys flavimaculata*) in the lower Pascagoula River. Herpetological Conservation and Biology 3:224-230.

Selman, W., C. Qualls, and M. Mendonca. 2009. Assessment of the impact of Hurricane Katrina on the yellow-blotched sawback (*Graptemys flavimaculata*) Year 2. Unpublished report submitted to the U.S. Fish and Wildlife Service and Mississippi Museum of Natural Science, Jackson, Mississippi. 62 pp.

Selman, W., C. Qualls, and J.C. Owen. 2013. Effects of Human Disturbance on the Behavior and Physiology of an Imperiled Freshwater Turtle. Journal of Wildlife Management 77:877-885.

Sgro C.M., Lowe A.J., Hoffmann A.A. 2011. Building evolutionary resilience for conserving biodiversity under climate change. Evolutionary Applications 4:326-337.

Shaffer M.L. and B.A. Stein. 2000. Safeguarding our precious heritage. Pgs. 301-321 In: Stein B.A., L.S. Kutner, and J.S. Adams (eds.). Precious heritage: the status of biodiversity in the United States. Oxford University Press, New York, NY.

Shively, S.H. 1999. Survey of the ringed map turtle (*Graptemys oculifera*) in the Bogue Chitto River, Louisiana. Unpublished report submitted to the Louisiana Natural Heritage Program, Baton Rouge, Louisiana.

Shute, P.W., R.G. Biggins, and R.S. Butler. 1997. Management and Conservation of Rare Aquatic Resources: A Historical Perspective and Recommendations for Incorporating Ecosystem Management. Pgs. 445-466 In: Benz, G.W. and D.E. Collins (eds). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_17/index.html <Last Accessed June 2020>

Smith D.R., N.L. Allan, C.P. McGowan, J.A. Szymanski, S.R. Oetker, and H.M. Bell. 2018. Development of a species status assessment process for decisions under the U.S. Endangered Species Act. Journal of Fish and Wildlife Management 9:1–19.

Soballe, D.M., B.L. Kimmel, R.H. Kennedy, and R.F. Gaugush. 1992. Reservoirs. Pgs. 421-474 In: Biodiversity of the Southeastern United States: Aquatic Communities. C.T. Hackney, S.M. Adams, and W.H. Martin (eds.). John Wiley and Sons, New York, NY.

Speer, P., H.G. Golden, and J.F. Patterson. 1964. Low-flow characteristics of streams in the Miss. Embayment in Miss. and Ala. Washington: Government Printing Office. p. I-8.

Spielman, D., B.W. Brook, and R. Frankham. 2004. Most species are not driven to extinction before genetic factors impact them. PNAS 101:15261-15264.

Sterrett, S.C. 2009. The ecology and influences of land use on river turtles in Southwest Georgia. Unpublished M.S. thesis, University of Georgia, Athens, GA. 103 pp.

Sterrett, S.C., L. L. Smith, S. W. Golladay, S. H. Schweitzer, and J. C. Maerz. 2011. The conservation implications of riparian land use on river turtles. Animal Conservation 14:38–46.

Sung, Yik-Hei and J.J. Fong. 2018. Assessing consumer trends and illegal activity by monitoring the online wildlife trade. Biological Conservation 227:219-225.

Suski, D.C. and S.J. Cooke. 2007. Conservation of aquatic resources through the use of freshwater protected areas: Opportunities and challenges. Biodiversity and Conservation 16:2015-2029.

Sutherland A.B., and J.L. Meyer. 2007. Effects of increased suspended sediment on growth rate and gill condition of two southern Appalachian minnows. Environmental Biology of Fishes 80:389−403.

Sweka, J. A., and K. J. Hartman. 2003. Reduction of reaction distance and foraging success in smallmouth bass, *Micropterus dolomieu*, exposed to elevated turbidity levels. Environmental Biology of Fishes 67:341-347.

Thomson, R.C., P.Q. Spinks, and H.B. Shaffer. 2018. Molecular phylogeny and divergence of the map turtles (Emydidae: *Graptemys*). Molecular Phylogenetics and Evolution 121:61-70.

U.S. Environmental Protection Agency (EPA). 2005. Pilot TMDL applications using the impervious cover model. Project # 10598-001-002, submitted by ENSR Corporation, Westford, MA. 81 pp.

U.S. Fish and Wildlife Service (USFWS). 1986. Endangered and threatened wildlife and plants: Threatened status for the ringed map turtle. Federal Register 51: 45907-45910.

USFWS. 1990. Endangered and threatened wildlife and plants: Threatened status for the inflated hell splitter. Federal Register 55: 39868-39872.

USFWS. 1991. Endangered and threatened wildlife and plants: Threatened status for the Gulf sturgeon. Federal Register 56: 49653-49658.

USFWS. 2000. Mobile River Basin Aquatic Ecosystem Recovery Plan. Atlanta, GA. 128 pp. Available online: ecos.fws.gov/docs/recovery_plans/2000/001117.pdf <Last accessed June 14, 2019>

USFWS. 2003. Endangered and threatened wildlife and plants: Designation of critical habitat for the Gulf sturgeon. Federal Register 68: 13370-13495.

USFWS. 2005. Final Rule. Inclusion of Alligator Snapping Turtle (*Macroclemys* [=*Macrochelys*] *temminckii*) and All Species of Map Turtle (*Graptemys* spp.) in Appendix III to the Convention on International Trade in Endangered Species of Wild Fauna and Flora. Federal Register 70: 74700-74712.

USFWS. 2006. Status assessment. Pascagoula map turtle – *Graptemys gibbonsi*. Unpublished report, Mississippi Ecological Services Field Office, Jackson, Mississippi. 5 pp.

USFWS. 2010. Ringed map turtle *(Graptemys oculifera)*. 5-Year Review: Summary and Evaluation. Southeast Region, Mississippi Ecological Services Field Office, Jackson, Mississippi. 15 pp.

USFWS. 2011. Bogue Chitto National Wildlife Refuge Comprehensive Conservation Plan. Southeast Region, Atlanta, Georgia. 88 pp. + appendices.

USFWS. 2016. USFWS species status assessment framework: an integrated analytical framework for conservation. Version 3.4. dated August 2016.

U.S. Forest Service. 2014. Land and Resource Management Plan. National Forests in Mississippi. USDA Forest Service R8-MB 144A. Jackson, Mississippi. 202 pp.

U.S. Geological Survey (USGS). 2017. Land-Cover Modelling Methodology-The FORE-SCE Model. https://www.usgs.gov/core-science-systems/eros/lulc/land-cover-modeling-methodology-fore-sce-model?qt-science_support_page_related_con=4#qt-science_support_page_related_con <Last Accessed July 2020>

USGS. 2019a. National Hydrography Dataset (ver. USGS National Hydrography Dataset Best Resolution (NHD) for Hydrologic Unit (HU) 4 - 2001 (published 20191002)), https://www.usgs.gov/core-science-systems/ngp/national-hydrography/access-national-hydrography-products <Last Accessed June 2020>

USGS. 2019b. Protected Areas Database. https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/pad-us-data-download?qt-science_center_objects=0#qt-science_center_objects <Last Accessed July 2020>

U.S. Navy. 2011. Integrated Natural Resources Management Plan. Naval Construction Battalion Center Gulfport, Stennis Western Maneuver Area, Stennis Space Center, Mississippi.

Valenzuela, N., R. Literman, J.L. Neuwald, and B. Mizoguchi. 2019. Extreme thermal fluctuations from climate change unexpectedly accelerate demographic collapse of vertebrates with temperature dependent sex determination. Scientific Reports 9:4254 [open access] https://doi.org/10.1038/s41598-019-40597-4

Vogt, R.C. and J.J. Bull. 1982. Temperature controlled sex-determination in turtles: ecological and behavioral aspects. Herpetologica 38: 156-164.

Vogt, R. C. and J. J. Bull. 1984. Ecology of hatchling sex ratio in map turtles. Ecology 65: 582-587.

Vogt, R.C., C.J. McCoy, and M. Pappas. 2019. *Graptemys pearlensis* (Pearl River Map Turtle) Reproduction. Herpetological Review 50:557-558.

Vučenović, J. and P. V. Lindeman. 2020. The diets of the Pearl and Pascagoula Map Turtles (*Graptemys pearlensis* and *Graptemys gibbonsi*).

Walsh, C. J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706–723.

Wang, L., J. Lyons, and P. Kanehl. 2001. Impacts of urbanization on streams habitat and fish across multiple spatial scales. Environmental Management 28:255-266.

Warren, Jr., M.L., P.L. Angermeier, B.M. Burr, and W.R. Haag. 1997. Patterns of fish imperilment in the Southeast. Pgs. 105-164 In: Benz, G.W. and D.E. Collins (eds.). Aquatic Fauna in Peril: The Southeastern Perspective. Southeast Aquatic Research Institute Special Publication 1, Lenz Design and Communications, Decatur, GA. https://www.sherpaguides.com/southeast/aquatic_fauna/chapter_5/index.html <Last Accessed July 2020>

Webster, P.J., G.J. Holland, J.A. Curry, and H.R. Chang. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. Science 309:1844-1846.

Weisberg, R.H. and L. Zheng. 2006. A Simulation of the Hurricane Charley Storm Surge and Its Breach of North Captiva Island. Florida Scientist 69:152-165.

Wenger, S.J., and M.C., Freeman. 2007. Stressors to Imperiled Fishes in the Etowah Basin Mechanisms, Sources and Management under the Etowah HCP. Unpublished document. 42 pp.

Wenger, S.J., J.T. Peterson, M.C. Freeman, B.J. Freeman, and D.D. Homans. 2008. Streams fish occurrence in response to impervious cover, historic land use, and hydrogeomorphic factors. Canadian Journal of Fisheries and Aquatic Sciences 65:1250-1264.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of the freshwater mussels of the United States and Canada. Fisheries 18:6–22.

Winston, M.R., C.M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. Transactions of the American Fisheries Society 120:98-105.

Wirth W., L. Schwarzkopf, L.F. Skerratt, and E. Ariel. 2018. Ranaviruses and reptiles. PeerJ 6:e6083 https://doi.org/10.7717/peerj.6083

Wolf. S., B. Hartl, C. Carroll, M.C. Neel, and D.N. Greenwald. 2015. Beyond PVA: Why recovery under the Endangered Species Act is more than population viability. BioScience 65:200-207.

Woodburn, D.B., A.N. Miller, M.C. Allender, C.W. Maddox, K.A. Terioe. 2019. *Emydomyces testavorans*, a new genus and species of onygenalean fungus isolated from shell lesions of freshwater aquatic turtles. Journal of Clinical Microbiology Volume 57, Issue 2 e00628-18. Available online: http://jcm.asm.org

Zackay, A. 2007. Random genetic drift and gene fixation. [online] https://www.metabolic-economics.de/pages/seminar_theoretische_biologie_2007/ausarbeitungen/zackay.pdf [Accessed 21 June 2018] 7pp.

APPENDIX A – LAND USE SUMMARIES FOR PEARL RIVER RESILIENCE UNITS

Table 1-A) Summary of current development across the Bogue Chitto unit and within riparian areas. B) Summary of current agriculture across the Bogue Chitto unit and within riparian areas. Streams in bold are mainstem reaches; all other streams are considered tributaries.

A)

Stream	Total Acres	Current Acres Developed	% Developed Current	Total Acres Riparian	Current Acres Developed Riparian	% Developed Riparian
Bogue Chitto	773543.88	51073.79	6.60	6074.26	60.94	1.00
Magees	773543.88	51073.79	6.60	1834.75	25.80	1.41
Topisaw	773543.88	51073.79	6.60	1306.12	18.90	1.45

B)

Stream	Total Acres	Current Acres Agriculture	% Agriculture Current	Total Acres Riparian	Current Acres Agriculture Riparian	% Agriculture Riparian
Bogue Chitto	773543.88	175388.53	22.67	6074.26	92.74	1.53
Magees	773543.88	175388.53	22.67	1834.75	69.39	3.78
Topisaw	773543.88	175388.53	22.67	1306.12	185.48	14.20

Table 2-A) Summary of current development across the Lower Pearl unit and within riparian areas. B) Summary of current agriculture across the Lower Pearl unit and within riparian areas. Streams in bold are mainstem reaches; all other streams are considered tributaries.

A)

Stream	Total Acres	Current Acres Developed	% Developed Current	Total Acres Riparian	Current Acres Developed Riparian	% Developed Riparian
East Pearl	1165606.50	77706.64	6.67	1987.98	10.67	0.54
Hobolochitto	1165606.50	77706.64	6.67	291.34	2.22	0.76
Holmes Bayou	1165606.50	77706.64	6.67	204.38	0.00	0.00
Lower Pearl	1165606.50	77706.64	6.67	5920.59	33.14	0.56

Navigation Canal	1165606.50	77706.64	6.67	84.07	0.44	0.53
Pushepatapa	1165606.50	77706.64	6.67	1210.94	19.35	1.60
West Hobolochitto	1165606.50	77706.64	6.67	2456.35	24.91	1.01
West Pearl	1165606.50	77706.64	6.67	1868.34	12.68	0.68

B)

Stream	Total Acres	Current Acres Agriculture	% Agriculture Current	Total Acres Riparian	Current Acres Agriculture Riparian	% Agriculture Riparian
East Pearl	1165606.50	144723.89	12.42	1987.98	4.45	0.22
Hobolochitto	1165606.50	144723.89	12.42	291.34	0.00	0.00
Holmes Bayou	1165606.50	144723.89	12.42	204.38	0.00	0.00
Lower Pearl	1165606.50	144723.89	12.42	5920.59	5.56	0.09
Navigation Canal	1165606.50	144723.89	12.42	84.07	0.00	0.00
Pushepatapa	1165606.50	144723.89	12.42	1210.94	8.90	0.73
West Hobolochitto	1165606.50	144723.89	12.42	2456.35	8.23	0.33
West Pearl	1165606.50	144723.89	12.42	1868.34	0.00	0.00

Table 3-A) Summary of current development across the Middle Pearl Silver unit and within riparian areas. B) Summary of current agriculture across the Middle Pearl Silver unit and within riparian areas. Streams in bold are mainstem reaches; all other streams are considered tributaries.

A)

Stream	Total Acres	Current Acres Developed	% Developed Current	Total Acres Riparian	Current Acres Developed Riparian	% Developed Riparian
Middle Pearl Silver	779923.50	37647.61	4.83	3798.50	15.79	0.42

B)

Stream	Total Acres	Current Acres Agriculture	% Agriculture Current	Total Acres Riparian	Current Acres Agriculture Riparian	% Agriculture Riparian
Middle Pearl Silver	779923.50	117150.98	15.02	3798.50	81.17	2.14

Table 4-A) Summary of current development across the Middle Pearl Strong unit and within riparian areas. B) Summary of current agriculture across the Middle Pearl Strong unit and within riparian areas. Streams in bold are mainstem reaches; all other streams are considered tributaries.

A)

Stream	Total Acres	Current Acres Developed	% Developed Current	Total Acres Riparian	Current Acres Developed Riparian	% Developed Riparian
Middle Pearl Strong	1265209.88	152007.75	12.01	5612.35	61.16	1.09
Pelahatchie	1265209.88	152007.75	12.01	2114.75	23.80	1.13
Purvis	1265209.88	152007.75	12.01	825.53	6.00	0.73
Strong	1265209.88	152007.75	12.01	4351.15	29.36	0.67

B)

Stream	Total Acres	Current Acres Agriculture	% Agriculture Current	Total Acres Riparian	Current Acres Agriculture Riparian	% Agriculture Riparian
Middle Pearl Strong	1265209.88	202578.92	16.01	5612.35	219.50	3.91
Pelahatchie	1265209.88	202578.92	16.01	2114.75	34.25	1.62
Purvis	1265209.88	202578.92	16.01	825.53	89.18	10.80
Strong	1265209.88	202578.92	16.01	4351.15	142.11	3.27

Table 5-A) Summary of current development across the Upper Pearl unit and within riparian areas. B) Summary of current agriculture across the Upper Pearl unit and within riparian areas. Streams in bold are mainstem reaches; all other streams are considered tributaries.

A)

Stream	Total Acres	Current Acres Developed	% Developed Current	Total Acres Riparian	Current Acres Developed Riparian	% Developed Riparian
Lobutcha	1576498.75	81862.08	5.19	2921.60	21.57	0.74
Tuscolometa	1576498.75	81862.08	5.19	1611.47	10.90	0.68
Upper Pearl	1576498.75	81862.08	5.19	3660.61	23.80	0.65
Yockanookany	1576498.75	81862.08	5.19	3655.28	20.68	0.57

B)

Stream	Total Acres	Current Acres Agriculture	% Agriculture Current	Total Acres Riparian	Current Acres Agriculture Riparian	% Agriculture Riparian
Lobutcha	1576498.75	263492.78	16.71	2921.60	28.02	0.96
Tuscolometa	1576498.75	263492.78	16.71	1611.47	28.91	1.79
Upper Pearl	1576498.75	263492.78	16.71	3660.61	18.68	0.51
Yockanookany	1576498.75	263492.78	16.71	3655.28	129.43	3.54

APPENDIX B – USFWS LEMIS REPORT

U.S. Fish and Wildlife Service Law Enforcement Management Information System (LEMIS) *Graptemys* report for exports from the United States from 2005 to 2019.

Year	Genus	Species	Specific Name	Generic Name	Wildlife Desc	Quant	Ctry IER	IE	Year	Quant
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	150	AR	E	2005	150
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,500	AU	E	2005	1,500
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	11,700	BE	E	2005	11,700
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,100	CH	E	2005	1,100
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	100	CN	E	2005	100
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	19,400	CZ	E	2005	19,400
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	12,176	DE	E	2005	12,176
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	56	DK	E	2005	56
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	35,158	ES	E	2005	35,158
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,000	FR	E	2005	2,000
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,374	GB	E	2005	2,374
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,380	HK	E	2005	1,380
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	6,705	HU	E	2005	6,705
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	22,517	IT	E	2005	22,517
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,699	JP	E	2005	4,699
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	9,275	KR	E	2005	9,275
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	25	MO	E	2005	25
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	14,531	MX	E	2005	14,531
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	50	MY	E	2005	50
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,550	NL	E	2005	4,550
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	6,800	PL	E	2005	6,800
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	25,350	PT	E	2005	25,350
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,009	TW	E	2005	1,009
2005	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,000	UA	E	2005	1,000
2006	GRAPTEMYS	GIBBONSI	PASCAGOULA MAP	TURTLE	LIV	4	DE	E	2006	4
2006	GRAPTEMYS	GIBBONSI	PASCAGOULA MAP	TURTLE	LIV	143	JP	E	2006	143
2006	GRAPTEMYS	GIBBONSI	PASCAGOULA MAP	TURTLE	LIV	31	TW	E	2006	31
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	250	AR	E	2006	250
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	500	BE	E	2006	500
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	17	CH	E	2006	17
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,450	DE	E	2006	2,450
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,000	ES	E	2006	4,000
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	500	GB	E	2006	500
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,000	HK	E	2006	1,000
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,500	HU	E	2006	2,500
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	25	ID	E	2006	25
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,500	IT	E	2006	1,500
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,435	JP	E	2006	1,435
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,200	KR	E	2006	4,200
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,575	MX	E	2006	3,575

Year	Genus	Species	Specific Name	Generic Name	Wildlife Desc	Quant	Ctry IER	IE	Year	Quant
2006	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	100	MY	E	2006	100
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	450	BE	E	2007	450
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,900	CZ	E	2007	1,900
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,000	ES	E	2007	3,000
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	350	GB	E	2007	350
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,000	HR	E	2007	2,000
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,000	HU	E	2007	2,000
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,000	IT	E	2007	4,000
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,000	KR	E	2007	1,000
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,300	MX	E	2007	3,300
2007	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	18,700	PT	E	2007	18,700
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,500	BE	E	2008	1,500
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,000	CZ	E	2008	3,000
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	2,100	DE	E	2008	2,100
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	6,050	ES	E	2008	6,050
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,450	GB	E	2008	3,450
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	500	HK	E	2008	500
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,900	HU	E	2008	1,900
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	12,000	IT	E	2008	12,000
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	4,600	MX	E	2008	4,600
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	20,000	PT	E	2008	20,000
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	1,000	RO	E	2008	1,000
2008	GRAPTEMYS	SPECIES	MAP	TURTLE	LIV	3,000	TW	E	2008	3,000
2013	GRAPTEMYS	GIBBONSI	PASCAGOULA MAP	TURTLE	SPE	2	CA	E	2013	2
2013	GRAPTEMYS	SPECIES	MAP	TURTLE	SPR	1	CA	E	2013	1
2018	GRAPTEMYS	PEARLENSIS	PEARL RIVER MAP	TURTLE	LIV	6	AT	E	2018	6
2019	GRAPTEMYS	SPECIES	MAP	TURTLE	SPR	1	FR	E	2019	1

Wildlife Desc- Wildlife Description

LIV- Live specimens (live animals or plants)

SPE- Specimen (scientific or museum)

SPR- Shell product (mollusc or turtle)

Ctry IER-Country Code

AR-Argentina
AT-Austria
AU-Australia
BE-Belgium
CA-Canada
CH-Switzerland
CN-China
CZ-Czech Republic
DE-Germany
DK-Denmark
ES-Spain
FR-France
GB-England
HK-Hong Kong
HR-Croatia
HU-Hungary
ID-Indonesia
IT-Italy
JP-Japan
KR-Republic of Korea (South)
MO-Macao
MX-Mexico
MY-Malaysia
NL-Netherlands
PL-Poland
PT-Portugal
RO-Romania
TW-Taiwan (Province of China)
UA-Ukraine

Notes:

Species Status Assessment Report
for the
Alligator Snapping Turtle
(*Macrochelys temminckii*)

Version 1.2

Alligator Snapping Turtle
Photo credit: Kevin Enge

March 2021
U.S. Fish and Wildlife Service
Southeast Region
Atlanta, GA

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We appreciate their input and comments, which resulted in a more robust status assessment and final report.

Suggested reference:

U.S. Fish and Wildlife Service. 2021. Species status assessment report for the alligator snapping turtle (*Macrochelys temminckii*), Version 1.2. March 2021. Atlanta, GA.

SUMMARY OF VERSION UPDATES

Differences between Version 1.1 (September 2020) and Version 1.2 (March 2021) of the alligator snapping turtle Species Status Assessment (SSA) report are minor. Below we briefly summarize specific reports and updates resulting from that new information are incorporated as appropriate in Version 1.2.

Carr et al. (2020, entire) selected sampling sites across six states within the range of the two alligator snapping turtle species. There were 183 trapping sessions that resulted in the capture of 2500 turtles, of which 509 were alligator snapping turtles, either *M. temminckii* or *M. suwanniensis* (Carr et al. 2002, Table 1). The number of turtles captured across states varied from 4 to 300 in each state surveyed. Catch per unit effort (# of turtles/trap-night [t-n]) was calculated for trapping sessions to allow for comparisons by water body and stream basin (Carr et al. 2020, p. 5). This value ranged from .0979 AST/trap-night in Georgia to .2044 AST/trap-night in Alabama.

The Louisiana Department of Wildlife and Fisheries provided a courtesy draft of their Louisiana Turtle Conservation Plan to the Service, which contained past and recent survey information as well as helpful information related to price/hatchling. The report also provided a description of how best to use and interpret Catch per Unit Effort (CPUE) data for turtles.

Johnson 2020 (entire) conducted an occupancy analysis of alligator snapping turtle and factors that influence occupancy in northeast Louisiana. The study found a balanced sex ratio of 1:1, but adult to juvenile ratio of 7:1, which may be related to low nest success. Another thesis completed by Shook (2020) identified potential human pressures and analyzed the relationship between maternal size and reproductive output in northeast Louisiana. It also identified a few additional threats including gunshot and road and railway crossings.

The new information gleaned from these studies and Kessler (2020; discussed in the modeling section of the SSA) did not alter our model approach, but did provide additional detail and in some cases helped us validate some model parameters.

A list of the updates to the analysis is provided below:

1. Additional detail on survey efforts completed since the last version and their methodologies (summarized above).
2. Additional explanation about the modeling effort added to Section 5.1.3.
3. Figures, Tables, and associated mean values and percent declines in projected abundance for each analysis updated (Section 5.2).
4. Additional clarification about model results added to Section 5.3.1.
5. Additional citations added to Literature Cited.
6. New information in Appendix E “Future Condition Model Methods and Results” include some minor corrections throughout; new paragraphs; Tables E5 – E12 updated; and Figures E2 – E12 updated.

EXECUTIVE SUMMARY

The Service was petitioned in 2012 to list the alligator snapping turtle as a threatened or endangered species under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531-1543) (Act). This Species Status Assessment serves as a compilation of the best available scientific information about the species as well as an assessment of its current and future resiliency, redundancy, and representation. The information detailed in this document will serve as the biological underpinning of the U.S. Fish and Wildlife Service's forthcoming decision on whether the alligator snapping turtle warrants protection under the Act.

The alligator snapping turtle is the largest species of freshwater turtle in North America and is among the most aquatic. Sexual maturity is achieved in 11-21 years for males and 13-21 years for females. No more than one clutch per year per female (average 27.8 eggs per clutch) has been observed in the wild, and they exhibit lower reproductive output than the smaller common snapping turtle (*Chelydra serpentina*). They do not appear to be particularly selective about nest sites, but nests have been observed across a range of distances – approximately 8 to 656 ft (2.5 to 200 m) landward from the nearest water. Temperature of the nest site is important because this species also exhibits temperature-dependent sex-determination, Type 2 – where more males are produced at intermediate incubation temperatures and more females are produced at the two extremes (Ernst and Lovich 2009, p. 16, 144-146). Most nesting occurs from May to July (Reed et al. 2002, p. 4) with areas in the southern part of the range (e.g., Georgia, Florida and Louisiana) beginning in April and extending through May and areas in the north/western portion of the range occurring from late May through June to early July (Ernst and Lovich 2009, p. 145, Carr et al. 2010, p. 87). Nest predation is a major source of mortality in many turtle populations. Growth is rapid until maturity (11-21 years of age), slowing after 15 years of age (Dobie 1971, p. 654). Alligator snapping turtles display sexual dimorphism with males being distinctly larger than females and having a greater anterior-to-vent tail length.

Alligator snapping turtles are associated with deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows), with shallower water occupied in early summer and deeper depths in late summer and mid-winter, representing a thermoregulatory shift (Ernst and Lovich 2009, p. 141). Hatchlings and juveniles tend to occupy shallower water, in comparison. Alligator snapping turtles are also associated with structure (e.g., tree root masses, stumps, submerged trees, etc.), and may occupy areas with a high percentage of canopy cover or undercut stream banks. Alligator snapping turtles are opportunistic predators and foragers and consume a variety of foods. Fish comprise a significant portion of the alligator snapping turtle's diet; however, crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetation (including acorns) have also been reported (Ernst and Lovich 2009, p. 147). Movements can be highly variable. In Black Bayou Lake and Bayou DeSiard daily distance traveled ranged from 91 to 377 ft per day (Sloan and Taylor 1987, p. 345).

A table of individual, population, and species needs for the alligator snapping turtle is below (Table ES1).

Table ES1. Individual, population, and species needs for alligator snapping turtles.

Individual Needs	
Life Stage	**Need**
Eggs	Temperatures 66° to 80° F (19° to 26.5° C) increasing to 79° to 98° F (26.1° to 36.5° C) as the season progresses
Eggs	Near shore areas (8 to 656 ft [2.5 to 200 m] landward from the nearest water) with appropriate temperatures (see above)
Hatchlings	Shallow water and increased canopy cover
Juveniles	Found in small streams with mud and gravel bottoms (e.g., 8-18 in [20-46 cm] deep)
Hatchling/Juvenile/ Adult	Primarily fish, but also crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetation (including acorns)
Juvenile/Adult	Deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows); shallower water in early summer and deeper depths in late summer and mid-winter (which may be a thermoregulatory shift)
Juvenile/Adult	Structure (e.g., tree root masses, stumps, submerged trees, etc.); may include a high percentage of canopy cover; or within stream banks
Adult	Mates
Adult	Suitable soils for nesting - generally not found in: 1) low forested areas and 2) areas with leaf litter and root mats
Population Needs (Resiliency)	
Individual needs at larger scale	For populations to persist, they need adequate conditions for breeding, feeding, sheltering, and survival as described above at a larger scale
Habitat Quantity and Connectivity	Areas of connected habitat must be sufficient in size to support enough alligator snapping turtles to allow individuals to find mates while avoiding inbreeding
Abundance	Populations need enough individuals to provide resilience against stochastic demographic and environmental variation
Species Needs	
Redundancy	Multiple resilient populations distributed throughout the species' range to buffer species against effects of catastrophic events on individual populations
Representation	Maintenance of variation within and among populations in terms of genetics (3 broad genetic lineages, with finer genetic structure among drainages), habitat types, and life history strategies (varies along north-south gradient), to allow the species to adapt to changing environmental conditions

Extensive commercial and recreational harvesting in the last century resulted in significant declines to many alligator snapping turtle populations. Commercial harvest depleted populations in Louisiana, Florida, Georgia and Alabama and is now prohibited in all states within the range of the species. Recreational harvest of alligator snapping turtles is prohibited in every state except for Louisiana and Mississippi. Although regulatory harvest restrictions have decreased the quantity of alligator snapping turtles being harvested, populations have not necessarily increased in response. This lag in population response is likely due to the demography of the species, specifically delayed maturity, long generation times, and relatively low reproductive output.

Currently, the primary negative influences on viability of alligator snapping turtles are: legal and illegal intentional harvest (including for export), bycatch associated with commercial fishing of catfish and buffalo, habitat alteration, and nest predation. Climate change and disease might negatively influence the species, but the impacts of these drivers on the species are more speculative due to a lack of information. Conversely, conservation measures that have been implemented for the alligator snapping turtle include head-starting and reintroductions, as well as various efforts to restore and improve habitat.

To determine the representation across the range of the species, we used a tiered approach (first using genetics and then life history and ecology) and delineated five representative units: Western, Southern Mississippi, Northern Mississippi, Alabama, and Apalachicola. Subdivision of representative units into analysis units was based primarily on Hydrologic Unit Code (HUC) 2 watershed boundaries. In creating analysis units, we strove to balance the needs to a) have units small enough to be able to capture the variation in the condition of the species (e.g., abundance, threats) across its range, while also b) retaining units large enough that species experts would be able to summarize information about the condition of the species for every unit (Figure ES1).

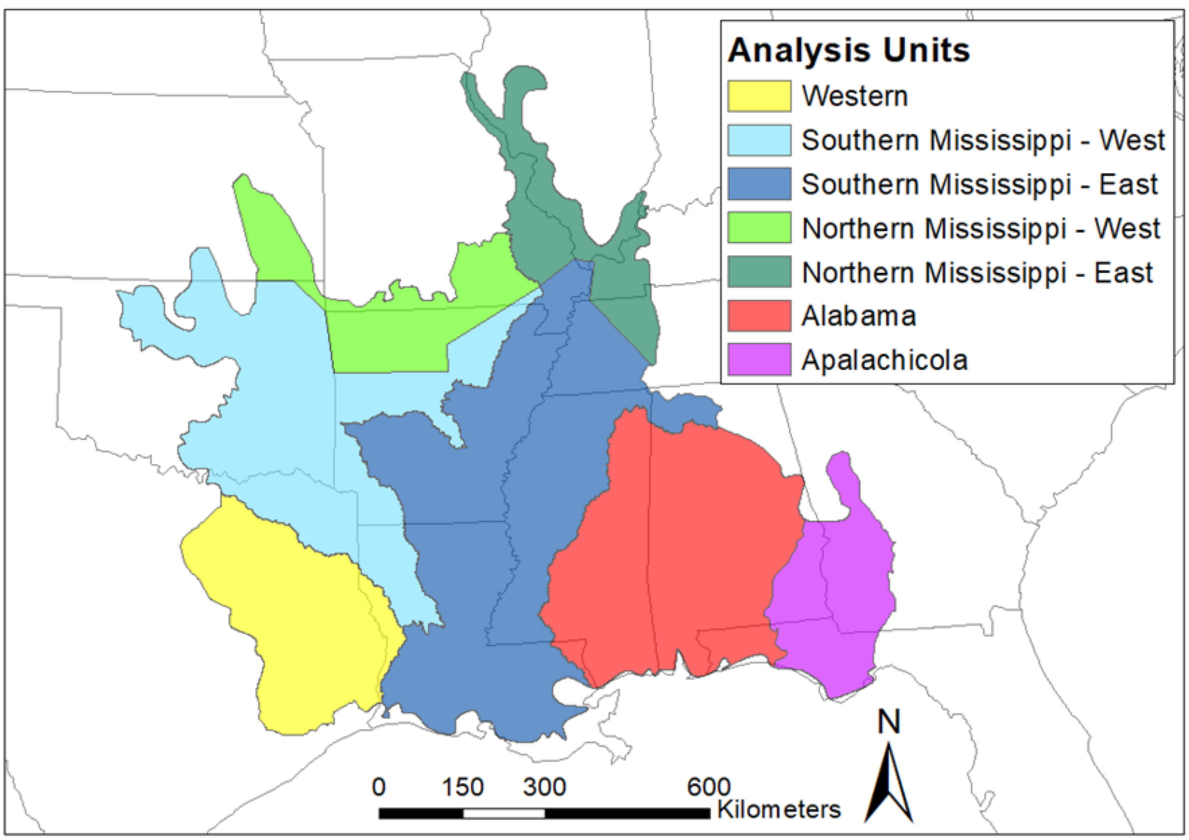


Figure ES1. Alligator snapping turtle analysis units. The two Southern Mississippi units (blues) make up one representative unit and the two Northern Mississippi units (greens) make up one representative unit; the remaining analysis units each make up a single representative unit.

Current Conditions

To assess the current condition of alligator snapping turtles, information was gathered from species experts about current abundance (our measure of resilience), current threats, and a comparison of the current and historical distribution. Estimates of abundance across analysis units range from a high of 200,000 alligator snapping turtles in the Alabama Unit to a low of 212.5 turtles in the Northern Mississippi – East Unit. Both the Northern Mississippi – East and Northern Mississippi – West Units, at the northern reaches of the species' range, have estimated abundances orders of magnitude smaller than most of the more southerly units. These northern units have also experienced more range contraction and local extirpation than more southern units.

The range-wide abundance of alligator snapping turtles is estimated to be between 68,154 and 1,436,825 (a range of 1,368,671; Table ES2). This enormous range in the estimated abundance illustrates the very high degree of uncertainty that exists in abundances at local sites and the ability to extrapolate local abundance estimates to a much broader spatial scale. Within these bounds, the most likely estimate of range-wide alligator snapping turtle abundance is 361,213 turtles, with 55% of these occurring in the Alabama Analysis Unit.

Alligator snapping turtles range-wide are believed to be exposed to the threat of incidental hooking on recreational trot and limb lines, with estimates of the percentage of turtles exposed to the threat ranging from 45% to 80%, with the exception of the North Mississippi – East Analysis Unit, where incidental hooking is not a significant threat. We received very little information about the extent of the threat of commercial fishing bycatch, suggesting either that this is not believed to be a significant threat, or that there is too much uncertainty in the extent of the threat for the experts to provide useful estimates. Legal harvest is limited to Louisiana and Mississippi, so this threat, despite its large potential impact on demography, is spatially limited to the analysis units in which those two states occur. There is wide variation in the estimated prevalence of illegal harvest across the species' range, with the highest estimates in the analysis units where legal harvest is also present. Estimates of the extent of nest predation vary. Estimates are lowest in the Southern Mississippi – West and Northern Mississippi – West Units (both 30%), with the highest extents in the remaining five analysis units (61-94%).

Because of the variation in analysis unit size and limitations in calculating true densities of alligator snapping turtles within units, we refrain from leaning heavily on comparisons of abundance or density between analysis units to summarize resilience other than to highlight general patterns. Resilience increases with abundance and density; where there are more individuals, populations will have a greater ability to withstand stochastic demographic and environmental events. Thus, resilience is highest in the core of the species' range, and lowest in the northern-most analysis units at the edge of the range. While we caution against leaning too heavily on comparisons of current abundance or density between populations because of high uncertainty contained in the information that generated the estimates, this is the best information currently available and these values will serve as useful baseline conditions against which to compare future resilience in the next chapter of this SSA.

Table ES2. Analysis units listed in descending order of estimated abundance (most likely estimate from expert elicitations) and densities expressed as estimated abundance per 2,471 ac (1,000 ha) of open water in each unit. Threats are listed where over 50% of alligator snapping turtles are exposed to harvest or over 50% of nests are exposed to nest predation by subsidized or non-native predators. Where the range of the species is contracting, the states experiencing the losses are noted.

Analysis Unit	Estimated Abundance	Abundance/ 1,000 hectares Open Water	Substantial Threats*	Range Contraction
Alabama	200,000	616.9	1) Adult harvest (Legal & Illegal) 2) Nest Predation 3) Incidental Hooking/Hook Ingestion	
Western	50,500	139.3	1) Nest Predation	
South MS - East	50,000	55.3	1) Adult harvest (Legal & Illegal) 2) Nest Predation	TN
Apalachicola	45,000	281.3	1) Nest Predation	
South MS – West	15,000	30.2	1) Incidental Hooking/Hook Ingestion	KS, possibly OK
North MS – West	500	4.7	1) Incidental Hooking/Hook Ingestion	KS
North MS - East	212.5	1.0	1) Nest Predation	IL, TN, KY, MO

*"Substantial" threats here refer to those threats estimated to reduce survival rates of an age class by 8 percent or more (see Figure 16 in Section 4.5.2): legal and illegal harvest reduce adult survival and nest predation reduces nest survival. To be listed for any given analysis unit, the substantial threat must be estimated to be impacting > 50 percent of the alligator snapping turtles in the unit.

No representative units have been lost compared to the historical distribution. The Northern Mississippi Representative Unit, which adds diversity in life history strategies within the species, currently has very low abundance within its two constituent analysis units relative to the other representative units, with an estimated 712.5 alligator snapping turtles total and a shrinking range. The representative units within the core of the species' range are estimated to support at least 45,000 alligator snapping turtles.

The species has experienced range contractions in the northern portions of the range (Oklahoma, Kansas, Missouri, Illinois, Kentucky, and Tennessee). Within the core of their range, however, alligator snapping turtles still seem to be widely distributed, though there are many gaps in the spatial extent of surveys. While the distribution of the species still encompasses much of its historical range, resilience within that range has decreased, largely from historical harvest pressures. The Northern Mississippi – East Analysis Unit has decreased in resilience and can only have limited contributions to redundancy, given current abundance (only 212.5 estimated abundance, influenced largely by introductions). While range contractions have occurred within various states, at present, the species occurs in all historically known states, except for Kansas where it is unknown if any populations or even individuals still persists.

Future Conditions

To assess future conditions and viability of the alligator snapping turtle, we constructed a female-only, stage-structured matrix population model to project alligator snapping turtle population dynamics over 50 annual time steps. We used the best available data from the literature to parameterize the population matrix, and elicited data from species experts to quantify stage-specific initial abundance, the spatial extent of threats, and threat-specific percent reductions to survival. To reflect differences among analysis units, we adjusted initial abundance and the demographic parameters within the matrix model based on the proportion of the population within the unit exposed to each threat. To account for potential uncertainty in the effects of each threat, we created six different scenarios, in which the threat-induced reductions to survival were unaltered, increased by 25%, or decreased by 25%, and the spatial extent of each threat left the same, increased by 25%, or reduced by 25% to simulate conservation actions. We used a fully stochastic projection model that accounted for uncertainty in the demographic parameters to predict future conditions of the alligator snapping turtle in five of the eight analysis units under the six different scenarios. We then used the model output to predict the probability of extinction and quasi-extinction, defined here as the probability that the total alligator snapping turtle population declined to less than 5% of the abundance in year one of the simulation (e.g., starting abundance).

Resilience for all analysis units is expected to decline drastically across all analysis units under all scenarios. We modeled scenarios that reflected uncertainty in the impact of threats on alligator snapping turtle demography, and all scenarios produced mean growth rates indicating population decline. With the exception of the Northern Mississippi – East Unit, all other analysis units were predicted to be quasi-extirpated within 50 years with a probability of over 98 percent. Though the risk of quasi-extirpation was lower in the Northern Mississippi – East Unit this analysis unit than the others, this was in part an artefact of the way that quasi-extirpation thresholds were defined, as a percentage of the initial abundance; even though quasi-extirpation risks were lower than other analysis units, the predicted abundances for this unit were still low, fewer than 51 female turtles, and still indicate that alligator snapping turtles will become very rare or disappear from this analysis unit.

Time to quasi-extirpation varied across analysis units and scenarios, but in general, the first analysis unit likely to reach the quasi-extirpation threshold was the Alabama Unit (12-22 years), followed by the Southern Mississippi – East Unit (after an average of 14-25 years depending on the scenario), the Apalachicola Unit (21-33 years), and finally the Northern Mississippi – East Unit where quasi-extirpation was not likely. The Western, Southern Mississippi – West, and Northern Mississippi – West analysis units were not included in the futures simulation modeling because we did not have adequate input data to do so. However, we have no evidence that alligator snapping turtle demographic trends in response to threats in these analysis units would be dramatically different from the range of analysis units that were modeled; therefore, it is likely that alligator snapping turtles in these analysis units will decline along similar trajectories as the modeled analysis units.

Future representation, referring to the ability of the species to adapt to changing environmental conditions over time, is similarly predicted to decline rapidly as alligator snapping turtles in every representative unit decline in abundance to quasi-extirpation or true extirpation. The loss of alligator snapping turtles across all representative units would represent losses in genetic diversity (2 broad genetic lineages), life history diversity along a north-south gradient, and finer scale genetic differences among drainages within the larger genetic lineages.

Future redundancy, or the ability to withstand catastrophic events, for alligator snapping turtles is expected to decline drastically over the next 50 years. Our future simulation model should be operated at the scale of the analysis unit, so we cannot provide precise predictions about which states or counties are most likely to lose or retain alligator snapping turtles in the future. At the analysis unit scale, however, all units were predicted to lose resilience at such a high rate that redundancy is not expected to remain across the landscape. Where alligator snapping turtles persist in the future, they are likely to be rare and not found in resilient groupings. Analysis units were predicted to reach quasi-extirpation thresholds in some cases within the next two decades, with more units becoming quasi-extirpated each subsequent decade within our 50-year modeling period. The addition of conservation actions, or different assumptions about the impact of threats on alligator snapping turtle demography, altered the time to quasi-extirpation by about a decade at most, typically less. No scenarios resulted in stable or increasing redundancy.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 – INTRODUCTION AND ANALYTICAL FRAMEWORK

The alligator snapping turtle is a reptile that is confined to river systems that flow into the Gulf of Mexico, extending from the Suwannee River in Florida to the San Antonio River in Texas. On July 11, 2012, we, the U.S. Fish and Wildlife Service (USFWS), received a petition dated July 11, 2012, from The Center for Biological Diversity (CBD) requesting that 53 species of reptiles and amphibians, including the alligator snapping turtle, be listed as endangered or threatened and that critical habitat be designated under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531-1543) (Act). On July 1, 2015, the Service announced our 90-day finding that the petition presented substantial scientific or commercial information indicating that the petitioned action may be warranted (80 FR 37568). On September 1, 2015, CBD posted supplemental information to regulations.gov in which they requested the Service to consider whether any populations of alligator snapping turtles should be considered a distinct species. A review of the status of the species was initiated to determine if the petitioned action is warranted. Based on the status review, the Service will issue a 12-month finding for the alligator snapping turtle. Thus, we conducted a Species Status Assessment (SSA) to compile the best available data regarding the species' biology and factors that influence the species' viability. The SSA Report is a summary of the information assembled and reviewed by the Service and incorporates the best scientific and commercial data available. This SSA Report documents the results of the comprehensive status review for the alligator snapping turtle and serves as the biological underpinning of the Service's forthcoming decision (12-month finding) on whether the species warrants protection under the Act.

The SSA framework (USFWS 2016, entire) is intended to be an in-depth review of the species' biology and threats, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long-term viability. The intent is for the SSA Report to be easily updated as new information becomes available and to support all functions of the Ecological Services Program of the Service, from candidate assessment to listing to consultations to recovery. As such, the SSA Report will be a living document that may be used to inform Endangered Species Act decision making, such as listing, recovery, Section 7, Section 10, and reclassification decisions (the latter four decision types are only relevant should the species warrant listing under the Act). Therefore, we have developed this SSA Report to summarize the most relevant information regarding life history, biology, and considerations of current and future risk factors facing the alligator snapping turtle. In addition, we forecast the possible response of the species to various future risk factors and environmental conditions to formulate a complete risk profile for the alligator snapping turtle.

The objective of this SSA is to thoroughly describe the viability of the alligator snapping turtle based on the best scientific and commercial information available. Through this description, we determined what the species needs to support viable populations, its current condition in terms of those needs, and its forecasted future condition under plausible future scenarios. In conducting this analysis, we took into consideration the likely changes that are happening in the environment – past, current, and future – to help us understand which factors drive the viability of the species.

For the purpose of this assessment, we define **viability** as a description of the ability of a species to sustain populations in the wild beyond a biologically meaningful time frame. Viability is not a specific state, but rather a continuous measure of the likelihood that the species will sustain populations over time (USFWS 2016, p. 9). Using the SSA framework (Figure 1), we consider what the species needs to maintain viability by characterizing the status of the species in terms of its **resiliency**, **representation**, and **redundancy** (U.S. Fish and Wildlife Service (USFWS) 2016, entire).

- **Resiliency** describes the ability of a population to withstand stochastic disturbance. Stochastic events are those arising from random factors such as weather, flooding, or fire. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Generally, populations need enough individuals within habitat patches of adequate area and quality to maintain survival and reproduction in spite of disturbance.

- **Representation** describes the ability of the species to adapt to changing environmental conditions over time. Representation can be measured through the genetic diversity within and among populations and the ecological diversity (also called environmental variation or diversity) of populations across the species' range. Theoretically, the more representation the species has, the higher its potential of adapting to changes (natural or human caused) in its environment.

- **Redundancy** describes the ability of a species to withstand catastrophic events. A catastrophic event is defined here as a rare, destructive event or episode involving multiple populations and occurring suddenly. Redundancy is about spreading risk among populations, and thus, is assessed by characterizing the number of resilient populations across a species' range. The more resilient populations the species has, distributed over a larger area, the better the chances that the species can withstand catastrophic events.

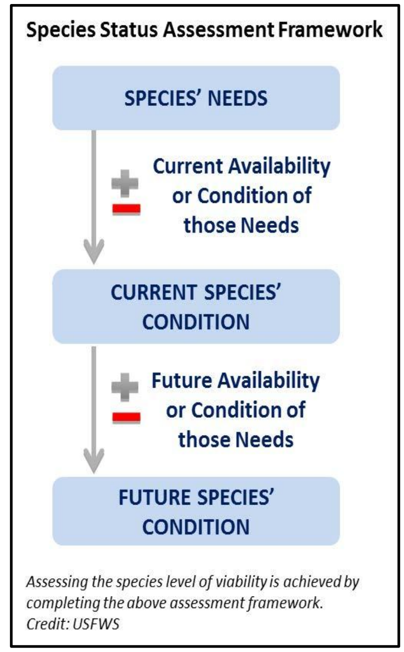


Figure 1. Species Status Assessment Framework

This SSA Report includes the following chapters:

1. Introduction;
2. Species Biology and Individual Needs. The life history of the species and resource needs of individuals;
3. Factors Influencing Viability. A description of likely causal mechanisms, and their relative degree of impact, on the status of the species;
4. Population and Species Needs and Current Condition. A description of what the species needs across its range for viability, and estimates of the species' current range and condition; and,
5. Future Conditions and Viability. Descriptions of plausible future scenarios, and predictions of their influence, on alligator snapping turtle resiliency, representation, and redundancy.

This SSA Report provides a thorough assessment of the biology and natural history and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the alligator snapping turtle. Importantly, this SSA Report does not result in, nor predetermine, any decisions by the Service under the Act. In the case of the alligator snapping turtle, the SSA Report does not determine whether the alligator snapping turtle warrants protections of the Act, or whether it should be proposed for listing as a threatened or endangered species under the Act. That decision will be made by the Service after reviewing this document, along with the supporting analysis, any other relevant scientific information, and all applicable laws, regulations, and policies. The results of the decision will be announced in the Federal Register. The contents of this SSA Report provide an objective, scientific review of the available information related to the biological status of the alligator snapping turtle.

CHAPTER 2 – SPECIES BIOLOGY AND INDIVIDUAL NEEDS

In this chapter, we provide biological information about the alligator snapping turtle, including its taxonomic history, morphological description, historical and current distribution, and known life history. We then outline the resource needs of individuals.

2.1 Taxonomy

The alligator snapping turtle (*Macrochelys temminckii*) is a member of the Family Chelydridae, Order Testudinata, Class Reptilia. This family includes two genera *Macrochelys* and *Chelydra*. *Chelydra* is represented by three species occurring within the Americas: 1) common snapping turtle found in North America (*Chelydra serpentina*), 2) South American snapping turtle (*Chelydra acutirostris*), and 3) Central American snapping turtle (*Chelydra rossignonii*). The nomenclatural history of the alligator snapping turtle is complex and continues to evolve. The species was first described in 1789 as *Testudo planitia* but it was placed in the genus *Macrochelys* by Gray in 1856. Although subsequent authors referred to the genus as *Macrochelys*, this placement was refuted and it was believed the alligator snapping turtle should be included in the genus *Macroclemys* (Smith 1955, p. 16). In 1995, Webb demonstrated that the genus *Macrochelys* has precedence over *Macroclemys*, and the Society for the Study of Amphibians and Reptiles adopted this revision in 2000

(Crother et al. 2000, p. 79). Accordingly, for the purpose of this report, we will use *Macrochelys* as the genus name.

Historically, the alligator snapping turtle was considered a single, wide-ranging species (*Macrochelys temminckii*) until a recent analysis of variation in morphology and genetic structure described two new species of alligator snapping turtles: the Apalachicola alligator snapping turtle (*Macrochelys apalachicolae*) and the Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) (Thomas et al. 2014, entire).

Three genetically distinct lineages of *Macrochelys* were identified morphologically, with *Macrochelys suwanniensis* being the most distinct (Thomas et al. 2014, p. 161). The carapace of *Macrochelys suwanniensis* can be differentiated by the presence of a large, lunate caudal notch, whereas *Macrochelys temminckii* and *Macrochelys apalachicolae* have narrow, triangular or U-shaped caudal notches that are more difficult to differentiate from each other. The skulls of *Macrochelys temminckii* and *Macrochelys apalachicolae* have large, globular squamosal projections, whereas the skulls of *Macrochelys suwanniensis* has an acute, sharp squamosal projection. In addition to these morphological differences, a reanalysis of genetic sequence data (data originally analyzed in Roman et al. 1999, entire) generated a similar evolutionary gene tree as the original analysis with three major clades of *Macrochelys temminckii* identified: 1) a western clade including populations from the Trinity River to Pensacola Bay (retained as *Macrochelys temminckii*), 2) a central clade from the Choctawhatchee River to the Ochlockonee River (corresponding to *Macrochelys apalachicolae*), and 3) an eastern clade restricted to the Suwannee River (corresponding to *Macrochelys suwanniensis*) (Thomas et al. 2014, p. 147-148).

A subsequent publication, however, argued that the morphological and genetic data presented by the former study did not support distinguishing *Macrochelys apalachicolae* from *Macrochelys temminckii* (Folt and Guyer 2015, entire). The authors tested for morphological differences among the three hypothesized populations by comparing the mean values and standard deviation of four variables (i.e., caudal notch depth, caudal notch width, caudal notch area and squamosal angle) analyzed in Thomas et al. (2014, entire). Results indicated the Suwannee population as distinct from the other two populations for mean values of all four variables. The statistical distribution of variables was also mostly non-overlapping and distinct when compared to the other populations; therefore, the data supported separation of the Suwannee population as a distinct species (Folt and Guyer 2015, p. 449-450). Comparison of the mean values between the western and central populations showed less differentiation. Significant differences were only shown for two of the four variables, and the statistical distribution of variables showed considerable overlap; therefore, the authors argued that the data did not support the separation of the central population (*Macrochelys apalachicolae*) from the western population (*Macrochelys temminckii*) (Folt and Guyer 2015, p. 449-450).

In addition, there are seven rivers between the Suwannee population and the central population that lack vouchered specimens (Ewert et al. 2006, p. 60-61). This distributional gap likely resulted in the genetic and morphological distinction of *Macrochelys suwanniensis* (Folt and Guyer 2015, p. 449). While genetic data suggest limited gene flow between the western and central populations, it does not necessarily eliminate the possibility of rare dispersal events. Barnacles have been observed growing on shells of *Macrochelys* in coastal areas, which implies a certain level of salt tolerance to make dispersal possible (Ernst and

Lovich 2009, p. 141). Microsatellite data have also suggested recent gene flow from Pensacola to Apalachicola (Echelle et al. 2010, p. 1380). This dispersal and gene flow would serve to maintain species connectivity between the central and western populations, while the geographic isolation of *Macrochelys suwanniensis* would limit dispersal and promote divergence (Folt and Guyer 2015, p. 449).

In addition to the above information, the Society for the Study of Amphibians and Reptiles recognizes two species of *Macrochelys*: 1) *Macrochelys temminckii* and 2) *Macrochelys suwanniensis*. The Turtle Taxonomy Working Group also concurred with the recognition of two species since Folt and Guyer (2015) reconsidered published data, critiqued the methods of Thomas et al. (2014), and provided evidence to support the distinction of *Macrochelys suwanniensis* (Rhodin et al. 2017, p. 26). They also agree that, to date, there is not enough evidence to distinguish *Macrochelys apalachicolae* from *Macrochelys temminckii*.

2.2 Species Description

The alligator snapping turtle (Figure 2) is the largest species of freshwater turtle in North America and is highly aquatic and somewhat secretive. They are primitive in appearance and are characterized by a large head, long tail, and an upper jaw with a strongly hooked beak. They have muscular legs and webbed toes with long, pointed claws. They have three keels with posterior elevations on the scutes of the carapace, which is dark brown and often has algal growth that adds to the alligator snapping turtle's camouflage. Their hinge-less plastron is significantly smaller than their carapace and is narrow and cross-shaped with a long, narrow bridge. The plastron is greyish-brown in color in adults; in juveniles it may be somewhat mottled with small whitish blotches. Their eyes are positioned on the side of the head and are surrounded by small, fleshy, pointed projections. Numerous epidermal projections are also present on the side of the head, chin and neck (Ernst and Lovich 2009, p. 138-139). Hatchlings look very similar to adults (Ernst and Lovich 2009, p. 146).

Figure 2. Alligator snapping turtle. Photo credit Eva Kwiatek.

2.3 Range and Distribution

Due to the aquatic nature of the species, the alligator snapping turtle is confined to river systems that flow into the Gulf of Mexico, extending from the Suwannee River in Florida to the San Antonio River in Texas (Figure 3). In the Mississippi Alluvial Valley, it is widely distributed from the Gulf to as far north as Indiana, Illinois, southeastern Kansas and eastern Oklahoma. In the Gulf Coastal Plain, its range extends from eastern Texas to southern Georgia and northern Florida. Historically, the alligator snapping turtle occurred over eastern Oklahoma, but today it is believed to be restricted to the east central and southeastern portion of the state (Ernst and Lovich 2009, p. 139). In addition, in a letter dated August 25, 2018, the State of Iowa Department of Natural Resources (DNR) informed the Service that the alligator snapping turtle record that was once considered evidence that this species existed in Iowa is no longer considered credible; and, a committee of regional herpetological experts recommended removing the species from the list of Iowa Species of Greatest Conservation Need. The species was removed from Iowa DNR's Wildlife Action Plan in 2015 (Iowa Department of Natural Resources 2015).

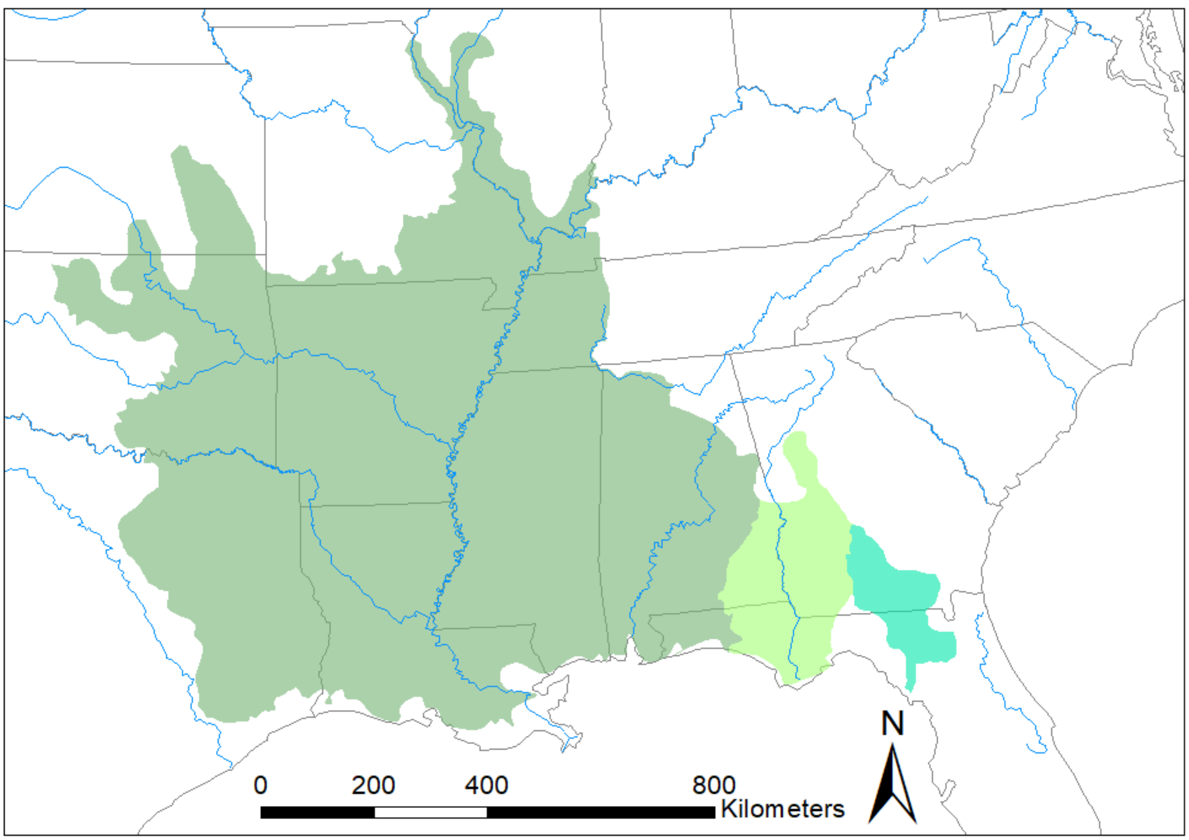


Figure 3. Range of the alligator snapping turtle. Different shades represent three main genetic lineages.

Current research indicates range-wide genetic divergence between populations of the species among river drainages. Three genetically distinct populations have been identified: the greater Mississippi River watershed (western), the Gulf coastal rivers east of the greater Mississippi River watershed (central), and the Suwannee River drainage (eastern) system (Roman et al. 1999, p. 138-139). Extirpation of any local population in one of the three drainage basins may lead to loss of genetic variability and vigor, increased vulnerability of remaining populations to disease and predation, difficulties in obtaining appropriate founder stock for possible use in future recovery efforts (if needed) and loss of the species' unique function and role in the ecosystem.

Alligator snapping turtles were historically found in 14 states: Alabama, Arkansas, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. Currently, the species is known to occur in Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. This list includes all historically occupied states except for Indiana and Kansas, where persistence is unknown. In Indiana, alligator snapping turtle eDNA has been collected in the water, but presence has not been confirmed with trapping. In Kansas, the species has not been detected since a 1991 record in Montgomery County (See Section 4.5.3 for methods of collecting this information).

2.4 Habitat

Alligator snapping turtles are generally found in deeper water of large rivers and their major tributaries; however, they are also found in a wide variety of habitats, including small streams, bayous, canals, swamps, lakes, reservoirs, ponds, and oxbows (a lake that forms when a meander of a river is cut off). Alligator snapping turtles more often select structure (e.g., tree root masses, stumps, submerged trees, etc.) than open water and may select sites with a high percentage of canopy cover (Howey and Dinkelacker 2009, p. 589; Harrel et al. 2006, p.66; Carr et al. 2007, p.37; Carr et al. 2010, p.43). The amount of suitable alligator snapping turtle within its range and a description of how those numbers were derived is presented in Appendix A.

In Florida, optimum habitat has been identified as swamp forests comprised of bald cypress and tupelos associated with flooded channels (Ewert et al. 2006, Ewert and Jackson 1994). In northeastern Louisiana, a variety of microhabitats associated with Black Bayou Lake and Bayou DeSiard were available (i.e., open water; bald cypress bordered channel; buttonbush with bald cypress and aquatics; flotant (floating marsh) with bald cypress or buttonbush; aquatics and emergents; bald cypress and aquatics) (Sloan and Taylor 1987, p. 346). Two individuals within Bayou DeSiard spent an average of 74.6% of the monitoring period in cypress-bordered channels that was in close proportion to that habitat's availability. Three turtles that utilized both Black Bayou Lake and Bayou DeSiard spent an average of 56.4% of the monitoring period in bald cypress-bordered channels. Eighteen percent of the total habitat available in the lake and bayou combined was bald cypress bordered channels; habitat use was three times greater than its availability. Six turtles in Black Bayou Lake spent most of their time in flotant with cypress or buttonbush habitat; habitat use was three times greater than its availability. In Arkansas and Missouri, juveniles were found in small streams with mud and gravel bottoms approximately 8 to 18 inches deep (20 to 46 cm) (Ernst and Lovich 2009, p. 141). In Arkansas, male and female alligator snapping turtles selected similar habitats throughout the year. Those habitats included sites with structure (either submerged or stream bank) and sites that had a high percentage of canopy cover. All alligator snapping turtles used sites with deep water or undercut stream banks during the summer months (Howey and Dinkelacker 2009, p. 593-594). In Kentucky, they occupied microhabitats in a lake near-shore in shallow water with a gravel or rocky substrate and underwater cover of some type (Koons and Scott 1993, p.134). In eastern Oklahoma, they were associated with overhead canopy and submerged cover (Riedle et al. 2006, p. 38). Hatchling alligator snapping turtles also prefer habitats with shallow water, woody debris, emergent vegetation (primarily buttonbush, bald cypress and water tupelo), vegetation mats and increased canopy cover (Spangler 2017, p. 46; Carr et al. 2007, p. 1). In general, the species uses shallower water in early summer and deeper depths in late summer and mid-winter, which may be a thermoregulatory shift (Fitzgerald and Nelson 2011). The presence of barnacles on some specimens may also indicate an ability to spend prolonged periods in brackish water (Jackson and Ross 1971, p.188-189).

2.5 Diet and Feeding

Alligator snapping turtles are opportunistic scavengers and consume a variety of foods. Fish comprise a significant portion of the alligator snapping turtle diet; however, crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetation (including acorns) have also been reported (Elsey 2006, p. 448-489). The alligator snapping turtle is the only turtle species that has a predatory lure (a small, worm-like appendage on the tongue; Figure 4). Both adults and juveniles use this lure to attract fish into striking range. The lure is white or pale pink in juveniles and mottled or gray in adults (Ernst and Lovich 2009, p. 147).

Figure 4. Alligator snapping turtle predatory lure. Photo credit: Ryan Bolton.

Experiments conducted on captive alligator snapping turtle hatchlings indicate that there are four phases to their feeding behavior (Ernst and Lovich 2009, p.148). In the first phase (waiting), the turtle remains motionless with its legs spread outward and its head held horizontal or tilted upward. In the second phase (luring), the jaws are opened at an approximate 70-degree angle, which can sometimes take one or two minutes. The wriggling lure can be seen in this phase. The mean distance between a turtle head and the fish is approximately 2.5 inches, and luring is often initiated after vigorous fish movement. The mean duration of luring attempts that did not end in an attack was 336 seconds. The third phase (attack) consists of the turtle rapidly closing its jaws without moving the head toward the fish. Seventy-five percent of all fish passing through the turtle's jaws and those biting the lure were captured. In the fourth phase (handling), after a fish is captured, it is held in the jaws for 1-83 seconds before it is swallowed. Swallowing is facilitated by several snaps of the jaws and large prey items are swallowed by extending the head forward. Occasionally, a turtle will utilize its claws to mutilate the posterior portion of the prey item while holding the anterior end in its jaws. Prey handling time decreases with experience.

2.6 Predation

Nest predation is a major source of mortality in many turtle populations and, historically, high levels of nest predation were likely common. Historically, those losses were offset by high survival rates of long-lived adults. These levels of nest predation, however, may be detrimental to turtle populations that are already in decline. In some species, certain aspects of turtle reproduction may also mitigate depredation risk, such as producing multiple clutches. Because of the alligator snapping turtle's low reproductive output, present levels of nest predation may be detrimental to that species. Currently, effects of high nest mortality may be exacerbated by increases in stressors such as habitat fragmentation and degradation, collection, harvesting, and climate change (Holcomb and Carr 2013, p. 478). In addition, populations of some nest predators have increased due to habitat fragmentation, the provision of supplemental food, and the decline of large carnivores (e.g., mesopredators). In turn, nest predation may be elevated above historical levels (Holcomb and Carr 2013, p. 478-479).

In a two-year study conducted at Black Bayou Lake in Louisiana, all 90 artificial nests constructed were depredated (Holcomb and Carr 2013, p. 482). These results are consistent with depredation rates on natural nests at the same location (Holcomb and Carr 2013, p. 485). Studies on common snapping turtle nest depredation resulted in similar findings. In Michigan, annual depredation rates averaged 70% with depredation levels reaching 100% in two years of the seven-year study (Congdon et al. 1987, p. 51). In a New York study, a common snapping turtle population experienced a 94.4% depredation rate over one year (Petokas and Alexander 1980, p. 242). At Black Bayou Lake, 86% of all artificial nests constructed were depredated within the first 24 hours and less than 6% survived beyond 48 hours (Holcomb and Carr 2013, p. 485). In the Michigan study, of the nests destroyed by predators, 59% occurred within the first 24 hours and 70% within six days (Congdon et al. 1987, p. 46). In Florida, however, observations suggested that nest predation seldom occurred until several days after egg laying (Ewert and Jackson 1994, p. 17).

Alligator snapping turtle nests are known to be depredated by raccoons (*Procyon lotor*) (Ewert et al 2006, p. 67). Nine-banded armadillos (*Dasypus novemcinctus*), Virginia opossums (*Didelphis virginiana*), and river otters (*Lontra canadensis*) have also been observed depredating artificial alligator snapping turtle nests (Holcomb and Carr 2013, p.482). Predators of hatchlings are likely to include large fish, wading birds, otters, and alligators (Ernst and Lovich 2009, p. 149). Red imported fire ants (*Solenopsis invicta*) are also known to cause significant decline in hatching success. Alligator snapping turtle hatchlings are most susceptible to fire ant-caused mortality during pipping (the process by which a hatchling breaks free from the egg shell) and when they are still in the nest prior to emergence. Should hatchlings make it out of the shell, they are still extremely susceptible to fire ants as they dig their way out of the nest and travel to water (Holcomb 2010, p. 12-13). There are no natural predators of large alligator snapping turtles.

2.7 Movement and Behavior

Alligator snapping turtles are among the most aquatic of freshwater turtles, and overland movements are generally restricted to nesting females and juveniles moving from the nest to water (Reed et al. 2002, p. 5). Most aquatic movement in adults occurs at night, whereas juveniles are mostly active during the day. In the Suwannee River, some adults continued moving between the floodplain and river channel after water levels fell and they had to travel over land at night (Enge et al. 2014, p. 24). Basking in this species rarely occurs and most reports consist of a single observation (Carr et al. 2011, p. 3; Ewert 1976, p. 154). In 2009, two instances of aerial basking and one of aquatic basking were observed on Black Bayou Lake National Wildlife Refuge (Carr et al. 2011, p. 3). Alligator snapping turtles cannot remain submerged for long periods of time compared to other aquatic turtles. At water temperatures of 21-24ºC (69.8-75.2ºF), submergence times range from 40 to 50 minutes (Ernst and Lovich 2009, p. 141).

Radiotelemetry has been used to study movements of alligator snapping turtles. In Kansas, a radio-tagged female moved 4.3 miles (6.9 km) upstream between April 11, 1986 and May 31, 1991. During the first two weeks, she traveled approximately 0.3 miles (0.46 km) and her fastest rate of travel was 27.6 feet/minute (8.4 meters/minute) for 12 minutes (Shipman et al. 1991, p. 8-9).

In Louisiana's Black Bayou Lake and Bayou DeSiard, the average daily distance traveled ranged from approximately 91 to 377 feet/day (27.8-115.5 m/day; Sloan and Taylor 1987, p. 345), and there was no significant difference between mean daily distances moved between resident and introduced turtles (Sloan and Taylor 1987, p. 348). The minimum home range varied from approximately 44 to 610 acres (18-247 ha; Sloan and Taylor 1987, p. 345), and there was no significant difference between resident and introduced turtles (Sloan and Taylor 1987, p. 348).

In 2010, Carr et al. reported no significant difference in total movements between males and females at Black Bayou Lake, Louisiana. Both males and females were less active during the winter (November and March) and summer (July to August) and most active during reproduction in the spring. During April the average daily distance traveled for males was 135 feet/day (41 meters/day), while female movement peaked in May (208 feet/day; 63.4 meters/day).

In Louisiana, home range sizes (determined via the minimum convex polygon method) in Black Bayou Lake were reported as approximately 70 acres (28.2 ha) for males and approximately 110 acres (44.8 ha) for females (Carr et al. 2010, p.18). In an earlier study (conducted in the same lake), home range sizes of both males and females were significantly larger and female home range sizes were smaller than males; males averaged approximately 357 acres (144.5 ha) and females averaged approximately 215 acres (87 ha; Sloan and Taylor 1987, p. 345). Because a large portion of the lake is within a national wildlife refuge and has received approximately 10 to 20 translocated turtles, this reduction in home range size over time may be due to an increase in density of alligator snapping turtles (Carr et al. 2010, p. 41).

In Arkansas, alligator snapping turtles were reported traveling an average distance of approximately 627 feet (191 m) and a maximum distance of 1.1 miles (1.8 km). One female moved 0.3 miles (495 m) downstream from a nest site in 20 days and then was found relocated 1.1 miles (1.8 km) upstream 28 days later (Trauth et al. 1998, p. 68). In Florida, the mean linear movement was greater for males (2.5 miles ± 0.5 miles; 4 km ± .8 km) than females (2.1 miles ± 0.2 miles; 3.4 km ± .3 km) and juveniles (1.7 miles ± 1.2 miles; 2.7 km ± 1.9 km) (Enge et al. 2014, p. 22-23).

Between March 1992 and June 1993, movement and habitat use were studied via radiotelemetry on 12 juvenile alligator snapping turtles in Bayou DeSiard, Louisiana. There were significant differences between male and female travel distances between marked locations (males approximately 0.2 miles [.32 km] and females approximately 0.1 miles [.16 km]) and mean home range length (males approximately 2.17 miles [3.49 km] and females approximately 0.88 miles [1.42 km]) (Harrel et al. 1996, p.60).

In 2006, nineteen hatchlings were tracked at Black Bayou Lake National Wildlife Refuge in Louisiana. Ten hatchlings were tracked during the spring and summer, and nine were tracked during the fall. Daily movement distances were greater in the spring than in the fall. During the spring and summer (April-August), hatchlings traveled an average distance of approximately 3.3 ft/day (1.01 m/day), and in the fall (September-December), approximately 3.1 ft/day (0.97 m/day). Daily movement distances were higher in April, June, and October. Average daily movement for the study year was approximately 3.2 ft/day (0.97 m/day; Carr et al. 2007, p.36).

2.8 Life Cycle and Reproduction

Sexual maturity is achieved in 11-21 years for males and 13-21 years for females (Figure 5) (Tucker and Sloan 1997, p. 589). Mating takes place and has been observed in captive alligator snapping turtles from February to October, but geographic variation among wild populations is not well understood (Reed et al. 2002, p. 4). Females ovulate in spring and apparently breed yearly, though poor foraging success may cause females to skip a breeding year. No more than one clutch per year per female has been observed in the wild, and they exhibit lower reproductive output than the smaller common snapping turtle (*Chelydra serpentina*; Reed et al. 2002, p. 4). Clutch sizes have been reported from across the species' range (9-61 eggs, with a mean of 27.8) (Ernst and Lovich 2009, p. 145); Georgia has reported as few as 9 eggs (Ernst and Lovich 2009, p. 145; Reed et al. 2002, p. 4); Florida reported 17-52 (mean 35.1; Ernst and Lovich 2009, p. 145); and Louisiana reported a mean of 23.8 eggs (Dobie 1971). Reproductive output also varies substantially among females but generally is positively correlated with body size (Reed et al., p. 4). Larger (older) females probably produce more eggs than recently matured females (Ernst and Lovich 2009, p. 145).

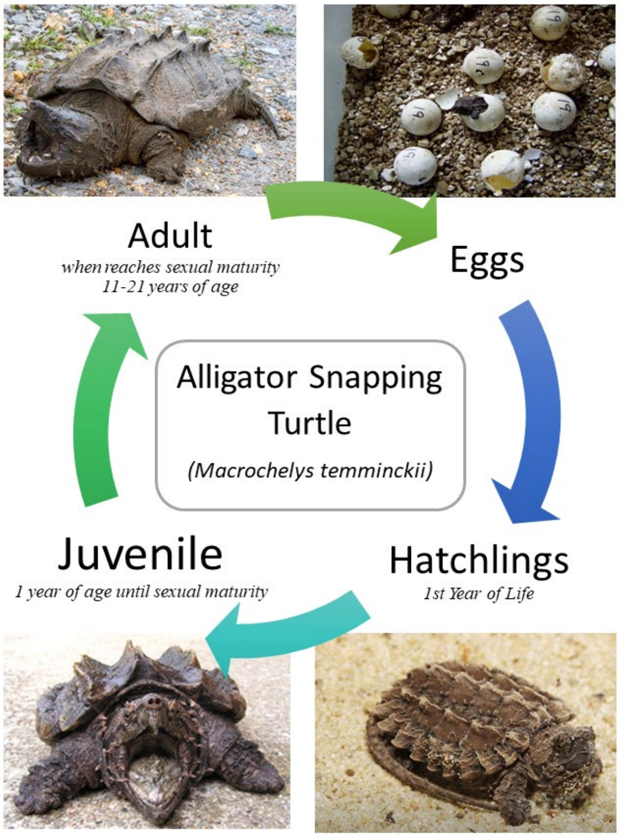


Figure 5. Alligator snapping turtle life cycle. Photo credits: Eva Kwiatek (top left), U.S. Fish and Wildlife Service (top right), Indiana DNR (bottom left), Kory Roberts (bottom right).

A detailed chronology of egg laying has been provided based on observations from near Lake Iamonia, Florida (Ewert 1976, p. 153). For this laying event, it took approximately 40 minutes for a female to lay her 36-egg clutch. When nest covering and estimated nest excavation times were factored in, the entire process took approximately 4 hours. Similarly, a female near Muckalee Creek in Georgia completed the entire nesting process in approximately 3.5 hours (Powders 1978, p. 155). Alligator snapping turtle eggs are spherical, chalky white (nearly opaque), pliable, with diameters ranging from 0.9 to 2 inches (22.9 to 51.8 mm) and weighing 16.9 to 36.1 grams (0.6 to 1.3 ounces; Ernst and Lovich 2009, p. 145).

Nesting females usually represent the only adult life stage to venture onto land (Ernst and Lovich 2009, p. 141). It is speculated that females leave the water during the late night or early dawn hours and complete nesting during the day (Ernst and Lovich 2009, p. 145). Alligator snapping turtles do not appear to be particularly selective regarding nest site conditions, though one researcher in Florida did observe a conspicuous absence of nests in low forested areas with leaf litter and root mats and on open sand bars (Ewert 1976, p. 151).

In a study at Black Bayou Lake in Louisiana, 41 alligator snapping turtle nest sites were located in areas with 46.7% canopy cover (Carr et al. 2007, p. 23).

Nests have been observed approximately 8-656 feet (2.5 to 200 m) landward from the nearest water (Ewert 1976, p. 150; Ewert et al. 2006, p.64; Jackson and Jensen 2003, p.363; Powders 1978, Trauth et al. 2004). Of 17 nests observed by Ewert (1976, p. 151), 16 averaged approximately 40 feet (12 m) from the nearest waterbody (with a range of 8-72 feet [2.4-22 m]), and one nest was observed at a distance of approximately 235 feet (72 m). In Louisiana, the documented distance to nearest water ranged from 4 to 285 feet (1.2-87 m) (Steen et al 2012, p. 124).

Internal temperature of nests in Florida indicated initial temperatures of 66°-80° Fahrenheit (F) (19°-26.5° Celsius [C]) increasing to 79°-98° F (26.1°-36.5° C) as the season progressed, with an incubation time of 105-110 days (Ernst and Lovich 2009, p. 145). This species also exhibits TSD-2 (temperature-dependent sex-determination, Type 2), where more males are produced at intermediate incubation temperatures and more females are produced at the two extremes (Ernst and Lovich 2009, p. 16, 146). Most nesting occurs from May to July (Reed et al. 2002, p. 4), with areas in the southern part of the range (e.g., Georgia, Florida and Louisiana) beginning in April and extending through May and areas in the north/western portion of the range probably occurring from late May through June to early July (Ernst and Lovich 2009, p. 145; Carr et al. 2010, p. 87).

After egg laying, hatchlings in Louisiana emerged from nests 96.5-143 days later (Holcomb and Carr 2011*a*, p. 225). In the same study, the estimated incubation period was 98-121 days, and the estimated time in the nest was 0.5-22 days (estimated incubation period and time in the nest was not reported for the 96.5 emergence day nest). Days to emergence were also shown to decrease as the temperature increased.

2.9 Age, Growth, Population Size Structure

In the absence of studies on verified unharvested populations, natural demographics and population structure are unknown for *Macrochelys* (Folt et al. 2016, p. 29). Apparent survival of adult males and females have been estimated at 0.98 for males and 0.95 for females in Georgia (Folt et al. 2016, p. 28) and 0.96 for males and 0.88 for females in Arkansas (Howey and Dinkelacker 2013, p. 6).

Hatchling turtles experience high mortality rates (Iverson 1991, entire). At Black Bayou Lake in Louisiana, estimated survival rates over a 49-day period were 61.0-81.6% (non-conservative versus conservative estimates) (Carr et al. 2007, p. 39). Potential predators of hatchlings in this study area include but are not limited to bowfin (*Amia calva*), three-toed amphiuma (*Amphiuma tridactylum*), and predatory water birds, such as the Great Blue Heron (*Ardea herodias*), Great Egret (*Ardea alba*), and Little Blue Heron (*Florida caerulea*). These species are often observed foraging in shallow water areas along the periphery of the lake (Carr et al 2007, p. 39).

Rate of survivorship of juveniles is estimated at only about 5%, with most mortality occurring in the first two years of life (Reed et al. 2002, p. 13). In a non-declining population of *Macrochelys*, however, juvenile apparent survival has been reported as 0.86 (Folt et al 2016, p. 27). Once mature, a turtle may live “a very long time if not taken by trappers”

(Ernst and Lovich 2009, p. 150). Mean generation time for the species has been reported at 31.2 years (range = 28.6-34.0 years, 95% CI) based on a demographic study in Georgia (Folt et al. 2016, p. 27). A male alligator snapping turtle caught as an adult lived for over 70 years at the Philadelphia Zoo and was estimated to be 80 years old at its death (Ernst and Lovich 2009, p. 147).

Growth data are also scarce for wild alligator snapping turtles. Annual caudal length growth rate has been reported as 5.3% in males and 5.2% in females. Weight gain in these turtles averaged 4.1% among males and 10.6% among females (Harrel et al. 1997, p. 129). Growth is rapid until maturity (11-13 years of age), slowing after 15 years of age (Dobie 1971, p. 654). Carapace scute rings can be used to determine annual growth intervals, but some discrepancy has been noted in the past (Powders 1978, Morris and Sweet 1985); the scute annuli are poorly correlated with internal bone annuli in the vertebrae and lower jaw (Dobie 1971, p. 653). Growth rate is influenced by many factors including availability of food and prevailing water temperatures; the length of the animal's activity period seems to be one of the most significant. Data from Louisiana suggest that annual growth starts in March and continues at least through July, though it is hypothesized that growth continues into late October (Dobie 1971, p. 653-654).

The sexual dimorphism of alligator snapping turtles can be measured using the relative length of the anterior-to-vent length of the tail. This measurement for males ranges from 4.5-10.5 inches (114-267 mm) and in mature females from 1.9-4.5 inches (48-114 mm) (Dobie 1971, p. 656). Turtles smaller than 28 pounds cannot be properly sexed externally, and it is often difficult to sex live animals between 28 and 55 pounds (Moler 1996, p. 6). Sexual dimorphism also exists in the maximum size and weight attained, with males exceeding females in both measures (Dobie 1971, p. 656). A sexual size dimorphism index estimate of -1.8 by mass (36 kg male/20 kg female) and -1.2 by length (53.8 cm CL male/44.6 cm CL female) has been calculated, favoring males (Ewert et al. 2006, p. 63).

An adult 1.4:1 sex ratio favoring males has been reported in northwestern Arkansas (Trauth et al. 1998, p. 242), whereas a 1:1 ratio was documented in southeastern Louisiana (Boundy and Kennedy 2006, p. 6) and Georgia (Jensen and Birkhead 2003, p. 29). An even adult sex ratio is consistent with predictions for long-lived turtles (Folt et al 2016, p. 29). An adult sex ratio of 1:2 (male:female) has been reported in Alabama (Folt and Godwin 2013, p. 214) and in Florida (Ewert and Jackson 1994, p. iii). A higher male to female sex ratio has also been reported from the Suwannee River in Florida (3.5:1) (Enge et al. 2014, p. 32), but it varied among sections of the river.

A ratio of juveniles to adults has been reported at 1:4 in Georgia (Jensen and Birkhead 2003, p. 29) and 1:3 in Alabama (Godwin 2004, p. 7). Another study in Georgia reported a greater proportion of adults than juveniles, which is a structure consistent with a general prediction for long-lived turtles like the alligator snapping turtle (Folt et al 2016, p. 29).

Relative abundance of various turtle species has been assessed at 14 sites in Louisiana and *Macrochelys* made up between 4% (Lake Arthur) and 12.5% (Lake Iatt) of the sample (Cagle and Chaney 1950, p. 387). These data, though, were collected in 1947 and may have been underreported due to trap design making it difficult for large individuals to enter. In Alabama, abundance has been reported as up to 15% (Godwin 2004, p. 217).

One metric that can be used as an indirect measure of abundance is Catch-Per-Unit-Effort (CPUE). Surveys that provide CPUE results include those that implement methods where traps are set and checked regularly over a set number of consecutive days at sampling locations across an area of the species' range. For the alligator snapping turtle, this is measured as the number of turtles caught (catch) per trap night (unit effort) and may be reported as Turtles per Trap-night (TTN). In Florida, CPUE has been reported as 0.22 (Enge et al. 2014, p. 30) and 0.25 (Moler 1996, p. 10). In Georgia, CPUE has been reported at 0.20 (Jensen and Birkhead 2003, p. 30), 0.09 (King et al. 2016, p. 582), and 0.21 (Folt et al. 2016, p. 26). In Alabama, CPUE has been reported as 0.062 and 0.081 (Folt and Godwin 2013, p. 213). In Arkansas, CPUEs of 0.13 and 0.10 were recorded (Howey and Dinkelacker 2013, p. 60). A high CPUE of 0.35 was recorded in Oklahoma (Riedle et al. 2008*b*, p. 102). The lowest CPUE was recorded as 0.057 in Louisiana, a state where heavy harvest occurred in the past (Boundy and Kennedy 2006, p. 6).

2.10 Summary of Species Biology and Individual Needs

The alligator snapping turtle is the largest species of freshwater turtle in North America (Ernst and Lovich 2009, p. 138) and is among the most aquatic. Sexual maturity is achieved in 11-21 years for males and 13-21 years for females. No more than one clutch per year per female (average 27.8 eggs per clutch) has been observed in the wild, and they exhibit lower reproductive output than the smaller common snapping turtle (*Chelydra serpentina*). They do not appear to be particularly selective about nest sites, but nests have been observed across a range of distances - approximately 8 to 656 feet (2.5 to 200 m) landward from the nearest water. Temperature of the nest site is important because this species also exhibits temperature-dependent sex-determination, Type 2 – where more males are produced at intermediate incubation temperatures and more females are produced at the two extremes (Ernst and Lovich 2009, p. 144-146; 16). Most nesting occurs from May to July (Reed et al. 2002, p. 4), with areas in the southern part of the range (e.g., Georgia, Florida and Louisiana) beginning in April and extending through May and areas in the north/western portion of the range probably occurring from late May through June to early July (Ernst and Lovich 2009, p. 145, Carr et al. 2010, p. 87). Nest predation is a major source of mortality in many turtle populations. Growth is rapid until maturity (11-21 years of age), slowing after 15 years of age (Dobie 1971, p. 654). Male and female alligator snapping turtles display sexual dimorphism, with males being somewhat larger than females and they also have a longer tail base (anterior to vent).

Alligator snapping turtles are associated with deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows); with shallower water occupied in early summer and deeper depths in late summer and mid-winter, which represent a thermoregulatory shift (Ernst and Lovich 2009, p. 141). In comparison, hatchlings and juveniles tend to occupy shallower water. Alligator snapping turtles are also associated with structure (e.g., tree root masses, stumps, submerged trees, etc.); and may occupy areas with a high percentage of canopy cover undercut stream banks. Alligator snapping turtles are opportunistic scavengers and consume a variety of foods. Fish comprise a significant portion of the alligator snapping turtle diet, but crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetation (including acorns) have also been reported (Ernst and Lovich 2009, p. 147). Movements can be highly variable but are generally a few to hundreds of feet per day.

The individual needs of alligator snapping turtles are summarized in Table 1.

Table 1. Alligator snapping turtle individual needs.

Life Stage	Need	Breeding, Feeding Sheltering, or Survival	Citation
Eggs	Temperatures 66° to 80° F (19° to 26.5° C) increasing to 79° to 98° F (26.1° to 36.5° C) as the season progresses, with an incubation time of 105-110 days (Ernst and Lovich 2009, p. 145); also exhibits TSD-2 (temperature-dependent sex-determination, Type 2 – more males are produced at intermediate incubation temperatures; more females are produced at the two extremes)	Survival, Sheltering	Ernst and Lovich 2009, p. 16, 146
Eggs	Near shore areas (8 to 656 feet [2.5 to 200 m]) landward from the nearest water) with appropriate temperatures (see above)	Survival, Sheltering	Ewert 1976, Ewert et al. 2006, Jackson and Jensen 2003, Powders 1978, Trauth et al. 2004 *in* Ernst and Lovich 2009, p. 145
Hatchlings	Shallow water and a high value for canopy cover	Survival, Sheltering	Spangler 2017, p. 46
Juveniles	Found in similar habitats as adults (see below). They may also be found in small streams with mud and gravel bottoms (e.g., 8-18 in [20-46 cm] deep)	Survival, Sheltering; Feeding	Ernst and Lovich 2009, p. 141
Juvenile/ Adult	Primarily fish but also crayfish, mollusks, smaller turtles, insects, nutria, snakes, birds, and vegetation (including acorns)	Feeding	Ernst and Lovich 2009, p. 147
Juvenile/ Adult	Deeper water (usually large rivers, major tributaries, bayous, canals, swamps, lakes, ponds, and oxbows); shallower water in early summer and deeper depths in late summer and mid-winter, which may be a thermoregulatory shift)	Shelter	Ernst and Lovich 2009, p. 141
Juvenile/ Adult	Structure (e.g., tree root masses, stumps, submerged trees, etc.); may include a high percentage of canopy cover; or undercut stream banks	Survival, Sheltering, Feeding	Howey and Dinkelacker 2009, p. 589 and p. 593-594
Adult	Mates	Breeding	
Adult	Suitable soils for nesting - generally not found in low forested areas with leaf litter and root mats and on open sand bars	Breeding	Ewert 1976, p. 151

CHAPTER 3 – FACTORS INFLUENCING VIABILITY

In this chapter, we provide information regarding negative and positive influences on viability of alligator snapping turtles, including legal and illegal intentional harvest, bycatch, habitat alteration, nest predation, climate change, disease, and conservation measures (Figure 6).

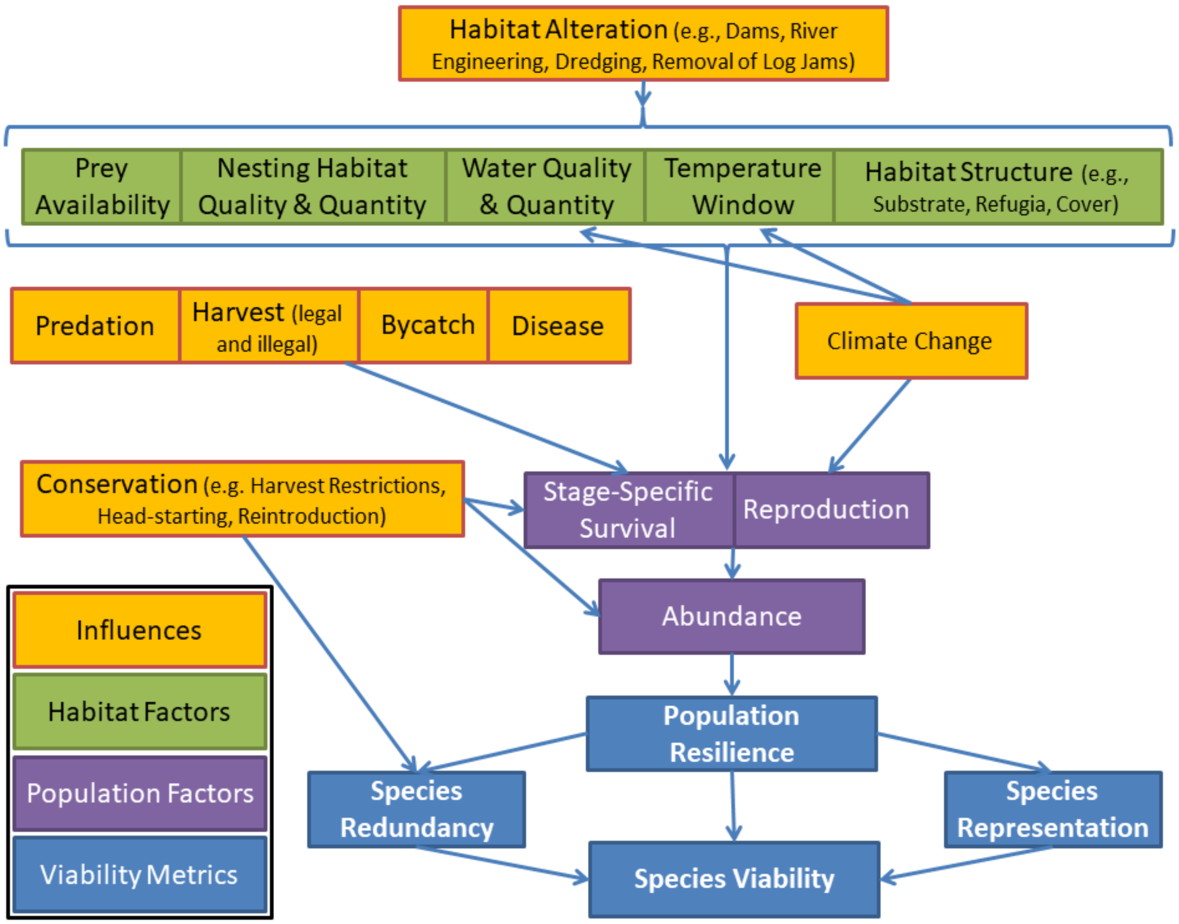


Figure 6. Simplified influence diagram illustrating how various impacts influence habitat and population factors that in turn influence the resilience of populations and viability of the species.

3.1 Harvest

3.1.1 Commercial Harvest

Extensive commercial and recreational take in the last century resulted in significant declines to many alligator snapping turtle populations across the species' range (Enge et al. 2014, p. 4). Commercial harvest of alligator snapping turtles reached its peak in the late 1960s and 1970s. During this time, Campbell's Soup Company purchased alligator snapping turtle meat for turtle soup. In addition, many New Orleans seafood restaurants also purchased large quantities of alligator snapping turtles from trappers in the southeastern states (Reed et al. 2002, p. 5). In the 1970s, the demand for turtle meat was so high that as much as three to four tons of alligator snapping turtles were harvested from the Flint River (Georgia) a day

(Pritchard 1989, p. 76). The Florida Game and Fresh Water Fish Commission (now the Florida Fish and Wildlife Conservation Commission) reported significant numbers of turtles being taken from the Apalachicola and Ochlocknee Rivers to presumably be sent to New Orleans restaurants (Pritchard 1989, p. 74-75). In addition, commercial harvest depleted populations in Louisiana and Alabama (Reed et al. 2002, p.5). Commercial harvest of alligator snapping turtles is now prohibited in all states within its range (See Table B1 in Appendix B).

3.1.2 Recreational Harvest

Recreational harvest of alligator snapping turtles is prohibited in every state except for Louisiana and Mississippi (See Table B1 in Appendix B). In Mississippi, recreational harvest is 1) limited to one turtle per year, 2) prohibited between April 1st and June 30th, and 3) limited only to individuals with a straight line carapace length of 24 inches or larger. In Louisiana, harvest of one alligator snapping turtle per day, per person, per vehicle/vessel is allowed with a fishing license. There are no reporting or tagging requirements, so the number of turtles harvested in Louisiana is unknown.

3.1.3 Impacts of Harvest

Because of the alligator snapping turtle's life history, specifically delayed maturity, long generation times, and relatively low reproductive output, they cannot sustain significant collection from the wild, especially of adult females (Reed et al. 2002, p. 8-12). The species does not reach sexual maturity until 11-21 years of age. A mature female typically only produces one clutch per year consisting of 8-52 eggs (Ernst and Barbour 1989, p. 133). The alligator snapping turtle is characterized by low survivorship in early life stages, but surviving individuals may live many decades once they reach maturity. Therefore, population growth rates of this species are extremely sensitive to the harvest of adult females. Adult female survivorship less than 98% per year is considered unsustainable, and a further reduction of this adult survivorship will generally result in significant local population declines (Reed et al. 2002, p. 9), though dynamics likely vary across the range of the species.

Although regulatory harvest restrictions have decreased the amount of alligator snapping turtles being harvested, populations have not necessarily increased in response. This lag in population response is likely due to the demography of the species, specifically delayed maturity, long generation times, and relatively low reproductive output. Twenty-two years after commercial harvest ended, surveys conducted during 2014 and 2015 in Georgia's Flint River revealed no significant change in abundance since 1989 surveys (King et al. 2016, p. 583). A similar study in Missouri and Arkansas detected population declines between the initial survey period in 1993-1994 and repeat surveys in 2009 over a decade after state-level protections were implemented (Lescher et al. 2013, p. 163-164). At Sequoyah National Wildlife Refuge in Oklahoma, an alligator snapping turtle population declined between 1997-2001 and 2010-2011 (Ligon et al. 2012, p. 40).

3.1.4 International Trade and Illegal Harvest

In 2006, the alligator snapping turtle was listed under CITES, as an Appendix III species to allow for better monitoring of exports. Prior to that listing up to 23,780 alligator snapping turtles/year were exported from the U.S. (Figure 7).

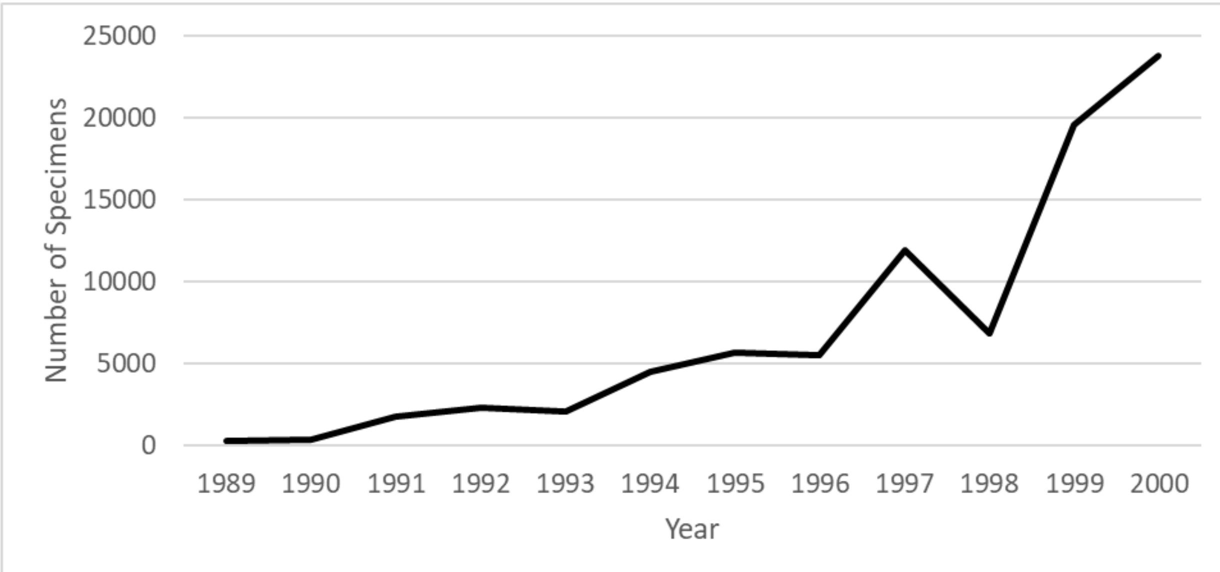


Figure 7. Number of alligator snapping turtle specimens shipped from the U.S. by year from 1989 to 2005 (data from USFWS 2005, p. 74702).

Since the CITES listing, up to 43,718 live alligator snapping turtles have been identified as "specimens taken from the wild" leaving the U.S. in a single year (Figure 8; USFWS 2018); however, nearly all of the turtles in this category were likely hatched in a captive facility. In general, turtle farms use long-term captive, wild-caught adults to produce the hatchlings that they sell, and CITES "requires an F2 offspring to qualify as captive" and all exported ASTs originated from 12 CITES permitted farms in Arkansas, Louisiana, Missouri and Mississippi (Boundy pers. comm. 2019). Branch of Permits in the Office of Management Authority has noted that they do not explicitly label these as captive-bred or captive-born because they cannot prove lawful acquisition of founder stock (Kanapaux pers. comm. 2019).

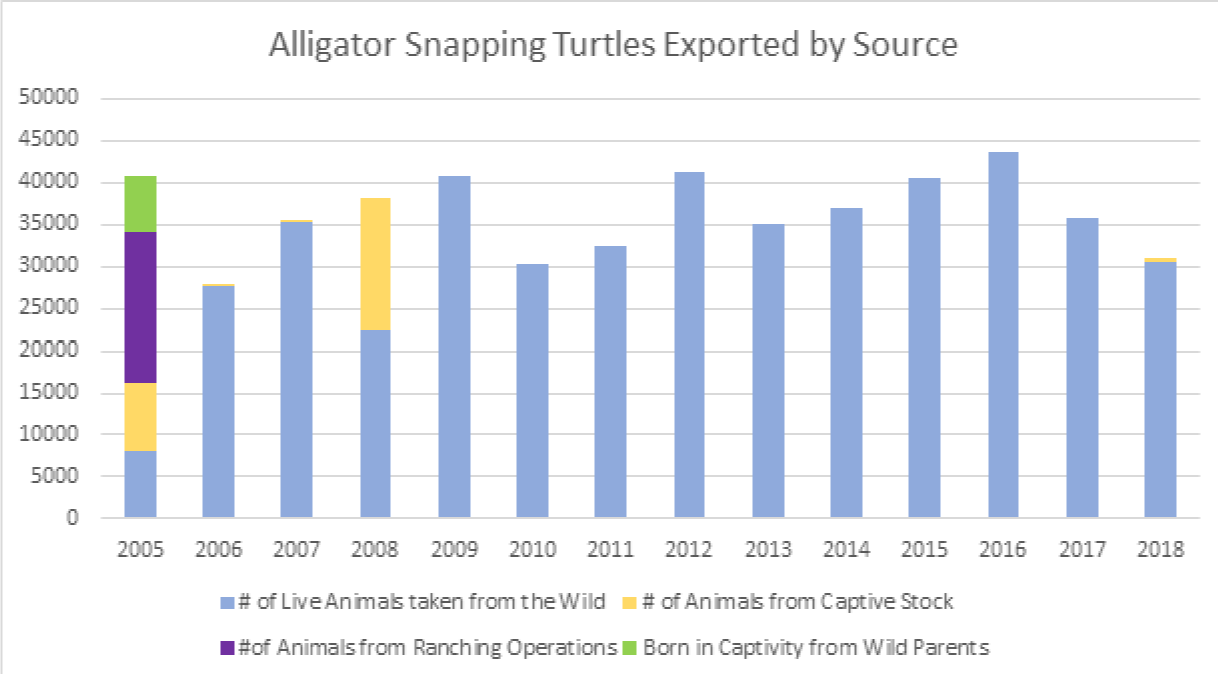


Figure 8. Alligator snapping turtle exports from a variety of sources since 2005. Though most of the turtles have been labelled as wild-caught, nearly all were likely sourced from captivity (USFWS 2018).

Illegal Harvest

There is some evidence of illegal harvest, as well. For instance, in 2017 three men were convicted of collecting 60 large alligator snapping turtles in a single year in Texas and transporting them across state lines violating the Lacey Act (Eastern District of Texas Department of Justice, 2017). While several closed cases involving alligator snapping turtle poaching exist, the extent of current removal from wild populations is also unknown because details of open cases cannot be disclosed due to ongoing investigations.

3.2 Bycatch

Alligator snapping turtles can be killed or harmed incidental to other fishing and recreational activities. Threats include capture as bycatch associated with commercial harvest of other species, ingestion of fish hooks and/or drowning when captured on trotlines (a fishing line strung across a stream with multiple hooks set at intervals) and limb lines (single hooks hung from branches), drowning from entanglement in various types of fishing line, and boat propeller strikes.

Commercial fish (e.g., catfish and buffalo fish [*Ictiobus*]) harvesting may result in adverse impacts to alligator snapping turtles. Commercial hoop nets are often completely submerged when set. Drowning can occur when the netting mesh size limits escape of alligator snapping turtles or they are unable to escape through the mouth of the trap (Frazer et al. 1990, p. 1151). To date, no data exist quantifying the number of alligator snapping turtles lost to commercial hoop nets, but Amity Bass (Louisiana Department of Wildlife and Fisheries [LDWF] biologist) expressed the opinion in an interview that the loss of alligator snapping turtles to commercial hoop nets is likely a significant threat.

Alligator snapping turtles ingest fish hooks incidentally, and depending on where ingested hooks lodge in the digestive tract, they can cause harm or death (Enge et al. 2014, p. 40-41). Fishing line attached to hooks can cause digestive blockage (Enge et al. 2014, p. 40-41). Twenty-five alligator snapping turtles were captured and radiographed between 2011 and 2013 from the Suwannee River (Enge et al. 2014, entire). Of these, three had fish hooks lodged in their gastrointestinal tracts; one of these turtles had three hooks embedded (Figure 9; Enge et al. 2014, p. 25, 28). On the Santa Fe River, a tributary to the Suwannee River, 4 of 11 radiographed turtles had hooks lodged in their upper digestive tracts (Enge et al. 2014, p. 40-41). Some of the ingested hooks might have come from limb lines intended to catch catfish. Surveys for limb lines at two sites along the Santa Fe River found 41 and 28 total limb lines in June and September 2013, respectively (Enge et al. 2014, p. 25, 28). In Florida, limb lines and trotlines are required to be labeled with the angler’s name and contact information, but most of the hooks observed during these surveys were not labeled (Enge et al. 2014, p. 40-41).

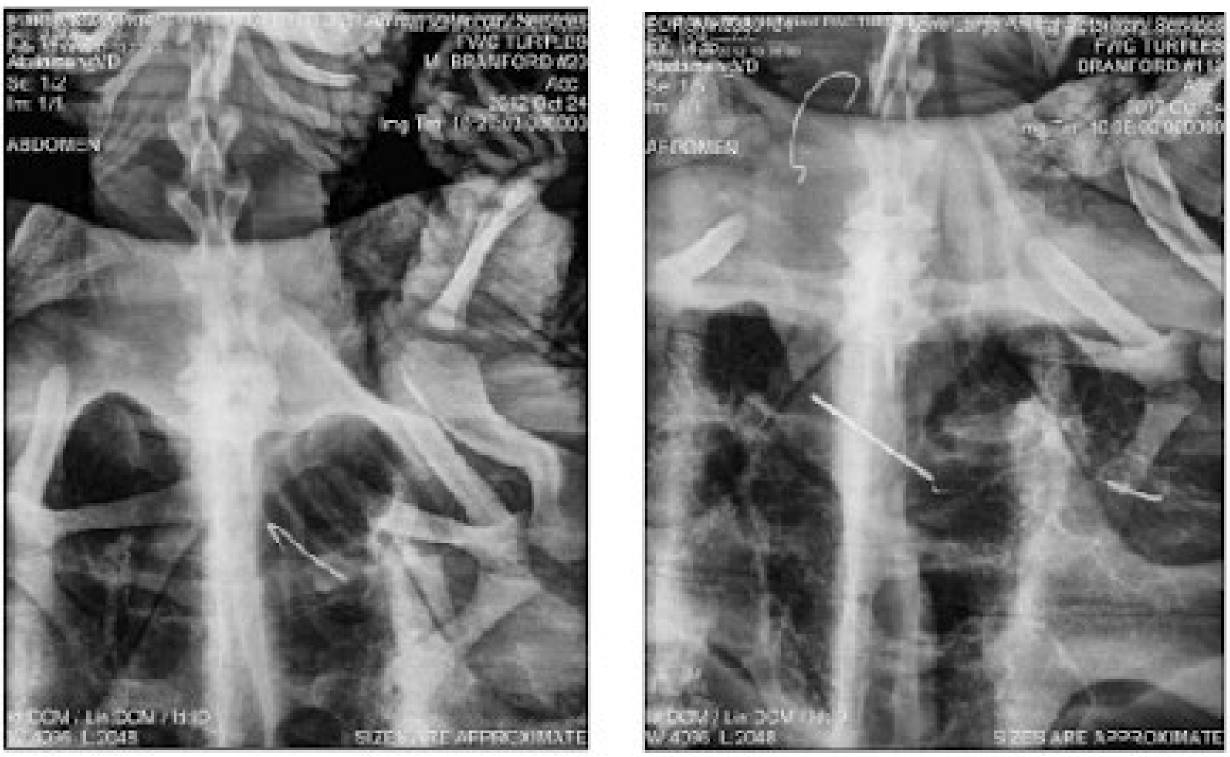


Figure 9. Radiographs of fishing hooks ingested by alligator snapping turtles. Photos from Enge et al. 2014, p. 32.

Trotlines are a threat to alligator snapping turtles; two marked turtles were caught and released by anglers on trotlines during the study by Enge et al. (2014, p. 40-41). Mortality of alligator snapping turtles caught on trotlines has also been observed in Oklahoma on lines that had seemingly been abandoned for a long time, and were thus illegal (Moore et al. 2013, p. 145). In Kansas, the most recent record of an alligator snapping turtle was one found alive caught by a trotline (Shipman 1993, p. 5). Damage caused by boat propellers can also injury alligator snapping turtles and cause extensive damage to their carapaces, though effects on population demographic rates are unknown (Figure 10) (Enge et al. 2014, p. 41).

Figure 10. Carapace damage presumably from boat propellers. Photos from Enge et al. 2014, p. 41.

3.3 Habitat Alteration

Alligator snapping turtle aquatic and nesting habitats have been altered by a number of anthropogenic disturbances. Dams change the hydrology of streams and could impede dispersal and genetic interchange for this highly aquatic species, but impoundments can also provide habitat for the species (Pritchard 1989, p. 84). Other activities and processes that can alter habitat include dredging, deadhead logging, removal of riparian cover, channelization, stream bank erosion, siltation, and land use adjacent to rivers (e.g., clearing land for agriculture). Deadhead logs and fallen riparian woody debris, where present, provide refugia during low-water periods (Enge et al. 2014, p. 40), resting areas for all life stages (Ewert et al. 2006, p. 62), and important feeding areas for hatchlings and juveniles. These activities are assumed to influence habitat suitability for alligator snapping turtles based on their habitat needs, but actual impacts of these processes on alligator snapping turtles have not been quantified.

3.4 Nest Predation

As described in Chapter 2, nest predation rates for the alligator snapping turtle are high. The most common nest predators are raccoons, but nests may also depredated by nine-banded armadillos, Virginia opossums, bobcats, and river otters. In addition to mammalian predators, invasive red imported fire ants pose a threat to alligator snapping turtle nests (Pritchard 1989, p. 69). Predation by fire ants was the suspected cause of nest failure in seven of 16 naturally incubated nests (in contrast to artificial nests) at Black Bayou Lake in Louisiana (Holcomb 2010, p. 51). Beyond nest failure, some hatchlings that did emerge were observed to have wounds inflicted by fire ants, including the loss of a limb or tail, which can lessen their chance of survival (Holcomb 2010, p. 72).

Hatchling mortality due to mammalian nest predation can be mitigated by either protecting nests in their natural setting by installing predator exclusion structures, or by head-starting nests, where eggs are incubated and hatched in captivity before releasing juveniles back into the wild. Hatchling mortality due to fire ants and other insects may also be mitigated by head-starting nests.

3.5 Nest Parasitism

In 2008, one of five alligator snapping turtle nests investigated in Louisiana was infested by the phorid fly *Megaselia scalaris*, the first documentation of infestation by fly larvae in alligator snapping turtles and for the family Chelydridae (snapping turtles; Holcomb and Carr 2011*b*, entire). This species of fly uses a variety of substrates for laying eggs; once the larvae emerge, they consume available organic material. Small holes in the eggs, misshapen eggs, fly puparia (hardened larval exoskeleton), and adult flies inside of eggs were found in the nests, along with remains of turtle hatchlings (Holcomb and Carr 2011*b*, p. 428). It appeared that the infestation played a significant role in the failure of the nest. While phorid flies can have a devastating effect on individual nests, it is unknown what impact this threat has at the population or species level.

3.6 Climate Change

Climate change might impact the alligator snapping turtle in several ways, including loss of habitat to sea level rise for those populations near coastal areas, impacts of drought on habitat and water availability, and physiological impacts on sex determination. In the southeastern United States, temperatures are predicted to warm by 4° to 8° F (2.2° to 4.4° C) by 2100 (Carter et al. 2014, p. 399). In the southern Great Plains (e.g., Texas and Oklahoma), increased temperatures and longer dry spells are predicted (Shafer et al. 2014, p. 445). In the Midwest, the northernmost portion of the alligator snapping turtle range, models predict warming of 5.6° to 8.5° F (3.1° to 4.7° C) by 2100, increased spring precipitation, and decreased summer precipitation (Pryor et al. 2014, p. 420, 424).

Alligator snapping turtles exhibit temperature dependent sex determination, and the relationship between temperature and sex determination has been investigated in laboratory settings (Ewert and Jackson 1994, entire). Male-biased sex ratios were associated with cool nests, and warm nests produced female-biased sex ratios (Figure 11). In addition to temperature effects on sex ratio, temperature was associated with nest viability, which was highest in nests with intermediate sex ratios (produced at intermediate temperatures) and lowest in nests with female-biased sex ratios (produced at warmer temperatures; Ewert and Jackson 1994, p. 28-29). Thus, warming temperatures might lead to alligator snapping turtle nests with strongly female-biased sex ratios and declining viability. These impacts could be exacerbated in human-altered areas that are warmer than surrounding natural areas.

Climate conditions also appear to limit the distribution of alligator snapping turtles. Ecological niche modeling has indicated that the distribution is limited by low precipitation on the western edge of the range, and by temperature along the northern edge of the range (Thompson et al. 2016, p. 431-432). At these northern limits of the range, adult alligator snapping turtles can survive, but they face constraints on reproduction imposed by the influence of temperature on embryonic development (Thompson et al. 2016, p. 431-432). A warming climate could shift the suitable range of the species farther north as northern latitudes become able to meet the incubation temperature needs of alligator snapping turtles.

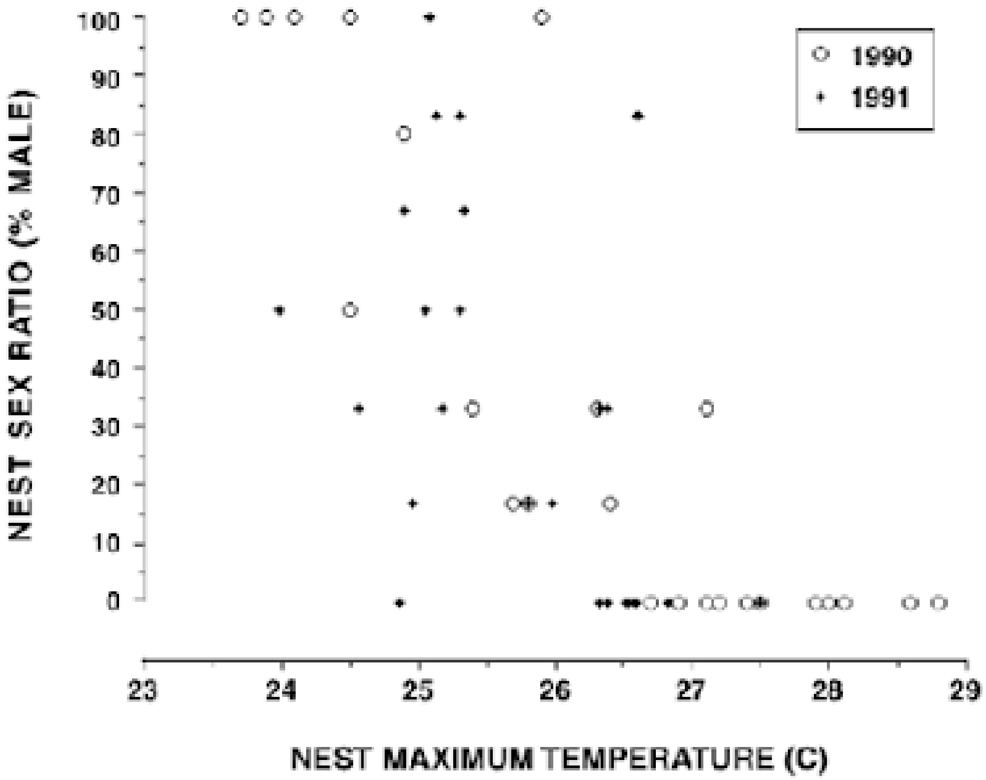


Figure 11. Hatchling sex ratios in nests of alligator snapping turtles in relation to nest temperature. Figure from Ewert and Jackson 1994, p. 26.

3.7 Disease and Health

Chaffin et al. (2008, entire) captured and assessed the health of 97 free-ranging alligator snapping turtles across nine sites in northwestern Florida and southwestern Georgia between 2001 and 2006. Assessed alligator snapping turtles had shell abnormalities, including worn, cracked, or broken scutes (n = 19), fresh or healed wounds resulting from trauma (n = 15), missing portions of the tail (n = 12), missing portions of the beak (n = 1), missing portions of claws (n = 1), and leech infestation (n = 46; Chaffin et al. 2008, p. 674). Protozoan parasites (*Haemogregarina*, species unknown), transmitted by leeches, were found in all but one turtle assessed. The team checked for infectious pathogens known to impact reptiles and found no evidence for exposure to West Nile virus, *Mycoplasma agassizii*, or ranavirus (Chaffin et al. 2008, p. 677). Exposure to herpes (HV1976, HV4295/7R/95) was indicated for 64% (7 out of 11) of alligator snapping turtles tested from Pataula Creek, Georgia. None were showing symptoms, and alligator snapping turtles likely co-evolved with a species-specific herpesvirus, but it is possible that exposure to stress could lead to an outbreak of herpes in these populations (Chaffin et al. 2008, p.677).

Mercury was detected in the blood in 93% of samples, which varied between 0.010 ppm and 1.840 ppm, and mercury was possibly sourced from atmospheric deposition and/or bioaccumulation through prey (Chaffin et al. 2008, p. 672). Mercury transferred by mothers to eggs is associated with decreased fertilization rates and proportion of eggs that hatch. Mercury is associated with increased embryonic mortality in common snapping turtles

(Hopkins et al. 2013, p. 2418-2419), but the levels of mercury detected in alligator snapping turtles were low relative to those detected in the common snapping turtle study and are unlikely to have very large effects on reproduction. More direct exposure to environmental mercury that leads to higher mercury levels in alligator snapping turtles would be expected to impact reproduction as well as other aspects of health.

3.8 Conservation Measures

3.8.1 Captive Rearing, Head-Starting, and Reintroductions

In this section, we describe conservation measures that have been implemented for the alligator snapping turtle including captive rearing, head-starting, and reintroductions. Head-starting refers to incubating and hatching eggs in captivity, retaining hatchlings in captivity during the time they would be most vulnerable in the wild, and subsequently releasing them into the wild as older juveniles when they are more likely to survive.

A captive breeding program at Tishomingo National Fish Hatchery in Oklahoma was initiated in 1999 to produce head-started alligator snapping turtles for reintroduction (Riedle et al. 2008*a*, p. 25). In 2007, 249 adult turtles (confiscated from a turtle farm in violation of its permits) and 16 juveniles (from Tishomingo National Fish Hatchery) were released into seven sites in southern Oklahoma, and follow-up monitoring occurred during May-August in 2007 and 2008 (Moore et al. 2013, p. 141). There were only seven confirmed instances of mortality, all within the first year after release, resulting from drowning on trotlines, a gunshot wound, and other suspicious circumstances (Moore et al. 2013, p. 144). When viable nests were found during follow-up surveys, they were covered with a mesh predator exclusion device. Only one viable nest was found during 2007 or 2008, while 25 depredated nests were found, which nevertheless indicates that released adults survived and were reproducing (Moore et al. 2013, p. 144).

From 2008 to 2010, 246 head-started juveniles (3 to 7 years old) were released in the Caney River in northeastern Oklahoma and were monitored until 2012 (Anthony et al. 2015, p. 44). Mean annual survivorship post-release was estimated to be 59%, 70%, and 100% for turtles aged 3, 4, and 5 at release, respectively (older turtles were not included in analysis due to low sample sizes) (Anthony et al. 2015, p. 46).

Head-starting, reintroduction, and monitoring of alligator snapping turtles were conducted between 2014 and 2016 in Illinois, Louisiana, and Oklahoma (Dreslik et al. 2017, entire). Released turtles included head-started juveniles, confiscations by law enforcement, classroom turtle rearing programs, and other captive breeding programs (Dreslik et al. 2017, p. 6, 13). Across three states (one site each in Oklahoma and Illinois, two sites in Louisiana), 548 turtles were released, the majority of which (465) were head-started at the Tishomingo National Fish Hatchery in Tishomingo, Oklahoma, and 372 of these were tracked using radio-telemetry (Dreslik et al. 2017, p. 22). Between 21.7% and 28.8% of released juveniles were confirmed dead within the first year, primarily from predation by raccoons, while 35.6% to 54.2% experienced radio transmitter failures and could not successfully be tracked (Dreslik et al. 2017, p. 19). The greatest predictors of survival for released juveniles were size at release, age, and time of year. Larger, older turtles had higher survival rates than smaller, younger turtles, and survival was lower over winter than other seasons (Dreslik et al. 2017, p. 22-25).

Survival rates from post-release monitoring were used in a series of stochastic population viability models that assessed different introduction scenarios that varied in the number of turtles released, the age classes released, and the number of release years (Dreslik et al. 2017, p. 28-33). For all modeled scenarios, reintroduced populations were expected to become extirpated after releases ceased, though varying the listed parameters could lengthen the amount of time to extirpation, and a 30% reduction in mortality across all age classes was needed to achieve population stability (Dreslik et al. 2017, p. 33). Based on these models, the authors conclude that reintroduction could have limited utility for conservation of alligator snapping turtles without other conservation efforts to increase survival rates (Dreslik et al. 2017, p. 41). Releasing adults rather than juveniles would also likely lead to improved outcomes but would bring additional logistical challenges of housing and caring for the turtles to an older age before release.

It is important to communicate that no conservation measures are likely to be effective in securing the viability of the alligator snapping turtle if the underlying causes of declines are not first addressed. Protection from the threats listed earlier in this chapter is crucial if head-starting and reintroductions are to be successful.

3.8.2 Integrated Natural Resource Management Plans

As part of the implementation of the Sikes Improvement Act (1997), the Secretaries of the military departments are required to prepare and implement integrated natural resource management plans (INRMP) for each military installation in the United States. Of the military installations with confirmed presence of alligator snapping turtles, substantial variability exists in direct management for this species. Many INRMPS have just documented presence on the installation (e.g., Little Rock Air Force Base [AFB] [USAF 2013, p. 5-15 to 5-16] and Robinson Maneuver Training Center [USANG 2018, p. 2-31; L-1] in Arkansas; Moody AFB [USAF 2014, p. 46-47] in Georgia; Naval Air Station [NAS] Joint Reserve Base New Orleans [USN 2012, p. 3-25] in Louisiana; and Eglin AFB, NAS Whiting Field Complex, and NAS Pensacola Complex in Florida). One INRMP references specific management for the species guided by the state wildlife action plan (i.e., Fort Chaffee [Arkansas] [USANG 2018, p. 120]), one states that project design considers state listed species and has best management practices in place for all activities (i.e., Red River Army Depot [Texas] [USA 2018, p. 48]), and one contains specific reference to activities being consistent with maintenance of reference stream conditions or offers direct measures to enhance habitat for this and other rare species (e.g., Ft. Benning [Georgia], [USA 2015, p. 28 and 209-210]). Among the measures employed at the latter base are invasive species management and additional restoration of upland habitat (e.g., tree planting). At this installation it appears that training and management are consistent with continued maintenance of intact and fully-functional systems where this species occurs. Additionally, in one case, while no specific reference to the species is made in the INRMP, the INRMP for Barksdale Air Force Base (Louisiana) (USAF 2017, p. 29) states, “Any state rare animals located on the installation will be protected to the extent practical. If state rare species are located on the installation, and protection is not practical, discussions with the state will be initiated to develop a documentation or management strategy.”

Several other installations in the range could have the species, but presence has not yet been documented at these installations. Among these are Maxwell AFB and NAS Whiting Field in Alabama; Pine Bluff Arsenal in Arkansas; Camp Beauregard Training Site, Camp Minden Training Center, and Camp Villere in Louisiana; Camp McCain and NAS Meridian in Mississippi; Hurlburt Field and Tyndall AFB in Florida; and McAlester Army Ammunitions Plant and Camp Gruber Maneuver Training Center in Oklahoma.

3.9 Summary of Factors Influencing Viability

Historically, extensive commercial and recreational take in the last century resulted in significant declines to many alligator snapping turtle populations. Commercial harvest depleted populations in Louisiana, Florida, Georgia and Alabama and is now prohibited in all states within its range. Recreational harvest of alligator snapping turtles is prohibited in every state except for Louisiana and Mississippi. Although regulatory harvest restrictions have decreased the number of alligator snapping turtles being harvested, populations have not necessarily increased in response. This lag in population response is likely due to the demography of the species, specifically delayed maturity, long generation times, and relatively low reproductive output.

Currently, the primary negative influences on viability of alligator snapping turtles are: legal and illegal intentional harvest (including for export), bycatch, habitat alteration, and nest predation. Climate change and disease might negatively influence the species, but the impacts of these on the species are more speculative due to a lack of information. Conversely, conservation measures that have been implemented for the alligator snapping turtle include captive rearing, head-starting, and reintroductions, as well as various efforts to restore and improve habitat.

CHAPTER 4 – POPULATION AND SPECIES NEEDS AND CURRENT CONDITION

In this chapter, we first discuss how we describe populations and species needs and how we delineated representative units and analysis units within the range of alligator snapping turtles. Then we describe how we collected information to assess resilience, and we summarize the current resilience of each analysis unit along with the redundancy and representation for the species.

4.1 Population Needs

For populations to persist, the needs of individuals (Table 1) must be met at a larger scale. These include nesting habitat (appropriate structure and substrate, location near water, temperature); habitat for hatchlings, juveniles, and adults (e.g., smaller streams for juveniles, deeper water for adults, with structure for refugia); food; and mates. These individual needs must be met within an area of habitat that can support enough alligator snapping turtles to survive, find mates, and reproduce while avoiding inbreeding depression. To persist, populations must be robust in size not only to avoid genetic effects from inbreeding, but also to provide resilience against stochastic demographic and environmental events. Later in this chapter we describe how we used abundance estimates and information about threats affecting abundances to describe resilience of analysis units (rather than populations, see Section 4.4) of alligator snapping turtles.

4.2 Species Needs

For the species to be viable, alligator snapping turtles require redundancy and representation of resilient populations or analysis units. Redundancy of resilient populations distributed across the species' range is necessary to buffer the species against the effects of catastrophic events on any single population or grouping of populations. Potential catastrophic effects that could eliminate or severely reduce population resilience include, but are not limited to large-scale destruction of nesting or river habitat from river engineering projects, drought, hurricanes, and chemical spills.

Representation refers to the breadth of genetic and environmental diversity within and among populations that contributes to the ability of the species to respond and adapt to changing environmental conditions over time. Maintaining resilient populations across the range of variation within the species will increase the amount of variation within the species on which natural selection can act, increasing the chances that the species will persist in a changing world. Our approach for defining and delineating representation for alligator snapping turtles is described in the following section.

4.3 Representative Units

In order to determine the representation across the range of the species, we used a tiered approach and delineated five representative units: Western, Southern Mississippi, Northern Mississippi, Alabama, and Apalachicola (Figure 12).

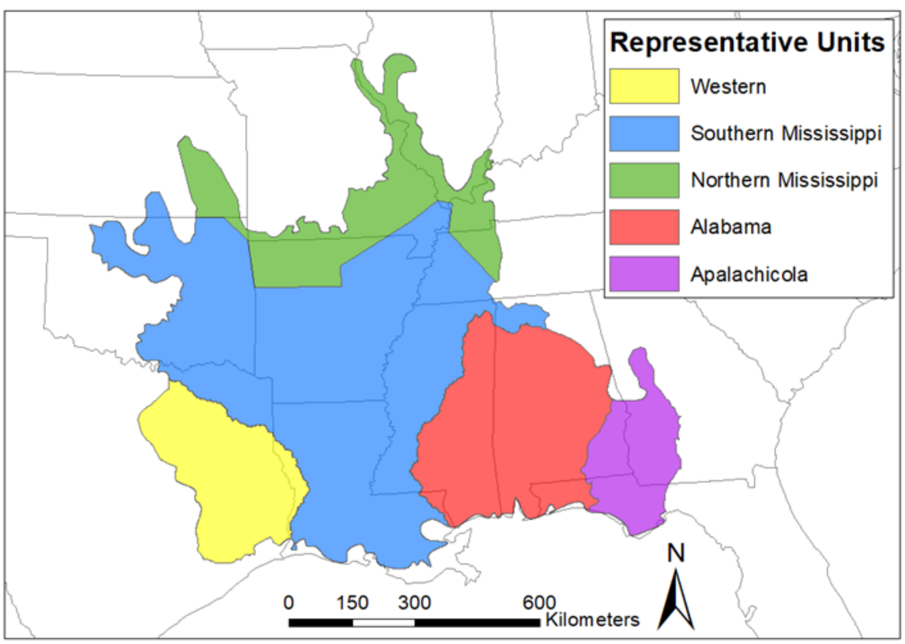


Figure 12. Alligator snapping turtle representative units.

At the coarsest scale, we divided the species' range into two parts corresponding to two proposed distinct genetic lineages (Thomas et al. 2014, p. 147, 152-154). This separated out the Apalachicola representative unit, while grouping the remaining four representative units to the west into the same lineage.

Because of the large geographic extent of the large western lineage, it was further divided to reflect genetic variation from east to west. Alligator snapping turtles are highly aquatic; movement and connectivity occur primarily via waterways (as opposed to over land), leading to genetic structuring among different drainages (Echelle et al. 2010, p. 1381-1382; Roman et al. 1999, p. 138). Based on these genetic studies, the aquatic dispersal mode of the species, and input from species experts, we further divided the larger western lineage into three units: the Mississippi River drainage, and a unit each to the east and to the west of the Mississippi River drainage.

The final tier of our strategy for delineating representative units was based on differences in ecology and life history rather than genetics. We split the Mississippi River drainage into a northern and southern unit. There have not been rigorous genetic studies to investigate genetic differences along a north-south gradient, but ecological differences do exist that likely lead to differences in genetic composition and adaptive capacity. Life history strategies vary latitudinally, and turtles in general produce larger clutches and smaller eggs in more northern latitudes compared to smaller clutches of larger eggs at more southerly latitudes (Iverson et al. 1993, p. 2449-2451). Differences in temperature latitudinally can also lead to differences in the timing of nesting. Thompson et al. (2016, p. 429) created a climate model that mapped suitable conditions for incubation and hatching under different nest initiation dates from May 1 to June 15 (Figure 13). In the southern portion of the species' range, there were no limitations to nest initiation dates. Farther north in the species' range (e.g., north-central Arkansas, Missouri, Illinois, Tennessee), limitations were indicated; alligator snapping turtles need to nest by early to mid-May to allow for enough warm days for complete development and hatching of the young.

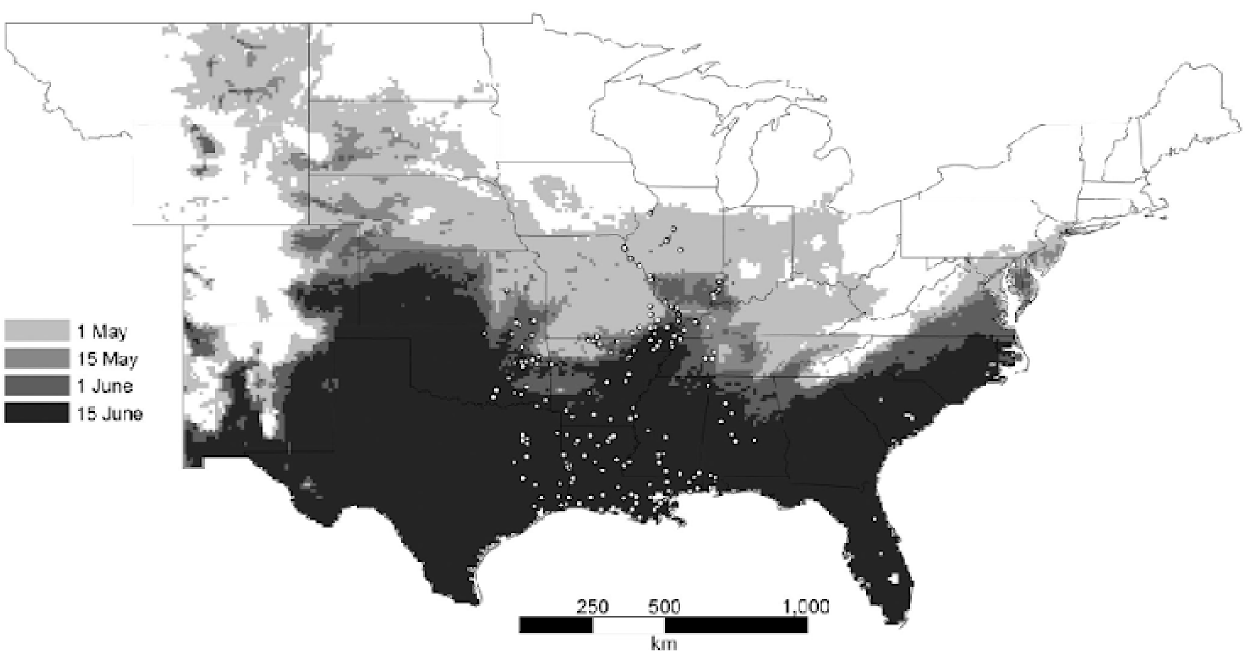


Figure 13. Areas predicted to be suitable for complete embryonic development of alligator snapping turtle eggs based on number of suitable degree days under four nesting scenarios with different nesting initiation dates: 1 May, 15 May, 1 June, and 15 June (dots are alligator snapping turtle occurrences) (Figure and caption from Thompson et al. 2016, p. 429).

We used the results from Thompson et al. (2016, p. 429) and spatial data depicting growing degree days (Matthews et al. 2018, p. 6) to determine the separation between the northern and southern Mississippi representative units. We note that the change in temperature from south to north is a gradient and does not occur abruptly at the border between the two units. Even though a true distinct boundary does not exist on the landscape between the two units, it is still important to acknowledge in the structure of our representative units that differences exist in habitat and the thermal environment between alligator snapping turtles in the southern reaches of the Mississippi drainage and those farther north. These differences in selective pressures likely lead to unique adaptations for the different conditions, and the loss of either the northern or the southern Mississippi unit would represent a significant loss in the diversity and adaptive capacity of the species.

4.4 Analysis Units

We divided the species' range into seven analysis units, nested within representative units, to assess resilience (Figure 14). These analysis units are not meant to represent "populations" in a biological sense; they do not represent groups of demographically linked interbreeding individuals. Delineating biological populations of the alligator snapping turtle is not feasible at this time because of the large spatial extent of the geographic range and the patchy availability of relevant information across the entire range. Rather, these units were designed to subdivide the species' range in a way that facilitates assessing and reporting the variation in current and future resilience across the range.

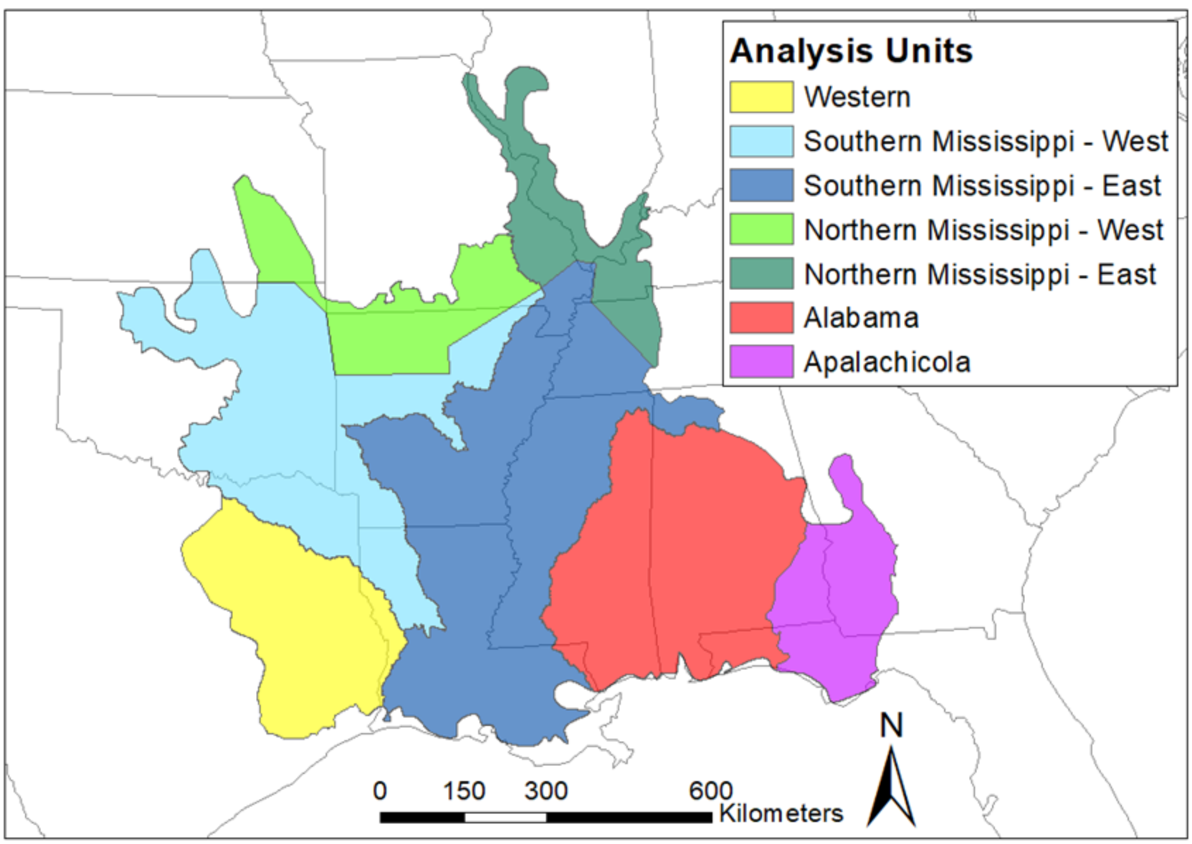


Figure 14. Alligator snapping turtle analysis units. The two Southern Mississippi units (blues) make up one representative unit and the two Northern Mississippi units (greens) make up one representative unit; the remaining analysis units each make up a single representative unit.

Subdivision of representative units into analysis units was based primarily on Hydrologic Unit Code (HUC) 2 watershed boundaries (Figure 15). When small fragments of a HUC were adjacent to larger HUCs in the same representative unit (e.g., a small sliver of a new HUC on the eastern edge of the Southern Mississippi representative unit), or where small portions of multiple HUCs combined (e.g., at the convergence of Missouri, Illinois, Indiana, Kentucky, and Tennessee), we grouped them into larger units to prevent having very small analysis units of a vastly different size than the others. Including very small analysis units in these cases would have posed challenges for collecting data from species experts for the current and future resilience assessment and would not be very informative for the overall status assessment of the species.

In creating analysis units in this way, we strove to balance the needs to: a) have units small enough to be able to capture the variation in the condition of the species (e.g., abundance, threats) across its range, while also b) retaining units large enough that species experts would be able to summarize information about the condition of the species for every unit. Using this strategy, the Western, Alabama, and Apalachicola representative units each contained a single analysis unit (representative unit = analysis unit), while the Southern Mississippi and Northern Mississippi representative units were each divided into an eastern and western analysis unit.

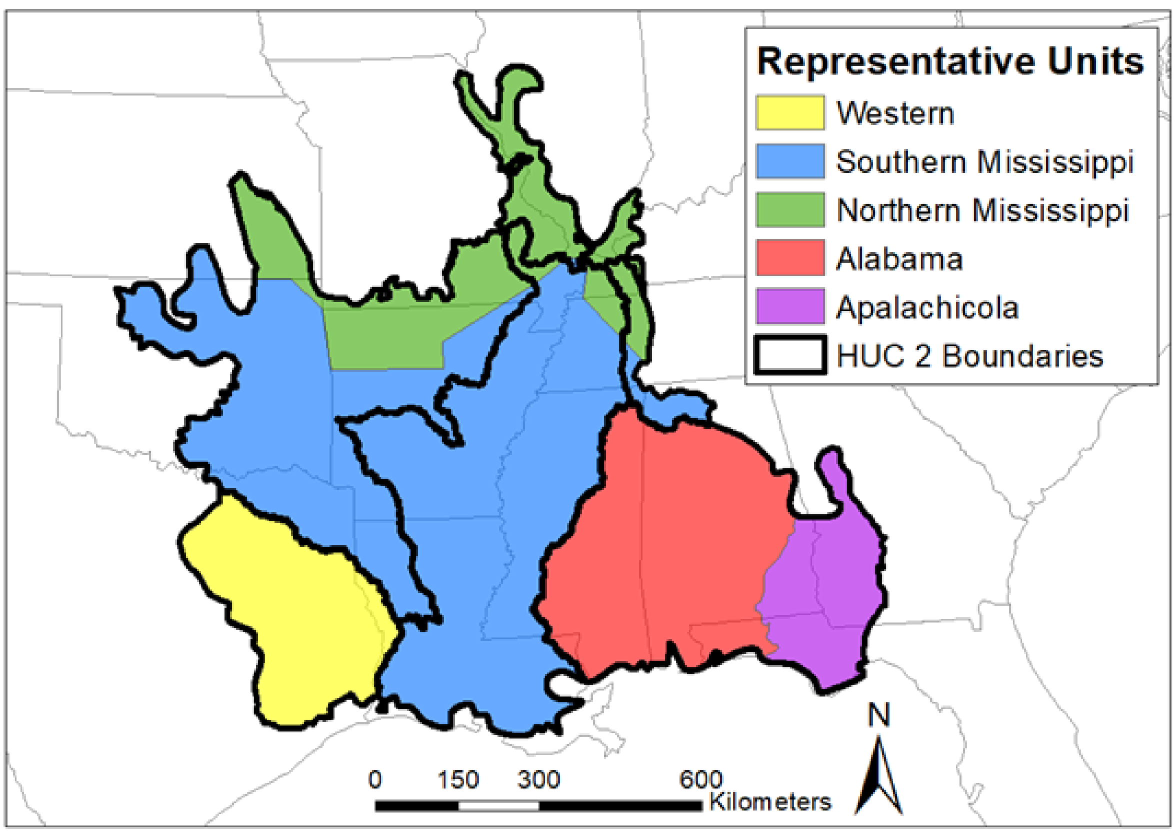


Figure 15. HUC 2 watershed boundaries within alligator snapping turtle representative units.

4.5 Current Condition Methods

To assess the current condition of the alligator snapping turtle, we surveyed species experts about current abundance, current threats, and a comparison of the current and historical distribution. We used an elicitation questionnaire sent to species experts to gather this information. The questionnaire included questions about alligator snapping turtles and impacts of influencing factors on their populations at both the range-wide scale and the analysis unit scale (the elicitation questionnaire can be found in Appendix C). The questionnaire was sent to 32 species experts after they viewed a webinar explaining the types of questions they would encounter and how their responses would be used. These experts were spread geographically throughout the species' range and collectively had many decades of experience working with alligator snapping turtles.

Current abundance is our measure for current resilience, along with information about current threats, conservation actions, and distribution serving as auxiliary information about the causes and effects of current versus historical abundances. For information about abundance, threats, and conservation actions that we elicited from species experts, "current" refers to the year 2019; for species distribution records, "current" refers to the years 2000-2019.

4.5.1 Current Abundance

We compared the historical and current ranges of alligator snapping turtles by querying state biologists or those with access to the state's natural heritage program data. To obtain estimates of abundance for each analysis unit, we used expert elicitation, using a 4-point elicitation procedure in a written questionnaire (Speirs-Bridge et al. 2010, p. 515). Experts of both *M. temminckii* and *M. suwanniensis* were asked to respond only for those analysis units for which they have experience or expertise. In this procedure, experts were asked what they estimated to be the lowest likely number, the highest likely number, and the most likely number of alligator snapping turtles in each analysis unit. They were then asked to report how confident they were that their interval (lowest estimate to highest estimate) captured the actual number of alligator snapping turtles (akin to a confidence interval). Finally, the experts were asked to describe how they generated their estimates.

For *M. temminckii* and *M. suwanniensis* combined, we received elicitation questionnaire responses from 14 species experts out of 32 queried for an overall response rate of 43.75%. For *M. temminckii* we had a total of 18 analysis unit-specific responses (one to four responses per analysis unit). Only 9 of those 18 responses included estimates of current abundance (one response for each analysis unit except for two responses for the Apalachicola Analysis Unit and Northern Mississippi – East Analysis Unit). Despite the large amount of expertise in the expert team we queried, there was a high degree of uncertainty about current abundances in each analysis unit. This uncertainty was sometimes expressed in non-responses (i.e., expert did not feel comfortable providing any estimates because they were too uncertain), and at other times was expressed as a large range between the low and high-end estimates, with relatively low confidence that the true value lies between those bounds.

In addition to analysis-unit-specific abundances, we also asked about overall density patterns across the species' range, specifically whether there are geographic patterns, and what factors seem to correlate with density. Experts responded that abundance and densities are probably higher in the south compared to the northern parts of the species' range, where populations are often small and isolated. Experts also expect that densities are likely lower in areas with either a more recent history of commercial or recreational harvest of alligator snapping turtles (more harvest pressure historically in the western part of the range [Louisiana, Arkansas, Mississippi], than the eastern [Florida, Georgia]), or more robust fisheries for other species that could be associated with increased incidental capture of alligator snapping turtles. Densities are also likely tied to habitat, with higher densities where there is more structure (e.g. sunken logs, undercut banks), available nesting habitat, and fewer nest predators.

4.5.2 Current Threats and Conservation Actions

We also elicited information about the prevalence of negative and positive influences on alligator snapping turtles in each analysis unit. Using the same 4-point elicitation format, we asked the species experts to estimate the extent of occupied area in each analysis unit where alligator snapping turtles are exposed to each of the following threats: incidental hooking on trot and limb lines, commercial fishing bycatch, legal collection or harvest, illegal collection or harvest (poaching), and nest predation by subsidized or non-native predators. In addition, we asked experts to describe and estimate the spatial extent of any other threats known to occur in their analysis units, as well as any conservation actions that are being implemented.

Because some experts have expertise in and responded for multiple analysis units, we received a total of 18 analysis unit-specific responses (one to four responses per analysis unit, with varying numbers of questions answered).

In addition to asking the expert team about the spatial extent of different threats in each analysis unit, we also asked about the demographic impact of different threats range-wide. We used 4-point elicitation to ask what effect commercial bycatch, incidental hooking, hook ingestion, legal harvest, illegal harvest, and nest predation have on the survival of relevant life stages (adults, juveniles, hatchings, nests) in areas where the threat occurs (Figure 16). We received usable responses from 10 experts, with varying numbers of questions answered by each. Legal and illegal harvest, where they occur, were estimated to have the highest impact on adult survival rates, with both causing reductions in survival of 18% (most likely estimate). Commercial and recreational bycatch and hook ingestion were estimated to have lower impacts on adult survival, with most likely reductions in survival of 7-9%. The estimated impacts of threats on juvenile survival were lower than impacts to adult survival with most likely impacts of a 6-8% reduction in survival where commercial bycatch, incidental hooking, and hook ingestion occur, and a 6-7% reduction in survival from legal and illegal harvest where they occur. Hatchlings are not estimated to be heavily impacted by any of the threats we explored. Nest survival is estimated to be heavily impacted by nest predation by subsidized or non-native predators (e.g., raccoons, fire ants), with a most likely estimate of 58% reduction in survival.

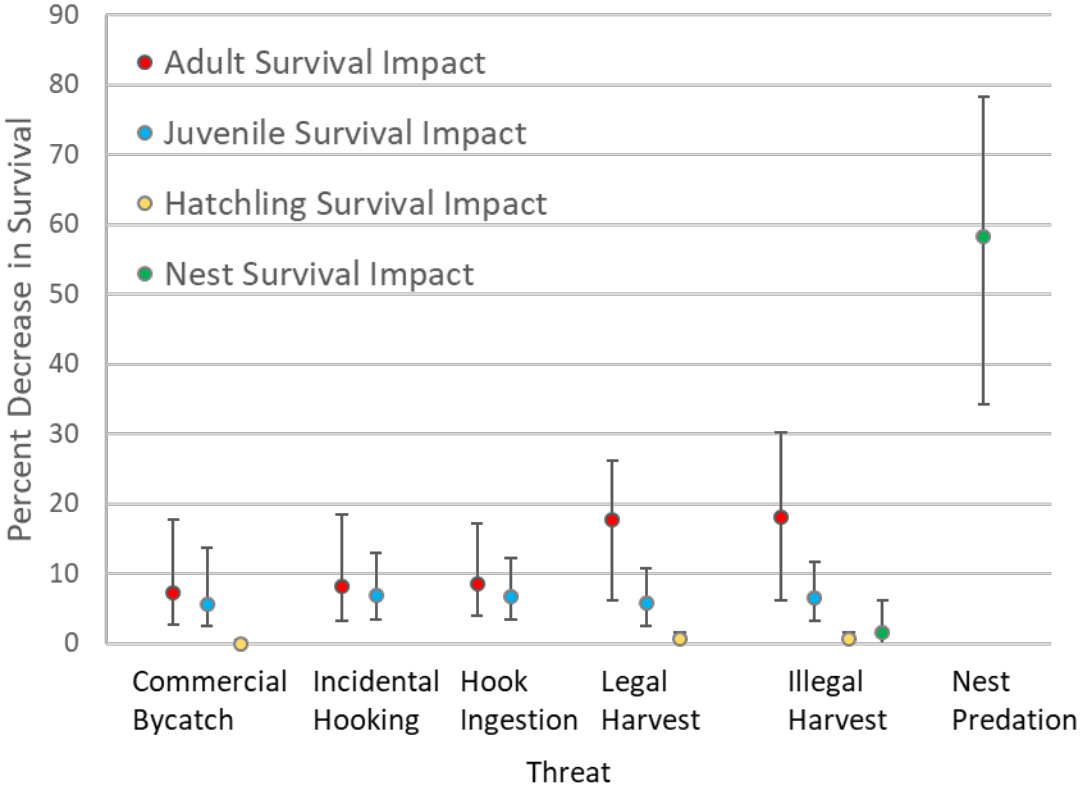


Figure 16. Expert-elicited magnitude of threats facing alligator snapping turtles in terms of the percent decrease to survival rates. Error bars indicate the average of lowest likely and highest likely estimates of impacts on survival, while circles indicate the average of most likely impacts on survival. The number of respondents for each metric ranged from 4 to 7.

4.5.3 Comparison with Historical Range

We compared the historical and current ranges of alligator snapping turtles by querying state biologists or those with access to the state's natural heritage program data. For each county or parish in their state, we asked for the current and historical status, and the date of the last confirmed record of alligator snapping turtles. For this exercise (in contrast to expert elicitation about current abundance, threats, and conservation actions), "current" referred to the time period from the year 2000 to the present (2019). For each county and time period (current and historical), alligator snapping turtle occupancy was classified as either occupied, not occupied, or unknown (Table 2). Respondents were also asked to describe, if known, the reasons behind any changes in occupancy status from historical to current.

Table 2. Definitions of Occupied, Not Occupied, and Unknown, for characterizing the current (since 2000) and historical (prior to 2000) distribution of alligator snapping turtles by county.

	Current	**Historical**
Occupied	Signifies that alligator snapping turtles are known or presumed to occur in this county now. In the absence of very recent records, currently occupied counties will include those with alligator snapping turtle records since 2000, provided that there is no evidence that the species has been extirpated since those most recent records	Signifies that alligator snapping turtles are known or believed to have occurred in this county prior to 2000
Not Occupied	Signifies that alligator snapping turtles have not been reported in this county since 2000, or if they have, there is evidence that they have been extirpated since then	Signifies that there is no evidence that alligator snapping turtles occurred in this county prior to 2000
Unknown	Signifies uncertainty in the current occupation `of this county by alligator snapping turtles. For example, counties with no recent records as a result of no recent surveys, but with no reason to believe that the species has been extirpated since the last records	Signifies uncertainty in the historical occupation of this county by alligator snapping turtles

4.6 Current Condition Results For Each Analysis Unit

Below, we report the current abundance, current threats and conservation actions, and comparison with the historical range for each of the eight analysis units. All of the information came from expert elicitations unless otherwise specified.

4.6.1 Western Analysis Unit

This analysis unit (Figure 17) encompasses parts of eastern Texas and western Louisiana. Main water bodies that currently or historically supported alligator snapping turtles include the Trinity River, Sabine River, and Neches River.

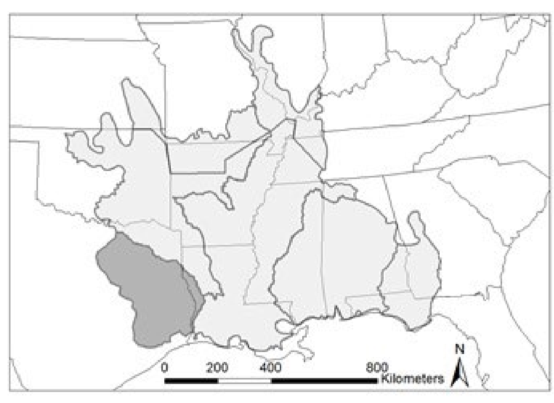


Figure 17. Western Analysis Unit.

Current Abundance

Current abundance in this analysis unit is estimated to be between 1,000 and 100,000 alligator snapping turtles, indicating a high degree of uncertainty resulting from limited monitoring and research. These estimates were extracted from information compiled to complete the NatureServe Conservation Rank Calculator in Texas in 2018, and thus are not associated with a most likely estimate like the expert-elicited values for other analysis units. In the absence of a mean or most likely estimate associated with this range in abundances, we took the center point, 50,500, as the most likely estimate.

<u>Current Threats and Conservation Actions</u>

We received little information about current threats and conservation actions in this analysis unit, but threats include:

- **Incidental hooking**, which is estimated to affect 31-71% of the species' range in this unit.
- **Nest predation**, which is estimated to affect 71-100% of the species' range in this unit.
- **Habitat alteration** via channelization, impoundments, and debris removal, which is estimated to affect 71-100% of the species' range in this unit.
- **Legal harvest** occurs in Louisiana, which makes up 6% of the area of this analysis unit.
- **Illegal harvest**, which occurs in this unit, though the extent and severity of this threat is unknown.

In Texas, which makes up the vast majority of this analysis unit, alligator snapping turtles are protected at the state level and there is no legal harvest.

<u>Comparison with Historical Range</u>

In this analysis unit, there have been no confirmed changes in the species' range (Figure 18). The only changes between historical and current times are changes between occupied status and unknown status in Texas. These changes are due to the *ad hoc* nature of surveys in this unit; there is not presently any evidence that the species has been extirpated from any counties within its historical range. Of the 26 counties in Texas in this unit with confirmed current alligator snapping turtle records, 18 of those were made within the last 10 years (since 2009). In three counties with current unknown status (Franklin, Houston, and Rains), alligator snapping turtles have not been recorded since 1985-1986.

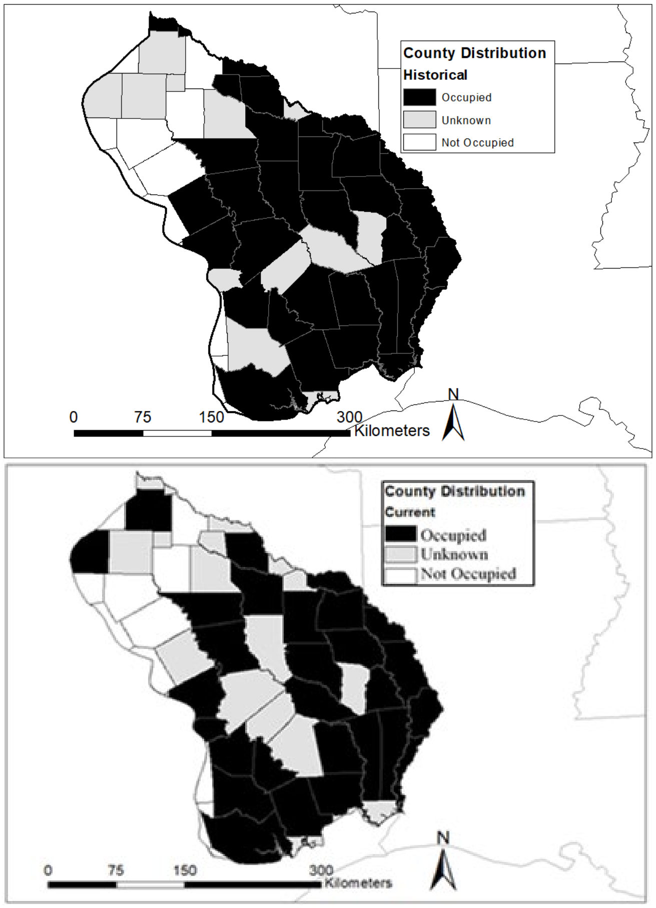


Figure 18. Historical and current distribution of alligator snapping turtles in the Western Analysis Unit.

4.6.2 Southern Mississippi – West Analysis Unit

This analysis unit (Figure 19) encompasses parts of northeastern Texas, Oklahoma, Kansas, Missouri, Arkansas, and northwestern Louisiana. Main water bodies that currently or historically supported alligator snapping turtles include but are not limited to the Arkansas River, Red River, Canadian River, East Fork Cadron Creek, Black Lake Bayou, Cheechee Bay, Saline Bayou, Black Lake, Clear Lake, Saline Lake, Cane River Canal, Black River, Boggy Bayou, Grand Bayou, Crichton Lake, Coushatta Bayou, Smith Island Lake, Loggy Bayou, Bayou Pierre, Wallace Lake, Smithport Lake, and Bayou Lumbra.

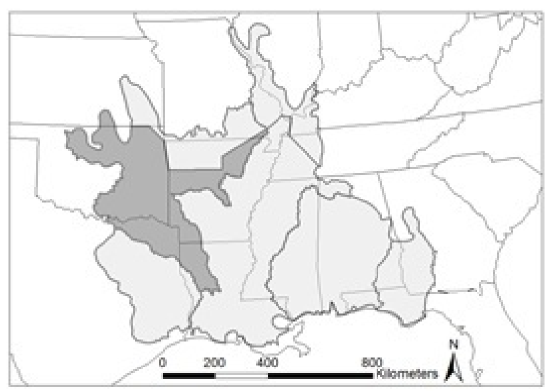


Figure 19. Southern Mississippi – West Analysis Unit

Current Abundance

Current abundance in this analysis unit is estimated to be 15,000 alligator snapping turtles, with 70% confidence that the true abundance is between 1,000 and 50,000. These estimates were based on nearly twenty years of sampling. Densities have been found to vary greatly between river segments in this unit, and populations are highly fragmented by impoundments.

Current Threats and Conservation Actions

Threats in this analysis unit include:

- **Incidental hooking**, which is estimated to affect 80% of the species' range in this unit (80% confidence that the true value lies between 60 and 100%).
- **Illegal harvest**, which is estimated to affect 1% of the species' range in this unit (100% confidence that the true value lies between 0 and 10%).
- **Nest predation**, which is estimated to affect 30% of the species' range in this unit (50% confidence that the true value lies between 10 and 80%).
- **Habitat fragmentation** was also identified as a threat to populations in this unit.
- **Legal harvest** occurs in Louisiana, which makes up 9% of the area of this analysis unit.

With the exception of Louisiana, alligator snapping turtles in this unit are **protected** at the state level with no legal harvest. Other conservation measures include **head-start and release programs** on the Caney, Verdigris, and Neosho river drainages in Oklahoma. The spatial extent and movements of alligator snapping turtles within these drainages are constrained by dams, but releases up and downstream of impoundments are expected to increase spatial extent over time.

Comparison with Historical Range

In this analysis unit, there have been no confirmed changes in the species' range in Louisiana, Arkansas, or the small portion of the unit that extends into Missouri (Figure 20).

In Texas, there have been changes from occupied to unknown status and vice versa, but no contractions of the species' range have been confirmed; the lack of recent records is likely more of an indication of a lack of recent surveys than a lack of alligator snapping turtles.

In Oklahoma, counties with unknown status on the edge of the species' range have had no confirmed records but did contain potentially suitable habitat and were adjacent to occupied counties. Because there are no historical records in these counties, there has been almost no trapping effort there, so the current status remains unknown. There are currently introductions ongoing in the lower Washita River above the Lake Texoma dam (Marshall and Johnston Counties), upper Caney River above the Hulah Reservoir dam (Osage County), and the upper Verdigris River above the Oologah Reservoir dam (Nowata County). These counties are designated as occupied historically, but with unknown current status, because it will not be apparent for many more years whether reintroduction efforts will be successful long-term. In the short term, there has been high survival of adults and larger juveniles (Dreslik et al. 2017, p. 20-21) and documented nesting attempts (Miller et al. 2014, p. 190). Potential range contractions in Oklahoma from the historical distribution to the present are likely the result of habitat modification (i.e., the channelization of rivers, clearing of floodplain habitat), habitat fragmentation caused by impoundments, and historical harvest/collection, which has been prohibited since 1992.

In Kansas, there have been no recent (since 2000) confirmed records of alligator snapping turtles. Two Kansas counties in this analysis unit have confirmed historical records, the most recent of which are from 1912 (Butler County) and 1958 (Cowley County). Alligator snapping turtles are not known to still occur in these counties, but the reason for their apparent disappearance is not known; most monitoring in Kansas occurred after significant perturbations already took place (e.g., historical harvest, fragmentation from impoundments), and most of the species' historical range in the state occurs on private lands with limited accessibility for surveying.

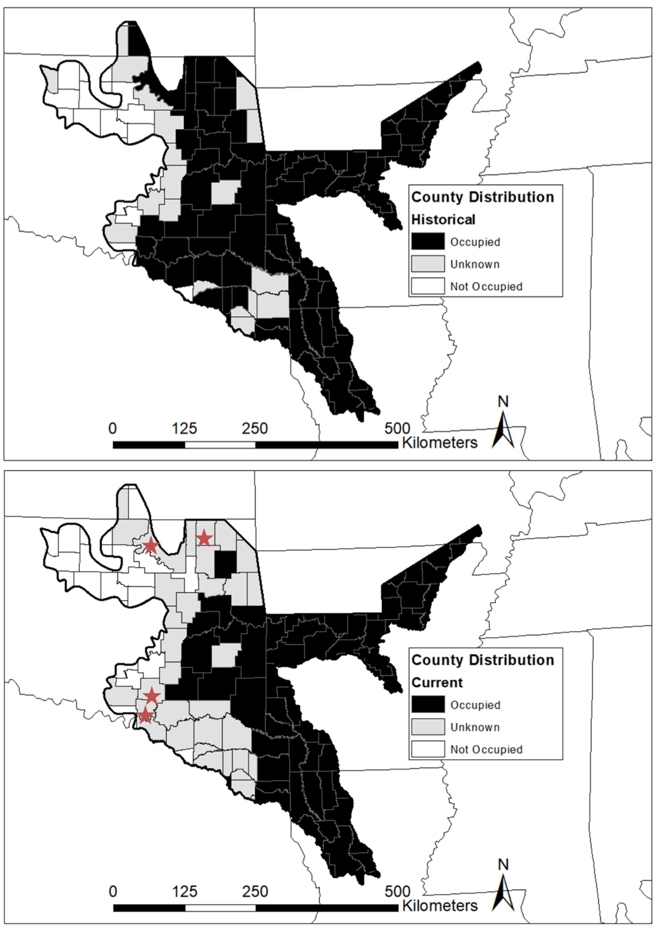


Figure 20. Historical and current distribution of alligator snapping turtles in the Southern Mississippi – West Analysis Unit. Counties in Oklahoma with ongoing reintroductions are indicated with stars.

4.6.3 Southern Mississippi – East Analysis Unit

Figure 21. Southern Mississippi – East Analysis Unit.

This analysis unit (Figure 21) encompasses parts of Louisiana, Arkansas, Mississippi, Alabama, Tennessee, and Missouri. Main water bodies that currently or historically supported alligator snapping turtles include the Mississippi River, Atchafalaya River, Red River, Ouachita River, Tensas River, Amite River, Tangipahoa River, and their affluents in Louisiana. Historically extensive bottomland hardwood forests associated with the alluvial plains of these rivers still provide extensive aquatic habitat for alligator snapping turtles in the form of bayous, sloughs, brakes (swamps), and oxbow lakes; stream modifications within Louisiana for drainage, irrigation, navigation, and recreational purposes have been extensive. The net effect of many impoundment projects has probably been to create more suitable, permanent aquatic habitat than was historically present; however, the transformation of the adjoining terrestrial environment includes significantly more edge habitat that is suitable for mesopredators such as raccoons. Also, it is common for roadways and railways to cross or border bodies of water, and in addition, many bodies of water are intersected by pipelines and other utility rights-of-way—these types of anthropogenic modifications near water create attractive edges that are used for nesting by *Macrochelys* (Carr et al., 2007).

Protected areas with confirmed presence of the species within the Louisiana portion of the unit include Kisatchie National Forest, numerous National Wildlife Refuges (e.g., Black Bayou Lake NWR, Upper Ouachita NWR, Tensas River NWR) and state Wildlife Management Areas (e.g., Russell Sage WMA, Boeuf WMA, Richard K. Yancey WMA, Loggy Bayou WMA) and within the Mississippi portion of the Unit, Big Black River.

Current Abundance

Current abundance in this analysis unit is estimated to be 50,000 alligator snapping turtles, with 80% confidence that the true abundance is between 2,000 and 75,000. These estimates were generated by extrapolating trapping information in the southern third of the unit to the rest of the unit.

Current Threats and Conservation Actions

Threats in this analysis unit include:

- **Incidental hooking**, which is estimated to affect 45% of the species' range in this unit (three experts responding, average bounds between 28 and 67%, average 73% expert confidence that the true value lies within their specified bounds).
- Bycatch from **commercial fishing**, which is estimated to affect 48% of the species' range in this unit (two experts responding, average bounds between 33 and 66%, average 83% expert confidence that the true value lies within their specified bounds).

- **Legal harvest**, which is estimated to affect 53% of the species' range in this unit (two experts responding, average bounds between 38 and 68%, average 90% expert confidence that the true value lies within their specified bounds). Harvest is legal in Louisiana and Mississippi.
- **Illegal harvest**, which is estimated to affect 63% of the species' range in this unit (three experts responding, average bounds between 43 and 90%, average 60% expert confidence that the true value lies within their specified bounds).
- **Nest predation**, which is estimated to affect 94% of the species' range in this unit (three experts responding, average bounds between 58 and 99%, average 93% expert confidence that the true value lies within their specified bounds).
- Habitat **fragmentation** was also identified as a threat to populations in this unit.

Outside of Louisiana and Mississippi, alligator snapping turtles in this unit are **protected** at the state level with no legal harvest. In Mississippi, harvest is limited to one alligator snapping turtle per person (with a hunting or fishing license) per year with a carapace length greater than 24 inches (female-biased protection), and with no possession allowed between April and June. Other conservation measures include a **head-start and release program** in Louisiana to supplement existing populations.

<u>Comparison with Historical Range</u>

In this analysis unit, there have been no confirmed changes in the species' range in Louisiana or Arkansas (Figure 22).

In Mississippi, there have been changes from occupied to unknown status and vice versa, but no changes of the species' range have been confirmed; the lack of recent records is likely more of an indication of a lack of recent surveys than a lack of alligator snapping turtles. It is assumed by Mississippi Natural Heritage Program personnel that historically occupied counties are still occupied, and that currently occupied counties were historically occupied. Alligator snapping turtles are presumed to occur state-wide, but there has not been adequate survey effort to confirm the presence or absence of the species in all counties, resulting in the large number of counties with both historical and current unknown status.

In Alabama, all counties included in this analysis unit are presumed to have been historically occupied, and most have changed to unknown status currently because of a lack of recent surveys; there is not current evidence that the species has been extirpated in these counties. The most recent confirmed records in counties within this unit were from 1980 in Lauderdale County.

In Tennessee, there has been an apparent contraction of the range of the species in this analysis unit. All counties in this unit are presumed to have been historically occupied, but there are recent records only for 7 out of 19 counties. The contraction is believed to be a result of habitat destruction caused by the channelization of most of the river systems in west Tennessee. There is also likely an element of limited survey effort constraining the current range; new locations are expected to be documented over the next several years with more surveys.

This analysis unit includes parts of 7 counties in Missouri. Of these, all but one are known to have been historically occupied, and it is likely that the remaining one, Scott County, was also historically occupied based on its proximity to other occupied counties and watersheds. The only change between the historical and current state is in New Madrid County, where the most recent record comes from 1993. The lack of recent records could be due solely to a lack of recent surveys, so its current status is unknown.

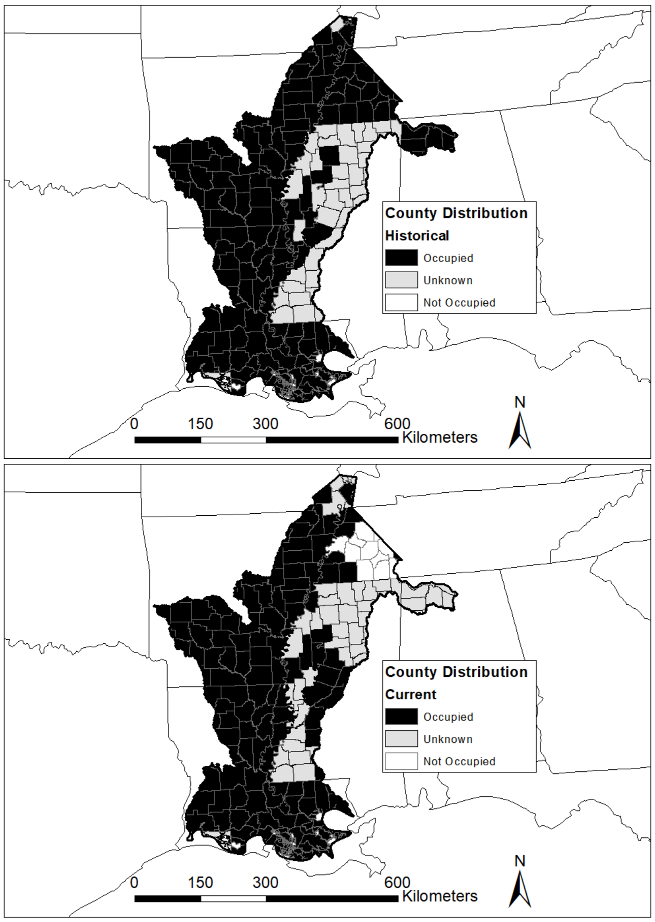


Figure 22. Historical and current distribution of alligator snapping turtles in the Southern Mississippi – East Analysis Unit.

4.6.4 Northern Mississippi – West Analysis Unit

This analysis unit (Figure 23) encompasses parts of Kansas, Oklahoma, Arkansas, and Missouri. Main water bodies that currently or historically supported alligator snapping turtles include the Neosho River and Verdigris River.

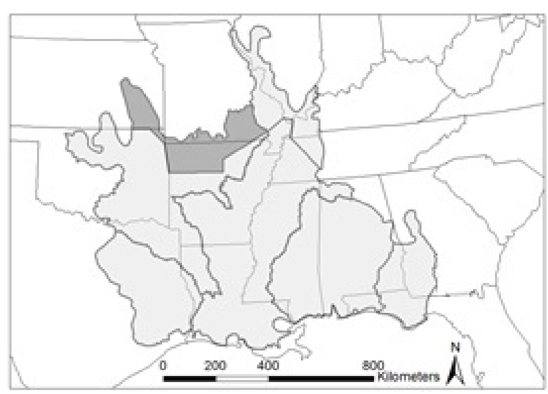


Figure 23. Northern Mississippi – West Analysis Unit.

<u>Current Abundance</u>

Current abundance in this analysis unit is estimated to be 500 alligator snapping turtles, with 60% confidence that the true abundance is between 10 and 1,000. These estimates were based on experience in the Neosho and Verdigris River in the northwest corner of this unit.

Current Threats and Conservation Actions

Threats in this analysis unit include:

- **Incidental hooking**, which is estimated to affect 80% of the species' range in this unit (80% confidence that the true value lies between 60 and 100%).
- **Illegal harvest**, which is estimated to affect 1% of the species' range in this unit (100% confidence that the true value lies between 0 and 10%).
- **Nest predation**, which is estimated to affect 30% of the species' range in this unit (50% confidence that the true value lies between 10 and 80%).
- **Habitat fragmentation** was also identified as a threat to populations.

Alligator snapping turtles in this unit are **protected** at the state level with no legal harvest. Other conservation measures include **head-start and release programs** on the Caney, Verdigris, and Neosho river drainages in Oklahoma. The spatial extent and movements of alligator snapping turtles are constrained by dams there, but releases up and downstream of impoundments are expected to increase their spatial extent over time.

Comparison with Historical Range

In this analysis unit, there have been no confirmed changes in the species distribution between the occupied and unoccupied state; the only changes between historical and current times are changes between occupied and unknown, and changes between unknown and unoccupied (Figure 24).

In Kansas, there have been no recent (since 2000) confirmed records of alligator snapping turtles.

Five Kansas counties in this analysis unit have confirmed historical records; the most recent record for each of these are: 1895 in Cherokee County, 1911 in Neosho County, 1938 in Labette County, 1967 in Lyon County, and 1991 in Montgomery County. Alligator snapping turtles are not known to occur in these counties, and the reason for their apparent disappearance is not known; most monitoring in Kansas occurred after significant perturbations already took place (e.g., historical harvest, fragmentation from impoundments), and most of the species' historical range in the state occurs on private lands with limited accessibility for surveying.

In the small portion of this unit that occurs in Oklahoma, there have been no confirmed changes in the species' range.

In the Arkansas portion of this analysis unit, there have not been confirmed changes in the species' range, but there is a lack of historical or recent records in the northwestern portion of the state, leading to a current designation of unknown status, though these counties are presumed to have been historically occupied based on availability of potential habitat and proximity to other occupied areas.

In the Missouri portion of this unit, there have not been confirmed changes in the species' range; counties with unknown current status that historically supported alligator snapping turtles likely still do, but there have not been recent surveys to confirm this.

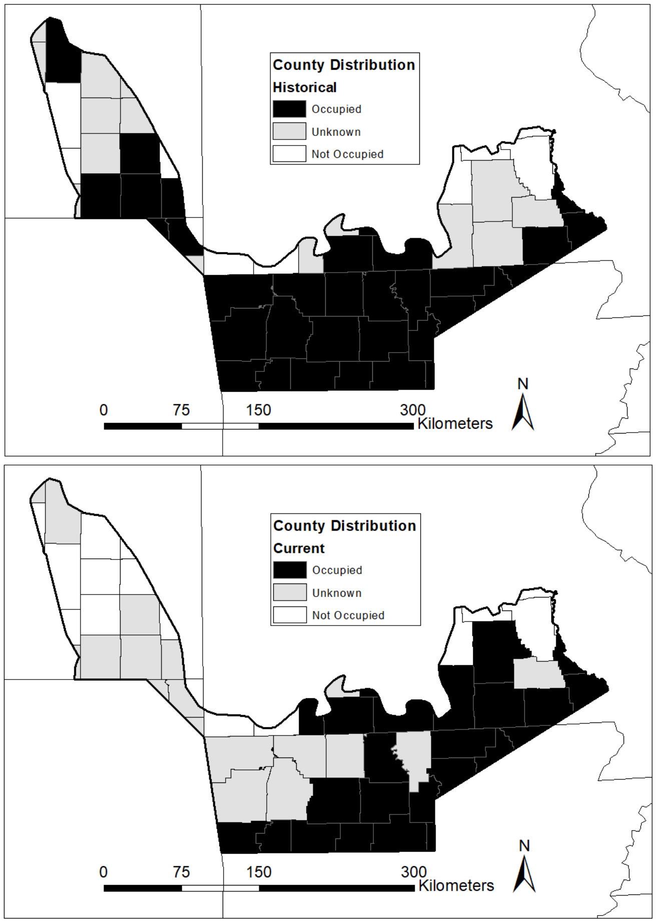


Figure 24. Historical and current distribution of alligator snapping turtles in the Northern Mississippi – West Analysis Unit.

4.6.5 Northern Mississippi – East Analysis Unit

This analysis unit (Figure 25) encompasses parts of Missouri, Illinois, Indiana, Kentucky, and Tennessee. Main water bodies that currently or historically supported alligator snapping turtles include the Mississippi River, Ohio River, Illinois River and Tennessee River.

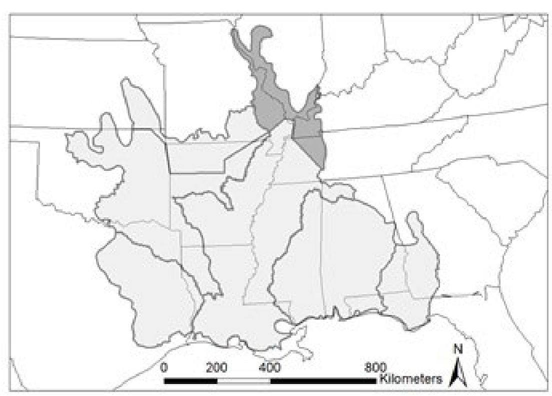


Figure 25. Northern Mississippi – East Analysis Unit

<u>Current Abundance</u>

Current abundance in this analysis unit was estimated by two expert respondents. One estimated the abundance to be 125 alligator snapping turtles, with 90% confidence that the true abundance is between 75 and 150. The other estimated the abundance to be 300 alligator snapping turtles, with 75% confidence that the true value is between 150 and 1,500. These estimates were based on experience associated with recovery efforts (translocations and monitoring) in the unit. Combined, these estimates produce an average estimate of 212.5 alligator snapping turtles, average lower bound of 112.5, and average upper bound of 825, and an average 82.5% confidence from the experts that the true value is between the bounds (i.e., the 82.5% confidence level does not apply to the average bounds, but describes on average how confident the experts were for this analysis unit).

Current Threats and Conservation Actions

Threats in this analysis unit include:

- **Nest predation**, which is estimated to affect 83% of the species' range in this unit (three experts responding, average bounds between 53 and 100%, average 88% expert confidence that the true value lies within their specified bounds).
- **Habitat alteration** from channelization, impoundments, sedimentation, and woody debris removal was also identified as a threat to populations in this unit.
- **Incidental hooking** and **illegal harvest** are not believed to be threats in this analysis unit. They were estimated to affect 0% of the species' range in this unit (two experts responding, average bounds between 0 and 2.5%, average 85% expert confidence that the true value lies within their specified bounds for incidental hooking, and 97% confidence for illegal harvest).

Alligator snapping turtles in this unit are **protected** at the state level with no legal harvest. Other conservation measures include reintroductions and associated monitoring in Illinois.

Comparison with Historical Range

In this analysis unit, the species' range has contracted in some areas (Figure 26). In Missouri, the species is no longer believed to occur in Lewis County in the northeastern portion of the state, where the last alligator snapping turtle record is from 1965. In other Missouri counties that were historically occupied, the species likely still occurs there, regardless of whether there have been recent surveys and records.

In Illinois, reintroductions are currently happening in Union County near the southern tip of the state. Excluding reintroductions, the most recent capture of an alligator snapping turtle in the state was 2017 in Union County (Kessler et al 2017, entire). Prior to this capture, the last verified record was in Union County in 1984. An additional 12 Illinois counties have confirmed historical records, the most recent record for each of these are: 1887 in Wabash County, 1892 in White and Adams Counties, 1907 in Alexander County, 1937 in Randolph and Massach Counties, 1950 in Rock Island County, 1954 in Calhoun County, 1960 in Jackson County, 1961 in Mason and Jersey Counties, and 1976 in Peoria County.

In Indiana, alligator snapping turtles are exceedingly rare. In 2012, an isolated specimen was caught on a limb line in Jackson County. Prior to that, no alligator snapping turtle records had been verified since 1991 in Morgan County . The current range in Indiana might be wider than previously thought (Figure 27); environmental DNA indicating alligator snapping turtle presence was detected in 2017 in Gibson and Pike counties, but has not been confirmed with captures. It is not certain how far away the turtles might be from where their DNA was detected.

In Kentucky, there have not been confirmed changes in the species' range; systematic surveys are not occurring in Kentucky and all occurrence records are opportunistic.

In Tennessee, there has been an apparent contraction of the range of the species in this analysis unit. All counties in this unit except Weakley County are presumed to have been historically occupied, but there are recent records only for 7 out of 13 historically occupied counties. The contraction is believed to be a result of habitat destruction caused by the channelization of most of the river systems in west Tennessee. There is also likely an element of limited survey effort constraining the current range; new locations are expected to be documented over the next several years with more surveys.

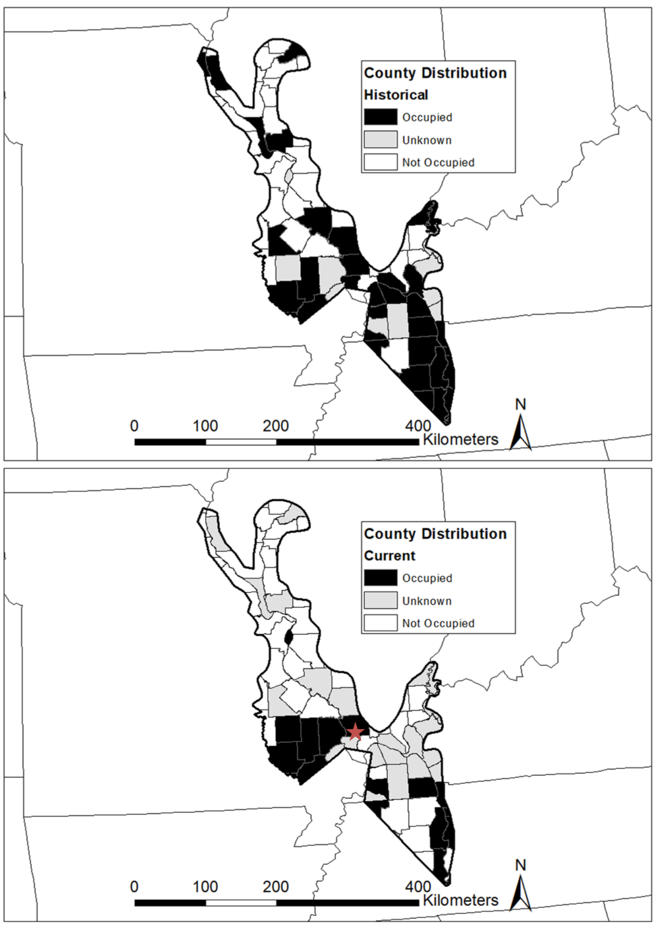


Figure 26. Historical and current distribution of alligator snapping turtles in the Northern Mississippi – East Analysis Unit. Union County in Illinois with ongoing reintroductions is indicated with a star.

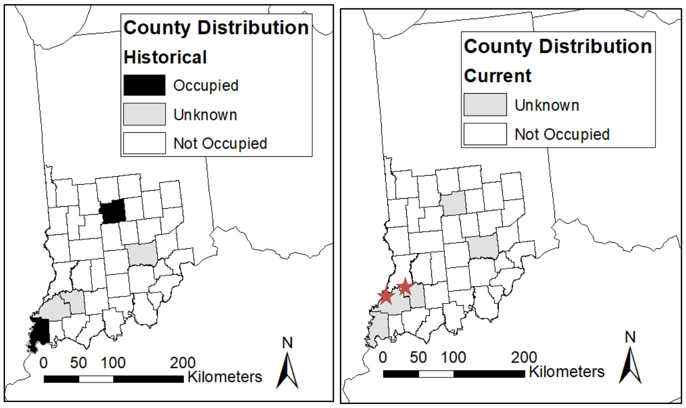


Figure 27. Historical and current distribution of alligator snapping turtles in Indiana. Stars indicate counties where environmental DNA from alligator snapping turtles was detected in 2017.

4.6.6 Alabama Analysis Unit

This analysis unit (Figure 28) encompasses eastern Mississippi, western Alabama, and small parts of Louisiana and Florida. Main water bodies that currently or historically supported alligator snapping turtles include but are not limited to the Alabama River, Pascagoula River, Pearl River, Jourdan River, Escambia River and Perdido River.

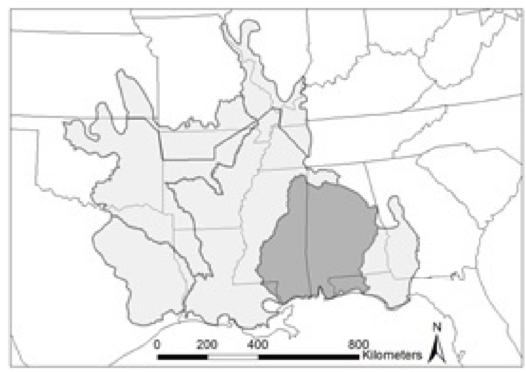


Figure 28. Alabama Analysis Unit.

<u>Current Abundance</u>

Current abundance in this analysis unit is estimated to be 200,000 alligator snapping turtles, with 66% confidence that the true abundance is between 50,000 and 1,000,000. These estimates were based on extrapolating localized experience to the larger unit.

<u>Current Threats and Conservation Actions</u>

Threats in this analysis unit include:

- **Incidental hooking**, which is estimated to affect 52% of the species' range in this unit (three experts responding, average bounds between 55 and 90% [average value does not fall between bounds because one expert provided only a mostly likely estimate with no bounds], average 65% expert confidence that the true value lies within their specified bounds).
- **Legal harvest**, which is estimated to affect 40% of the species' range in this unit (two experts responding, average bounds between 34 and 55%, average 70% expert confidence that the true value lies within their specified bounds). Harvest is legal in Mississippi.
- **Illegal harvest**, which is estimated to affect 58% of the species' range in this unit (three experts responding, average bounds between 68 and 95% [average value does not fall between bounds because one expert provided only a most likely estimate with no bounds], average 58% expert confidence that the true value lies within their specified bounds).

- **Nest predation**, which is estimated to affect 83% of the species' range in this unit (three experts responding, average bounds between 53 and 100%, average 88% expert confidence that the true value lies within their specified bounds).
- **Habitat alteration** from channelization, impoundments, headcutting, desnagging, dredging, unregulated water use, and water contamination was also identified as a threat to populations in this unit.

Outside of Mississippi, alligator snapping turtles in this unit are **protected** at the state level with no legal harvest. In Mississippi, harvest is limited to one alligator snapping turtle per person (with a hunting or fishing license) per year with a carapace length greater than 24 inches (female-biased protection), and with no possession allowed between April and June.

<u>Comparison with Historical Range</u>

In this analysis unit, there are no confirmed changes in the species' range (Figure 29). While there are not historical or recent occurrence records from every county, there is no evidence that the species has been extirpated from these areas, and the lack of records could be from lack of surveys.

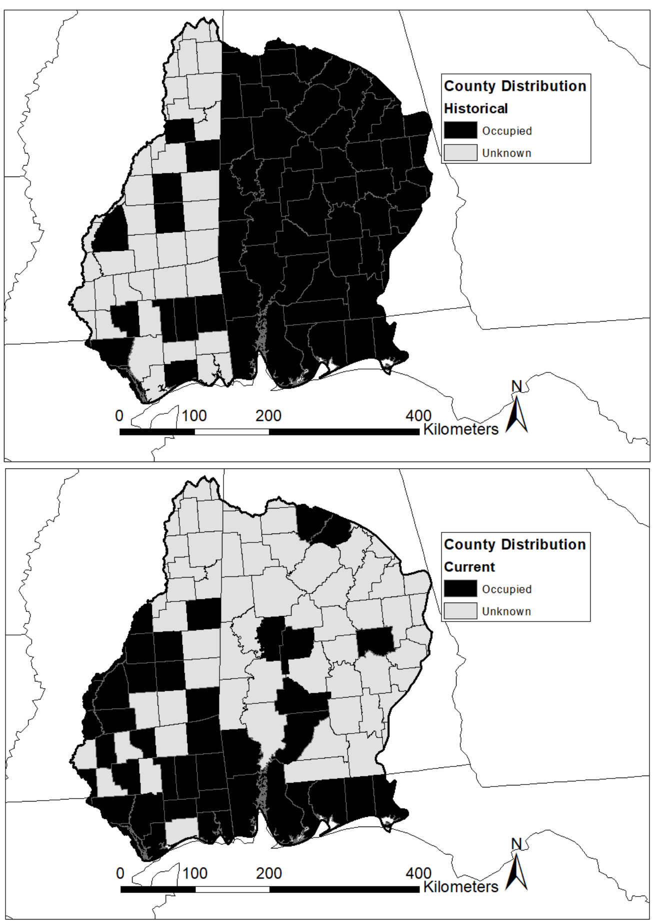


Figure 29. Historical and current distribution of alligator snapping turtles in the Alabama Analysis Unit.

4.6.7 Apalachicola Analysis Unit

This analysis unit (Figure 30) encompasses parts of the Florida panhandle, southeastern Alabama, and Georgia. Main water bodies that currently or historically supported alligator snapping turtles include the Apalachicola River, Chipola River, Ochlockonee River, Flint River, Chattahoochee River, Choctawhatchee River, and associated permanent freshwater habitats. The latter include floodplain swamp forest dominated by bald cypress and water tupelo trees, with tannic or turbid waters (Ewert and Jackson 1994). Lakes supporting the species are either impounded sections of large rivers (Lake Seminole: Apalachicola, Lake Talquin: Ochlockonee) or natural lakes with at least occasional connection to a river.

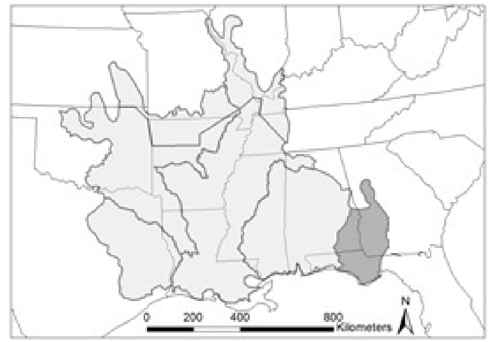


Figure 30. Apalachicola Analysis Unit.

<u>Current Abundance</u>

Current abundance in this analysis unit was estimated by two expert respondents: one estimated the abundance to be 10,000 alligator snapping turtles, with 50% confidence that the true abundance is between 5,000 and 20,000. The other estimated the abundance to be 80,000 alligator snapping turtles, with 70% confidence that the true abundance is between 25,000 and 400,000. These estimates were based on extrapolating localized experience to the larger unit. Combined, these estimates produce an average estimate of 45,000 alligator snapping turtles, average lower bound of 15,000, and average upper bound of 210,000, and an average 60% confidence from the experts that the true value is between the bounds of their individual estimates (i.e., the 60% confidence level does not apply to the average bounds, but describes on average how confident the experts were for this analysis unit).

<u>Current Threats and Conservation Actions</u>

Threats in this analysis unit include:

- **Incidental hooking**, which is estimated to affect 45% of the species' range in this unit (two experts responding, average bounds between 20 and 80%, average 70% expert confidence that the true value lies within their specified bounds).
- **Illegal harvest**, which is estimated to affect 38% of the species' range in this unit (two experts responding, average bounds between 28 and 60%, average 63% expert confidence that the true value lies within their specified bounds).
- **Nest predation**, which is estimated to affect 61% of the species' range in this unit (three experts responding, average bounds between 55 and 70%, average 61% expert confidence that the true value lies within their specified bounds).
- **Habitat alteration** from siltation, desnagging, dredging, impoundments, and unregulated water use, and alteration of nesting habitat was also identified as a threat to populations in this unit.

Throughout this entire analysis unit, alligator snapping turtles are **protected** at the state level with no legal harvest.

Comparison with Historical Range

In this analysis unit, there are no confirmed changes in the species' range (Figure 31). While there are not historical or recent occurrence records from every county, there is no evidence that the species has been extirpated from these areas, and the lack of records could be from lack of surveys.

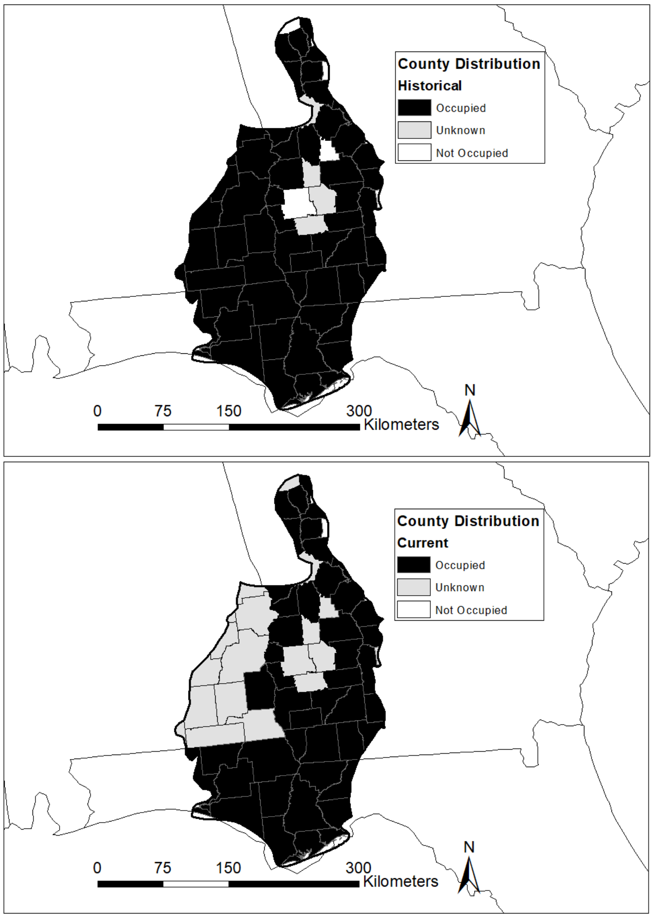


Figure 31. Historical and current distribution of alligator snapping turtles in the Apalachicola Analysis Unit.

4.7 Current Condition Overall Results

In this section, we summarize the above results to describe the current resilience, redundancy, and representation for alligator snapping turtles.

4.7.1 Current Resilience

As noted before, abundance is our measure for current resilience, with information about current threats and distribution serving as auxiliary information.

Estimates of abundance across analysis units range from a high of 200,000 alligator snapping turtles in the Alabama Unit to a low of 212.5 turtles in the Northern Mississippi – East Unit (Figure 32). Both the Northern Mississippi – East and Northern Mississippi – West Units, at the northern reaches of the species' range, have estimated abundances orders of magnitude smaller than most of the more southerly units. These northern units have also experienced more range contraction and local extirpation than more southern units.

Range-wide the abundance of alligator snapping turtles is estimated to be between 68,154 and 1,436,825 (a range of 1,368,671). This enormous range in the estimated abundance illustrates the very high degree of uncertainty that exists in abundances at local sites and the ability to extrapolate local abundance estimates to a much broader spatial scale. Within these bounds, the most likely estimate of range-wide alligator snapping turtle abundance is 361,213 turtles, with 55% of these occurring in the Alabama Analysis Unit.

Just as there are scarce data to estimate current abundances, there is little information with which to make rigorous comparisons between current and historical abundances. Dramatic population depletions occurred in Louisiana, Alabama, Georgia, the Florida panhandle, and elsewhere in the range during the 1960s and 1970s, but information about the magnitude of the changes come from anecdotal observations by trappers (Pritchard 1989, p. 74, 76, 80, 83). Since that time, harvest has been banned in a large portion of the species' range (all states except Louisiana and Mississippi). There are limited data available describing how populations have responded to reduced harvest pressure. Population dynamics in Georgia, Arkansas, and Oklahoma were modeled using relatively recent survival rates (i.e., from mark-recapture studies conducted during the late 1990s-2010s; Folt et al. 2016, p. 28). Results from these models suggest that the population in Spring Creek, Georgia, has been growing, but those in East Fork Cadron Creek, Arkansas (data from Howey et al. 2013), and Big Vian Creek, Oklahoma (data from East et al. 2013) are still in decline. Twenty-two years after commercial harvest ended, surveys conducted during 2014 and 2015 in Georgia's Flint River revealed no significant change in abundance since 1989 surveys (King et al. 2016, p. 583). A similar study in Missouri and Arkansas detected population declines between the initial survey period in 1993-1994 and repeat surveys in 2009 over a decade after state-level protections were implemented (Lescher et al. 2013, p. 163-164). However, an additional study in Arkansas spanning 20 years, documented an increase in abundance of both adult male and female alligator snapping turtles within Salado Creek (Trauth et al. 2016, p. 242). At Sequoyah National Wildlife Refuge in Oklahoma, an alligator snapping turtle population declined between 1997-2001 and 2010-2011 (Ligon et al. 2012, p. 40).

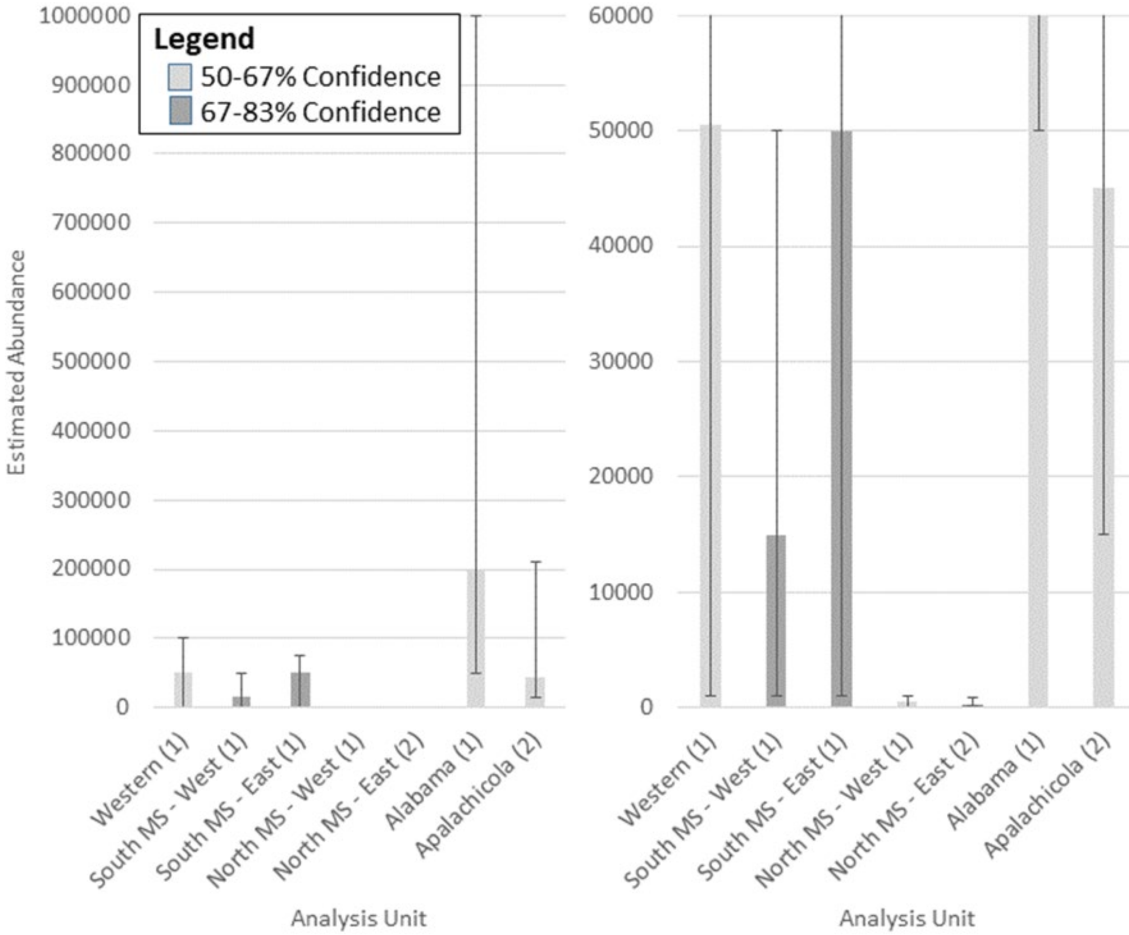
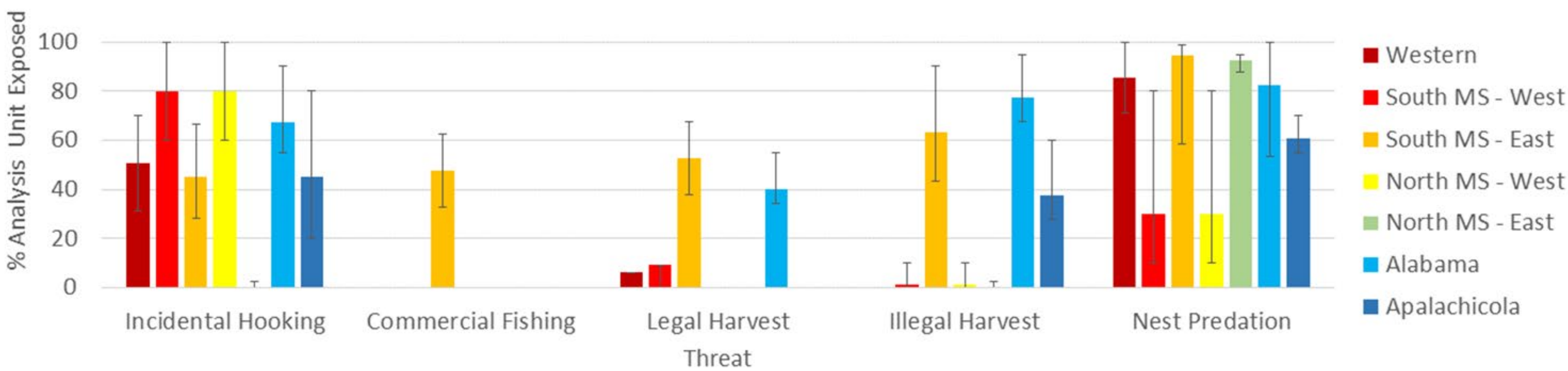


Figure 32. Estimated abundance of alligator snapping turtles in each analysis unit. Y-axis zoomed in on right. Darker bars show higher confidence of species experts in their estimates, and the number of experts that provided estimates for each unit is indicated in parentheses. Though the bars cannot easily be seen in the zoomed in graph on the right, there was 67-83% expert confidence in abundance estimates in the North MS - East Unit and 50-67% expert confidence in abundance estimates in the North MS - West and Suwannee Units.



% Analysis Unit Exposed
100
80
60
40
20
0
Incidental Hooking
Commercial Fishing
Legal Harvest
Illegal Harvest
Nest Predation
Threat
Western
South MS - West
South MS - East
North MS - West
North MS - East
Alabama
Apalachicola

Figure 33 displays the spatial extent of different threats across the analysis units. Alligator snapping turtles range wide are believed to be exposed to the threat of incidental hooking on recreational trot and limb lines, with estimates of the percentage of turtles exposed to the threat ranging from 45% to 80%, with the exception of the North Mississippi – East Analysis Unit, where incidental hooking is not a significant threat. We received very little information about the extent of the threat of commercial fishing bycatch, suggesting either that this is not believed to be a significant threat, or that there is too much uncertainty in the extent of the threat for the experts to provide useful estimates. Legal harvest is limited to Louisiana and Mississippi, so this threat, despite its large potential impact on demography, is spatially limited to the analysis units in which those two states occur. There is wide variation in the estimated prevalence of illegal harvest across the species' range, with the highest estimates in the analysis units where legal harvest is also present. Estimates of the extent of nest predation vary and are estimated to be lowest in the Southern Mississippi – West and Northern Mississippi – West Units (both 30%), with the highest extents in the remaining five analysis units (61-94%).

In Table 3, we have listed the analysis units in descending order of resilience, where resilience is measured by the estimated current abundance. Because analysis units do not correspond with biological populations, we do not make any statements about what abundance might constitute a "viable" or "highly resilient" population size; the actual grouping of these estimated turtles into populations in unknown. Also, the analysis units chosen for this assessment vary in size and are not directly related to biological populations, and abundance within a unit is influenced by the size of the unit. In order to control for the size of units, we also calculated a density of alligator snapping turtles, reported in Table 3 as the number of turtles per 1,000 hectares of open water in the unit (as delineated by the 2016 National Land Cover Database; Yang et al. 2018, entire). Note that these are rough densities meant only to correct abundances for analysis unit size so that units can be more appropriately compared relative to each other; they are not intended to serve as actual estimates of density in alligator snapping turtle habitat. Because of the variation in analysis unit size and limitations in calculating true densities of alligator snapping turtles within units, we refrain from leaning heavily on comparisons of abundance or density between analysis units to summarize resilience other than to highlight general patterns. Resilience increases with abundance and density; where there are more individuals, populations will have a greater ability to withstand stochastic demographic and environmental events. Thus, resilience is highest in the core of the species' range, lowest in the northern-most analysis units at the edge of the range.

While we caution against leaning too heavily on comparisons of current abundance or density between populations because of high uncertainty contained in the information that generated the estimates, these values are the best information currently available and will serve as useful baseline conditions against which to compare future resilience in the next chapter of this SSA.

Table 3. Analysis units listed in descending order of estimated abundance (most likely estimate from expert elicitations) and densities expressed as estimated abundance per 1,000 hectares of open water in each unit. Analysis units are highlighted where over 50% of alligator snapping turtles are exposed to harvest or over 50% of nests are exposed to nest predation by subsidized or non-native predators. Where the range of the species is contracting, the states experiencing the losses are noted.

Analysis Unit	Estimated Abundance	Abundance/ 1,000 hectares Open Water	Substantial Threats*	Range Contraction
Alabama	200,000	616.9	1) Adult harvest (Legal & Illegal) 2) Nest Predation 3) Incidental Hooking/Hook Ingestion	
Western	50,500	139.3	1) Nest Predation	
South MS - East	50,000	55.3	1) Adult harvest (Legal & Illegal) 2) Nest Predation	TN
Apalachicola	45,000	281.3	1) Nest Predation	
South MS - West	15,000	30.2	1) Incidental Hooking/ Hook Ingestion	KS, possibly OK
North MS - West	500	4.7	1) Incidental Hooking/ Hook Ingestion	KS
North MS - East	212.5	1.0	1) Nest Predation	IL, TN, KY, MO

*"Substantial" threats here refer to those threats estimated to reduce survival rates of an age class by 8% or more (see Figure 16 in Section 4.5.2): legal and illegal harvest reduce adult survival and nest predation reduces nest survival. To be listed for any given analysis unit, the substantial threat must be estimated to be impacting >50% of the alligator snapping turtles in the unit.

4.7.2 Current Representation

Representation refers to the breadth of diversity within and among populations of a species, which allow it to adapt to changing environmental conditions. Because of how we delineated analysis units (rather than biological populations that we could not delineate), there are only one or two analysis units in each representative unit. Because of this mismatch in scale between analysis units and biological populations, we present representation here both in terms of analysis units and abundance (Table 4), under the assumption that representative units with higher abundances will be more able to contribute to future adaptation than those with lower abundances.

No representative units have been lost compared to the historical distribution. The Northern Mississippi Representative Unit, which adds diversity in life history strategies within the species, currently has very low abundance within its two constituent analysis units relative to the other representative units, with an estimated 712.5 alligator snapping turtles total and a shrinking range. However, alligator snapping turtles in Illinois have been introduced from Southern Mississippi breeding stock, diluting the presence of unique genetic characteristics in the Northern Mississippi Representative Unit. The representative units within the core of the species' range, which also contain only one or two analysis units each, are estimated to support at least 45,000 alligator snapping turtles.

Table 4. Representative units listed in descending order of estimated abundance. Where the range of the species is contracting, the states that have experienced losses are noted.

Representative Unit	Number Analysis Units	Estimated Abundance (Most Likely)	Range Contraction
Alabama	1	200,000	
Southern MS	2	65,000	TN, KS, possibly OK
Western	1	49,500	
Apalachicola	1	45,000	
Northern MS	2	712.5	KS, IL, TN, KY, MO

4.7.3 Current Redundancy

Redundancy refers to the number and distribution of resilient populations across a species' range, which provides protection for the species against catastrophic events that impact entire populations. We delineated seven analysis units across the species' range (Figure 14), and none have been lost compared to the historical distribution. As described above, each representative unit contains one or two analysis units (Table 4).

Though the number of analysis units has not changed, redundancy for alligator snapping turtles has been reduced in terms of the distribution within analysis units, with range contractions in the northern portions of the species' range (Oklahoma, Kansas, Missouri, Illinois, Kentucky, and Tennessee; Figure 34). Within the core of their range however, alligator snapping turtles still seem to be widely distributed, though there are many gaps in the spatial extent of surveys. While the distribution of the species still encompasses much of its historical range, resilience within that range has decreased, largely from historical harvest pressures. With the range contractions and decreases in abundance, the Northern Mississippi – East Analysis Unit has decreased in resilience such that it is not a robust contributor to redundancy (only 212.5 estimated abundance, influenced largely by introductions).

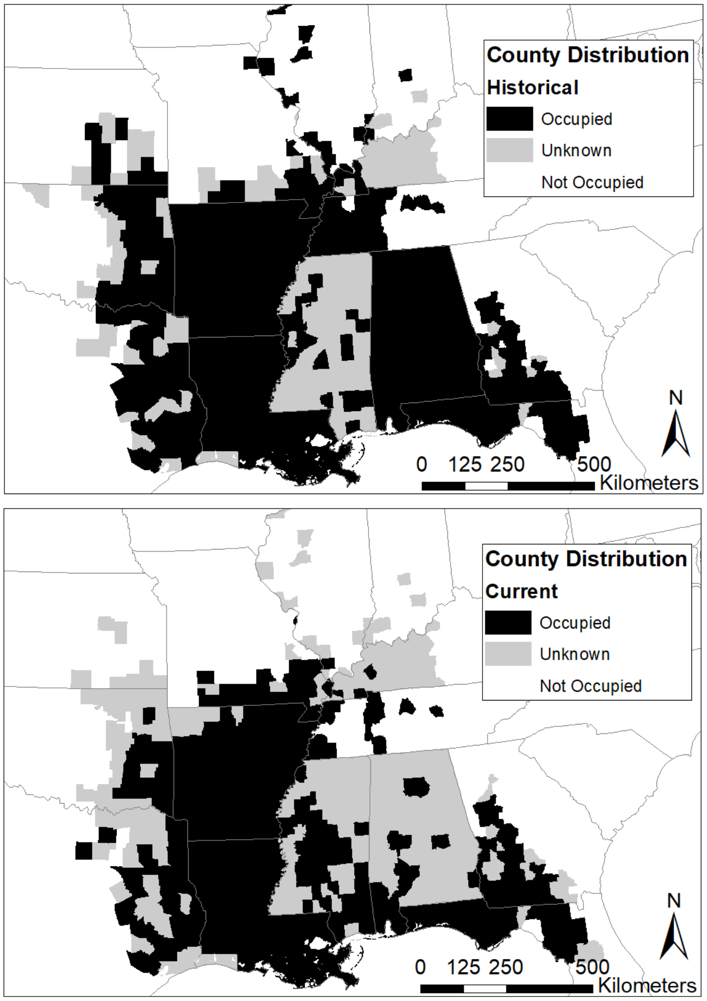


Figure 34. Historical and current alligator snapping turtle and Suwannee snapping turtle distribution by county or parish. Unknown status can be caused by a lack of recent surveys to detect turtles where they likely still exist (especially in core of the species' range), or can represent counties or parishes where the species has been searched for and not detected, and may be absent, but there is still a chance that the species persists there undetected.

A table with the information used to generate Figure 34, the current and historical status of each county within the alligator snapping turtle and Suwannee snapping turtle range, can be found in Appendix D. To summarize, of 422 counties that were historically occupied by alligator snapping turtles, 278 are still occupied, 124 have unknown status, and 20 are not occupied. Of 155 counties with an unknown historical status, 39 are currently occupied, 107 have unknown status, and 9 are not occupied. Seven counties that were not historically occupied currently have unknown status.

This concludes the assessment of the current condition of alligator snapping turtles across their range. In the next section, we continue to use the expert-elicited information about the extent and magnitude of threats to the species to forecast their condition into the future.

4.8 Summary of Population and Species Needs and Current Condition

In order to determine the representation across the range of the species, we used a tiered approach (first using genetics and then life history and ecology) and delineated five representative units: Western, Southern Mississippi, Northern Mississippi, Alabama, and Apalachicola. Subdivision of representative units into analysis units was based primarily on Hydrologic Unit Code (HUC) 2 watershed boundaries. In creating analysis units, we strived to balance the needs to a) have units small enough to be able to capture the variation in the condition of the species (e.g., abundance, threats) across its range, while also b) retaining units large enough that species experts would be able to summarize information about the condition of the species for every unit.

Current Resilience

To assess the current condition of alligator snapping turtles, information was gathered from species experts about current abundance (our measure of resilience), current threats, and a comparison of the current and historical distribution. Estimates of abundance across analysis units range from a high of 200,000 alligator snapping turtles in the Alabama Unit to a low of 212.5 turtles in the Northern Mississippi – East Unit. Both the Northern Mississippi – East and Northern Mississippi – West Units, at the northern reaches of the species' range, have estimated abundances that are orders of magnitude smaller than most of the more southerly units. These northern units have also experienced more range contraction and local extirpation than more southern units. Among the southern units, the Suwannee Analysis Unit on the far eastern portion of the species' range has the lowest abundance.

Range wide, the abundance of alligator snapping turtles is estimated to be between 68,154 and 1,435,825 alligator snapping turtles (a range of 1,368,671). This enormous range in the estimated abundance illustrates the very high degree of uncertainty that exists in abundances at local sites and the ability to extrapolate local abundance estimates to a much broader spatial scale. Within these bounds, the most likely estimate of range-wide alligator snapping turtle abundance is 361,213 turtles, with 55% of these occurring in the Alabama Analysis Unit.

Alligator snapping turtles range-wide are believed to be exposed to the threat of incidental hooking on recreational trot and limb lines, with estimates of the percentage of turtles exposed to the threat ranging from 45% to 80% except for the North Mississippi – East Analysis Unit, where incidental hooking is not a significant threat. We received very little information about the extent of the threat of commercial fishing bycatch, suggesting either that this is not believed to be a significant threat or too much uncertainty exists in the extent of the threat for the experts to provide useful estimates. Legal harvest is limited to Louisiana and Mississippi, so this threat, despite its large potential impact on demography, is spatially limited to the analysis units in which those two states occur. There is wide variation in the estimated prevalence of illegal harvest across the species' range, with the highest estimates in the analysis units where legal harvest is also present. Estimates of the extent of nest predation vary and are estimated to be lowest in the Southern Mississippi – West and Northern Mississippi – West Units (both 30%), with the highest extents in the remaining five analysis units (61-94%).

Because of the variation in analysis unit size and limitations in calculating true densities of alligator snapping turtles within units, we refrained from leaning heavily on comparisons of abundance or density between analysis units to summarize resilience other than to highlight general patterns. Resilience increases with abundance and density; where there are more individuals, populations will have a greater ability to withstand stochastic demographic and environmental events. Thus, resilience is highest in the core of the species' range and lowest in the northern-most analysis units at the edge of the range. While we caution against leaning too heavily on comparisons of current abundance or density between populations because of high uncertainty contained in the information that generated the estimates, these values will serve as useful baseline conditions against which to compare future resilience in the next chapter of this SSA.

Current Representation

No representative units, which each contain one or two analysis units, have been lost compared to the historical distribution. The Northern Mississippi Representative Unit, which adds diversity in life history strategies within the species, currently has very low abundance within its two constituent analysis units relative to the other representative units, with an estimated a total of 712.5 alligator snapping turtles and a shrinking range. This representative unit supports an estimated abundance of only 2,000 turtles. The representative units within the core of the species' range, which also contain only one or two analysis units each, are estimated to support at least 45,000 alligator snapping turtles.

Current Redundancy

The species has experienced range contractions in the northern portions of its range (Oklahoma, Kansas, Missouri, Illinois, Kentucky, and Tennessee). Within the core of the range, however, alligator snapping turtles still seem to be widely distributed, though there are many gaps in the spatial extent of surveys. While the distribution of the species still encompasses much of its historical range, resilience within that range has decreased, largely from historical harvest pressures. The Northern Mississippi – East Analysis Unit has decreased in resilience, but can only have limited contributions to redundancy given currently estimated abundance (only 212.5 estimated abundance, influenced largely by introductions). While range contractions have occurred within various states, the species presently occurs in all historically known states, except Indiana and Kansas, where its persistence is unconfirmed.

CHAPTER 5 – FUTURE CONDITIONS AND VIABILITY

In this chapter, we describe the methods used to project alligator snapping turtle populations into the future under different plausible scenarios, then summarize the results in terms of resilience, redundancy, and representation. Sections 5.1 and 5.2 contain a summary of the modeling methods and results – a more detailed technical report can be found in Appendix E.

5.1 Future Projection Model

We constructed a female-only, stage-structured matrix population model (Caswell 2001, p. 33) to project alligator snapping turtle population dynamics over annual time steps for 50 years in each analysis unit. We based our model on the peer reviewed and published model in Folt et al. (2016, p. 24) and updated the model to reflect the appropriate structure of matrix population models (Kendall et al. 2019, p. 33) and to better support the needs of the SSA. Our conceptual model of the alligator snapping turtle life cycle (Figure 35) upon which the model was based used a pre-breeding census structure with two life stages: juveniles included individuals ≥1 year-old that had not reached reproductive maturity, and adults included mature, breeding individuals. Because of the pre-breeding census structure, hatchlings were not included as a distinct life stage, but hatchling production and survival were incorporated into adult fecundity in the model. For each annual time step, individuals in the juvenile stage that survived the year could either remain a juvenile or transition to the adult stage. Individuals in the adult stage that survived the year could contribute to breeding. This quantitative model incorporated demographic rates extracted from the literature as well as expert elicitation for adult survival, juvenile survival, hatchling survival, proportion of juveniles that recruit into the adult stage, fecundity, proportion of females that breed annually, proportion of hatchlings that are female, clutch size, nest survival, and nest success (as described in the next section). This model was run for 50 annual time steps. This time frame was chosen because it reflected a time period in which existing threats and environmental conditions were likely to remain relevant, and patterns in the output were apparent within less than 50 years (i.e., no additional information was gained by running the model for a longer period of time).

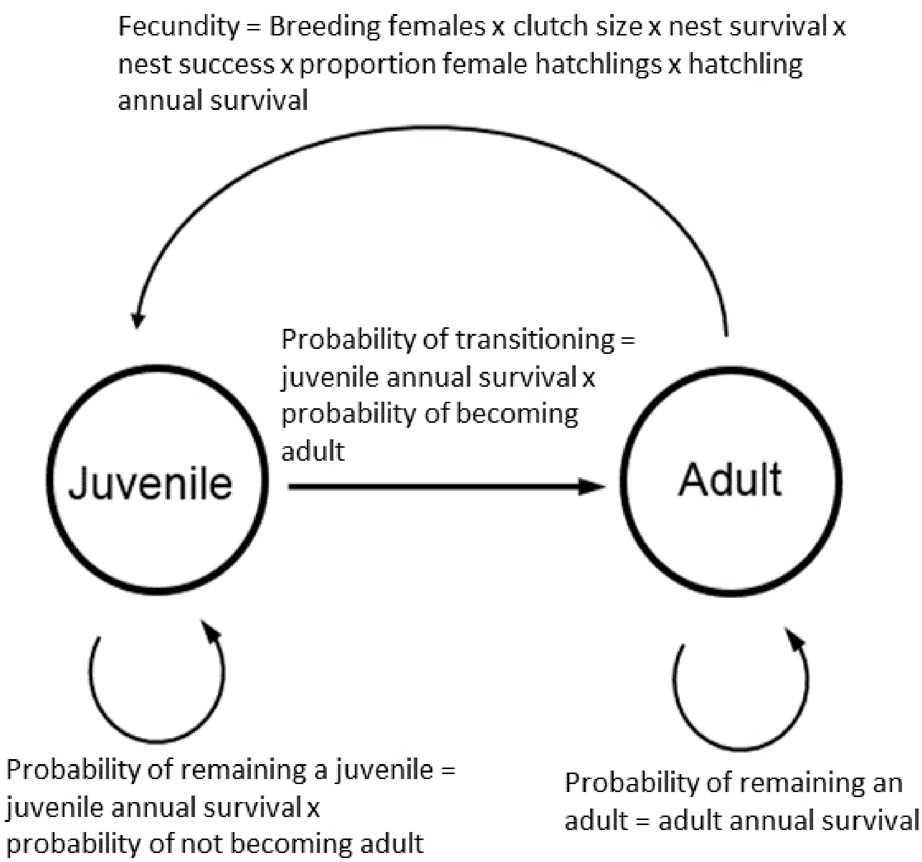


Figure 35. Alligator snapping turtle life cycle diagram for a female only two-stage pre-breeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals). At each time step, juveniles can remain in their current stage, which is the product of juvenile survival and one minus the annual

proportion of juveniles that recruit to the adult stage class. Alternatively, juveniles may transition to the adult stage with probability defined by the product juvenile survival and the annual proportion of juveniles that recruit to the adult stage class. Adults represent the terminal stage, therefore the probability that an individual remains in this stage is simply their annual survival probability. The arc shows the adult fecundity contribution, the number of juvenile females produced by each adult alligator snapping turtle annually. Adult fecundity is the combined product of the annual probability that an adult female breeds, clutch size, the proportion of nests in which one egg hatches (i.e., nest survival), the proportion of eggs from which a hatchling emerges in surviving nests (i.e., nest success), the proportion of female hatchlings, and hatchling survival from nest emergence to one year of age. The quantities used for each of the demographic parameters and their sources are given in Table 5.

5.1.1 Model Parameterization

The population model was parameterized (i.e., values input into the model) using demographic information pulled from literature on alligator snapping turtles or the closely related common snapping turtle, with information gaps filled in using expert elicitation (further details about how values were derived in Appendix E). When possible, we selected demographic parameters from reference populations that had minimal exposure to threats, meaning their parameter estimates were a closer approximation of the parameter's "true" value and less impacted by the effects of threats and stressors. We incorporated stochasticity (i.e., randomness, particularly due to annual variation or uncertainty) into our modeling framework by modeling each demographic parameter as a draw from a statistical distribution based on the parameter's mean and sampling standard deviation. These random draws were performed within a simulation framework that contained two nested loops: an inner loop that specified the number of annual time steps to project forward (50 years) and an outer loop that specified the number of times to replicate the 50-year loop (500 iterations). Final results were then compiled and summarized from all 500 iterations of the 50-year model, which varied between iterations because of the stochastic elements in the model.

Table 5. Summary of data sources used to parameterize the demographic population model for alligator snapping turtles. The Sampling Variance column reflects the amount of variation in the parameter's mean value (μ) attributed to sampling error, and is equal to μ×(1−μ)×0.10, with the exception of the clutch size demographic parameter. The Process Variance column reflects the temporal fluctuation in a parameter due to demographic or environmental stochasticity, and was set to (Sampling Variance)×0.05 for all parameters.

Demographic Parameter[a,b]	**Mean (μ)**	**Sampling Var. (σ_S^2)**	**Process Var. (σ_P^2)**	**Source**	**Source Location**
Juvenile survival (except Northern Mississippi - East Unit)	0.860	0.0277^2	0.01053^2	Folt et al. 2016	Spring Creek, Georgia
Juvenile survival Northern Mississippi - East Unit	0.730	0.0354^2	0.01082^2	Dreslik et al. 2017	Illinois
Juvenile to adult transition probability	0.020	0.0111^2	0.00889^2	Tucker and Sloan 1997	Louisiana
Adult survival	0.950	0.0174^2	0.00969^2	Folt et al. 2016	Spring Creek, Georgia
Proportion of females that breed annually	0.980	0.0112^2	0.00894^2	Dobie 1971	Southern Louisiana
Clutch Size	33.200	10.0000^2	5.00000^2	Weighted average[b]; Folt et al. 2016 (SD)	Multiple
Nest survival	0.130	0.0269^2	0.01037^2	Ewert et al. 2006	Lower Apalachicola River, Florida
Nest success	0.723	0.0358^2	0.01097^2	Ewert et al. 2006	Lower Apalachicola River, Florida
Proportion of female hatchlings	0.500	0.0400^2	0.01090^2	Expert opinion	–
Hatchling survival to one year	0.150	0.0285^2	0.01060^2	Expert opinion	–

[a]Demographic parameter mean, sampling variance, and process variance values apply to all modeled analysis units with the exception of juvenile survival (φ_J), which used different values for the Northern Mississippi – East Unit.
[b]Mean clutch size (*CS*) was derived using a weighted mean across multiple studies, using the sample size (number of nests) from each study as weights. Full details are given in Table E2.

Table 6. Threat-specific percent reductions (mean ± standard deviation) to alligator snapping turtle survival parameters, derived from remote expert elicitation among a team of taxon experts. These quantities were assumed to remain constant across the alligator snapping turtle's range, meaning that the percent reduction attributed to a specific threat was not assumed to vary among analysis units, though the proportion of the population exposed to a particular threat within an analysis unit may vary. The mean values contained within each cell represent the percent reductions under the "expert-elicited threat" scenarios, with or without conservation actions; these means were reduced or increased by 25% for the "decreased threat" and "increased threat" scenarios, respectively.

	Commercial Bycatch	**Recreational Bycatch**	**Hook Ingestion**	**Illegal Collection**	**Subsidized Nest Predators**
Hatchling Survival	0.0001 ± 0.0007	–	–	0.0047 ± 0.0028	–
Juvenile Survival	0.0403 ± 0.0258	0.0579 ± 0.0205	0.0615 ± 0.0195	0.0565 ± 0.0191	–
Adult Survival	0.0630 ± 0.0361	0.0741 ± 0.0351	0.0824 ± 0.0322	0.1947 ± 0.0625	–
Nest Survival	–	–	–	0.0110 ± 0.01167	0.6075 ± 0.1154

We used expert elicitation, as described in Section 4.5 of this report, to inform model parameters related to initial abundance, habitat loss mechanisms, the spatial extent of threats, and expected reductions to survival rates in response to specific threats. Expert responses included a minimum, maximum, and most likely estimate for numerical values, as well as the percent confidence of the respondent that the true value was between the minimum and maximum (Speirs-Bridge et al. 2010, p. 515). The most likely, minimum, and maximum values were used to back-calculate a distribution for each expert response, assumed to be a normal (bell curve) distribution, with a mean value and a measure of error. The mean and error values from each expert were combined into a weighted average, with each response weighted by the percent confidence of the expert in their response (more details in Appendix E).

During the expert elicitation process, we asked all participants to provide an estimate of total abundance within their analysis unit(s) of expertise and to clarify which sex or age classes (hatchlings, juveniles, adults) their estimate included. We then combined the responses across experts and initialized the starting abundance for each analysis unit assuming a stable stage distribution. However, except for the Northern Mississippi – East Unit, the expert-elicited abundance estimates included hatchlings, which were not included as a stage class in our model due to the pre-breeding census structure. For the purposes of initializing abundance in the remaining units, we re-formulated our projection model to reflect a postbreeding census structure with three stages (hatchlings, juveniles, adults) and multiplied the proportion of hatchlings at stable stage by the expert-elicited total abundance estimates to obtain the expected initial abundance of juveniles and adults only. We then created a series of stochastic variables to generate stage-specific initial abundances that were unique to each analysis unit, scenario, and iteration combination (See Appendix E for more details).

5.1.2 Model Scenarios

We projected future conditions for alligator snapping turtles under six different scenarios, across which the levels of threats and conservation actions varied. Species experts identified six primary potential threats that were likely to reduce stage-specific survival probabilities (Table 6): commercial fishing bycatch (influenced hatchling, juvenile, and adult survival), recreational fishing bycatch (influenced juvenile and adult survival), hook ingestion (influenced juvenile and adult survival), legal collection (influenced hatchling, juvenile, and adult survival), illegal collection (i.e., poaching; influenced hatchling, juvenile, and adult survival), and subsidized nest predators (influenced nest survival). The baseline nest survival value that we used (Table 5) was based on a study in which 40 of 46 nests (87%) were depredated by raccoons (*Procyon lotor;* Ewert et al. 2006, p. 67). Therefore the subsidized nest predator effect was meant to reflect additional threats to nest survival, such as depredation of emerging neonates from fire ants (*Solenopsis* spp.).

In the expert elicitation questionnaire, we asked the respondents to provide the following threat-related quantities: percent reduction to stage-specific survival rates attributed to each threat and the spatial extent of each threat within their analysis unit(s) of expertise. Thus, reductions to survival rates attributed to each threat were assumed to be the same across all analysis units, though the spatial extent of each threat (i.e., the proportion of the alligator snapping turtles exposed to the threat) varied among analysis units. For example, ingesting a fishing hook would be expected to produce the same percent reduction in survival across the entire range, though the probability that an individual alligator snapping turtle encounters that threat would vary among analysis units. As such, we determined that legal collection likely violated this assumption, as regulations for legal AST collection differed among states (LDFW 2019a, MFWP 2019, websites). Therefore, we decided to model the effects of legal collection as a direct reduction in juvenile and adult abundances (see Legal Collection section in Appendix E) that varied across analysis units, rather than a reduction to demographic parameters. For each analysis unit, we calculated threat-adjusted survival rates, accounting for reductions in stage-specific survival rates resulting from the percent reduction in survival expected from a given threat multiplied by the spatial extent of the threat, for each threat occurring in a given analysis unit. Lastly, to reflect spatial heterogeneity in threat occurrence and overlap within each analysis unit, we calculated a weighted average of each survival parameter, based on the probable occurrence and overlap of all possible threat combinations (see Threat Weighted Survival Estimates section in Appendix E).

We built scenarios around the potential uncertainty regarding *a*) the magnitude of the impact of threats on survival rates and *b*) the presence or absence of conservation actions. First, we defined three different "threat levels" by adjusting the demographic effect of each threat (percent reduction in stage-specific survival) up and down 25% relative to the compiled expert elicitation responses. The only exceptions to this structure, in addition to legal collection mentioned in the previous paragraph, was subsidized nest predators, in which the percent reduction to nest survival remained the same across all threat levels. These three levels reflect that there was a great deal of uncertainty in the impact that each threat has on survival rates, and allowed us to explore what the future condition might be if the mean estimates of threat magnitude either under- or overestimated the true impacts by 25%.

Next, we defined conservation action either as absent or present in the future. Where present, conservation action was modeled to reduce the spatial extent of threats (proportion of analysis unit exposed to threat) by 25%. This led to six different scenarios of expert-elicited threats, decreased threats, or high threats, with conservation action absent or present (Table 7). For example, the Decreased Threats + scenario reduced survival rate impacts by 25% and decreased the spatial extent of threats by 25%, relative to the mean expert-elicited quantities. Conservation actions that could decrease the spatial extent of threats include but are not limited to: increased enforcement or law enforcement presence to reduce poaching or bycatch on illegally set trot or limb lines, increasing the size of protected areas that prohibit recreational fishing or certain gear (e.g., trotlines, hoopnets), additional harvest restrictions in some areas, and management actions that reduce the densities of nest predators. The actual amount that any of these actions would influence the prevalence of threats will depend on factors like the time, money, personnel, and conservation partners available, but we selected a 25% reduction to explore how much a change of that amount affected future population dynamics.

For this report, scenarios with conservation actions present are indicated with a "+" (e.g., Expert-Elicited Threats +). Specific scenario names will be capitalized (e.g., Decreased Threats, Decreased Threats +), but threat levels will be in lowercase when we refer to both scenarios of a given threat level (e.g., decreased threats scenarios).

Table 7. Description of six future scenarios modeled for alligator snapping turtles for each analysis unit. Scenario names are given in quotation marks. Reductions or increases in value were in relation to the expert-elicited values. Threats manipulated across scenarios in this way included recreational and commercial bycatch, hook ingestion, and illegal collection.

	Conservation Absent	Conservation Present
Decreased Threat Magnitude	***"Decreased Threats"*** ● Impact of threats: *Reduced 25%* ● Spatial extent of threats: *Expert-elicited*	***"Decreased Threats + "*** ● Impact of threats: *Reduced 25%* ● Spatial extent of threats: *Reduced 25%*
Expert-Elicited Threat Magnitude	***"Expert-Elicited Threats"*** ● Impact of threats: *Expert-elicited* ● Spatial extent of threats: *Expert-elicited*	***"Expert-Elicited Threats + "*** ● Impact of threats: *Expert-elicited* ● Spatial extent of threats: *Reduced 25%*
Increased Threat Magnitude	***"Increased Threats"*** ● Impact of threats: *Reduced 25%* ● Spatial extent of threats: *Expert-elicited*	***"Increased Threats + "*** ● Impact of threats: *Increased 25%* ● Spatial extent of threats: *Reduced 25%*

Note that the threat level scenarios (expert-elicited, decreased, increased) varied in the magnitude of the impact of threats on survival where they occur, reflecting uncertainty in their true values. Conversely, the conservation scenarios (absent or present) varied in the spatial extent (the proportion of the population within the analysis unit exposed to the threat) of threats rather than their magnitude. For example, in either Expert-Elicited Threats scenario, the survival rate where recreational bycatch occurs is expected to remain the same whether conservation actions are present or absent, but in the Expert-Elicited Threats + scenario, the spatial extent of any given analysis unit exposed to recreational bycatch was reduced by 25% compared to the non-conservation scenario. Also note that only the means for survival rate impacts and spatial extent of threats, and not the standard deviations, were adjusted across the different scenarios.

Our modeling framework incorporated three effects believed to influence alligator snapping turtle demography that were not incorporated into scenarios as described above: legal collection, head-start and adult releases, and habitat loss. Unlike the threat-specific reductions in survival rates, these effects were consistent across all future condition scenarios, though they were subject to stochastic variation among iterations and time steps. Legal collection and release effects were applied directly to the stage-specific abundances at the beginning of each time step, whereas the effect of habitat loss was incorporated into the adult fecundity element in the transition matrix where its effect depended on total abundance.

Legal Collection

Regulations for legal collection differ among states, which did not align with analysis units (LDFW 2019*a*, website; MFWP 2019, website). Therefore, we decided to model the effects of legal collection as an annual reduction in abundance that varied across analysis units, rather than a reduction in survival rates. Collection of alligator snapping turtles is legal only in Mississippi and Louisiana. Legal collection in Mississippi was not incorporated into the model because the harvest restrictions (> 61 cm carapace length) functionally exclude females, which typically do not exceed 50 cm in carapace length (Folt et al. 2016, p. 24), and thus would have had no effect on our female-only population model. In Louisiana, current regulations allow for any angler with a freshwater fishing license to take one alligator snapping turtle of any size per day (LDWF 2019*b*, website). Within our modeling framework, we restricted the effects of legal collection to the two modeled analysis units that overlapped geographically with Louisiana: Southern Mississippi – East and Alabama. The annual reduction in abundance due to legal collection in these analysis units was based on using freshwater fishing license and specialty permit sales for wire traps and hoopnets (often used to catch turtles) from 2012-2017 as an index of take (LDWF 2019*b*, website), and the proportion of each analysis unit that overlapped Louisiana (See Appendix E for more details on how license and permit data were used).

Head-Starts and Adult Releases

Several states within the alligator snapping turtle's range have initiated head start release programs, in which alligator snapping turtles are raised for several years in captivity and then released into the wild population as juveniles (Dreslik et al. 2017, p. 13). Similarly, states also opportunistically release adult alligator snapping turtles confiscated from illegal activities (e.g., poaching) into wild populations. We included juvenile and adult releases within the model, though only for the first ten time steps within an iteration, to avoid having alligator snapping turtle population persistence be contingent on head start activities (i.e., conservation-dependent). We parameterized the releases in the model based on statistics from Illinois described in Dreslik et al. (2017, p. 13; juvenile females: ~30 individuals/year, adult females: ~12). The mean number of releases did not vary among analysis units or scenarios, but because of the uncertainty and variability in the simulations, the specific value drawn for each year in each unit in each iteration varied. Specifically, for the first ten time steps of each iteration, the number of released juveniles and adults were drawn from Poisson distributions.

Habitat Loss

We asked the species expert team to list habitat loss mechanisms within their analysis unit(s) of expertise. After adjusting for linguistic differences among responses (e.g., "desnagging" and "removal of large woody debris" were two answers that reflected the same mechanism), we summarized the number of unique habitat loss mechanisms within each analysis unit and calculated the mean across experts. We imposed a population ceiling (i.e., carrying capacity) that was annually reduced by a habitat loss rate, which equaled the mean number of unique threats in the unit, divided by 100. The initial population ceiling was determined based on the summarized expert elicitation values for the maximum possible number of alligator snapping turtles currently within the analysis unit, after adjusting for sex ratios and presence of hatchlings in the estimate. Thus, the population ceiling for each analysis unit at each time step was calculated deterministically and was not subject to stochastic variation across simulation iterations. To incorporate the effects of habitat loss on alligator snapping turtle demography within the model, we included a function that set adult fecundity to zero if total abundance (juveniles and adults) in any time step exceeded the population ceiling. While this function was included in the model, abundances were so far below population ceilings that the effect of habitat loss did not have an impact on modeling results (See Appendix E Figure 13).

5.1.3 Model Structure Summary, Limitations, Model Validation, and Sensitivity Analysis

Values for alligator snapping turtle initial abundances, demographic parameters, threats, and conservation measures were acquired from the literature and expert elicitation, as well as measures of error or uncertainty that were also incorporated into the stochastic model structure. For each analysis unit, at each annual time step, abundances of juveniles and adults were estimated based on *a*) baseline (minimal threats) demographic rates, *b*) changes in stage-specific survival rates due to the magnitude and spatial extent of threats in the analysis unit, *c*) reductions in abundance if legal collection is present in the unit, *d*) increases in abundance resulting from releases of juveniles and adults for the first 10 time steps, and *e*) a constantly declining population ceiling imposed by habitat loss and associated decline in adult fecundity if the population ceiling is exceeded. Of the five elements listed, only *b*), changes in survival rates in response to threats, varied across the six defined scenarios. For each analysis unit and scenario, this model structure was repeated for 50 annual time steps, and each 50-year stochastic projection was then repeated 500 times to generate summary statistics and predictions about the future condition of alligator snapping turtles.

Before we move on to present the modeling results, we must address the limitations of this model to keep in mind when interpreting the results. The precision and accuracy of model outputs depend heavily on the precision and accuracy of the information going into a model. In the case of the alligator snapping turtle, there is a large amount of uncertainty in the information that went into the model, including estimates of current abundance, age class proportions, impact of threats on stage-specific demographic rates, spatial extent of threats, and variability of these metrics across and within analysis units. We relied heavily on expert elicitation to obtain these values. Wherever possible, the uncertainty in these values was incorporated into the model structure itself, but others we were unable to address; for example the assumptions we had to make that baseline demographic rates are largely uniform across the range of the species. Future modeling efforts would be greatly improved with further study into these aspects of alligator snapping turtle biology, demography, response to and prevalence of threats, and how these vary across the range of the species.

We also acknowledge an ongoing concern raised with regard to the model used herein, is that it does not match the published estimates of population growth for the Folt et al. (2016, entire) model and conflicts with the perceived stability of AST populations from some catch-per-unit-effort studies for this species. As for validating model inputs, for several parameters, especially population threats as noted above had to rely on expert elicitation rather than data analysis or published literature. Furthermore, estimates of variance for many elicited parameters were small, suggesting that the experts generally agreed with each other, even though they the values were elicited independently from each expert.

For validating model predictions, the first thing to note is that the Folt et al. (2016, p. 23) paper primarily studied AST in an isolated area with little or no illegal collection, bycatch, or hook ingestion threats. The original formulation of the Folt model had multiple errors in the timing of abundance accounting (pre- vs post- breeding census) and in the juvenile to adult transition parameter (Caswell 2001, Kendall et al., 2019), and mis-specified (under-estimated) the variance for multiple parameters. Correcting those errors changed the prediction form a population that was growing 3% annually to one that was declining 3% annually. The modeling effort used in the SSA further modified the (corrected) Folt baseline model to account for dispersal of juveniles. Direct estimation of dispersal requires that mark-recapture data be collected according intensive study designs such as Pollock's robust design (Pollock et al., 1982, entire, Kendall et al., 1997, entire), which has not been applied to field studies of AST or closely related species. This modification (upward adjustment of the Juvenile survival parameter by 5%; Table E1) restored the threat-free, baseline population trajectory predictions to stability for all units except Northern Mississippi–East (Figure E12). Dispersal is more likely among the juvenile age class compared to adults, but no estimates of this parameter were available from mark recapture studies, so reincorporating these factors into the projection model seemed sensible.

An additional component of Folt et al. (2016) evaluated population status and trajectories for a population in Arkansas and one on a wildlife refuge in Oklahoma, where several of these threats are present, and the authors predicted rapid declines for those populations based on estimated demographic rates at those sites. These results in the published literature match fairly well with predicted trajectories for populations exposed to threats in the model. For example, in their simulation modeling, Steen and Robinson (2017, p. 1338) found that hook ingestion alone caused alligator snapping turtle populations that were increasing to reverse the predicted trend and decline by >50% in 30 years. Furthermore, since the completion of our work on the AST SSA report (RTM Version 1.0, October 2019), Ethan Kessler completed a PVA model for AST in southern Illinois (within the Northern Mississippi – East analysis unit) for his dissertation (Kessler, 2020). Radio telemetry was used to directly estimate true survival (i.e. survival probability is not biased low due to emigration or dispersal) and growth rates for AST populations (and the benefits of head starting and captive release programs). Kessler combined the parameters estimated from his study with productivity values from the peer reviewed literature into a PVA and reported a population growth rate (λ) of 0.95 (Kessler 2020, pg. 126) which is identical to the mean asymptotic population growth rates that we estimated for the Northern Mississippi – East unit across all scenarios (Table E6). Further, Kessler's analysis identified several of the same threats (especially recreational fishing bycatch), that were incorporated into the modeling used in the SSA, as key factors for future abundance and population growth rates. Of note, Kessler reported a catastrophic recreational bycatch incident in which a local resident illegally set a hoopnet and abandoned the device due to a sustained flooding event that limited trap accessibility. The abandoned hoopnet trapped and eventually drowned six adult and subadult alligator snapping turtles, including two individuals with radio transmitters (Kessler, personal communication). Kessler reports that the introduced population exhibits unstable demography and that reintroduction efforts are likely to fail unless bycatch can be reduced (Kessler 2020, pg. 116). It is not possible to fully validate model predictions from any single predictive model, but three independent models with similar results may bolster confidence in model predictions provided in the SSA.

Modelers also conducted additional model output sensitivity analyses using a regression-based approach to link realized lambda (year to year population change in the simulation output) to the stochastically generated threat levels and demographic rates each year. The regression analysis treats the realized lambda as the dependent variable and the stochastically drawn annual values of survival and each threat as independent variable in regression models. The effect (strength) of each parameter and threat can be assessed and compared using the regression slope estimates and model selection analysis to identify the most influential effects on population growth. This analysis concluded that the illegal collection impacts on adult survival and its spatial extent has the greatest effect on population growth in our model followed by hook ingestion impacts on adult survival and recreational fishing bycatch impacts on adult survival (Table E10). Each of these three threats are modeled as percent reductions in adult and juvenile survival, as well as the proportion of the population exposed to the threat, thus the results of this regression analysis match the Eigen elasticity analysis and expectations for this analysis, given long-lived species life history. Experts believed that illegal collection caused up to a 19.5% reduction in survival (Table E3) and that it affected a minimum of 30% of the population in all regions except Northern Mississippi–East (Table E4). Given the magnitude and spatial extent of this threat, it is not surprising that it has the greatest effect on realized lambda in the model.

Lastly, legal collection of AST is permitted in Mississippi and Louisiana. Therefore, the effects of legal harvest were not included in lambda regression sensitivity analysis (in which all analysis units were pooled) because it only occurs in the analysis units that overlap with Louisiana. During the SSA model building process, we originally elicited the spatial extent and magnitude of legal collection from the expert team to implement the effect as a reduction in survival (as done for other threats such as commercial or recreational bycatch). However, we realized that the magnitude of reduction on survival attributed to legal collection likely varied across states due to the differences in policy/take limits: size restrictions in Mississippi (AST < 24 in. carapace length are protected) effectively prohibit the legal collection of females, whereas one AST of any size can be legally collected per day in Louisiana.. Therefore, we modeled legal collection as a direct reduction to abundance. Louisiana does not collect data regarding legal collection of turtle species, therefore, we used an index based on the annual number of freshwater fishing permits sold in Louisiana from 2010-2017 (Eq. 5 in Appendix E). To provide additional clarification, we ran an alternative set of scenarios that omitted legal harvest for the Southern Mississippi–East and Alabama analysis units (that overlap with Louisiana), and compared them against the model output in the SSA (Table E11). Note that with the exception of Table E11, all other output in the SSA contains the effects of legal harvest in these units. In general, the probability of quasi-extinction (p_{QX}) was insensitive to the inclusion of legal collection for both analysis units, though the probability of extinction was slightly reduced for the Alabama analysis unit.

5.2 Future Condition Results by Analysis Unit

We derived a series of summary statistics to evaluate alligator snapping turtle trends in abundance and evaluate potential variation among analysis units and alternate scenarios. Here we define an extirpation event as the total population (juveniles + adults) declining to zero individuals, whereas a decline to less than 5% of the starting population size was considered quasi-extirpation. For each analysis unit and scenario combination, we estimated extirpation and quasi-extirpation probabilities by calculating the proportion of iterations in which the population reached those thresholds (calculated elasticity values and stable stage distributions can be found in Appendix E). For the iterations in which abundance reached extirpation or quasi-extirpation, we estimated the mean number of years until the population reached the specified threshold. Additionally, we generated the asymptotic population growth rate (λ) for each of the analysis unit/scenario combinations. A λ value of 1 indicates stability, while values greater than 1 indicate growth, and values less than 1 indicate decline. Probabilities of extirpation or quasi-extirpation are discussed in this document using guidance from the Intergovernmental Panel on Climate Change about how to describe uncertainty (Table 8; Mastrandrea et al. 2011, p. 680). In the written summaries below for each analysis unit, we highlight the time to extirpation or quasi-extirpation only for those scenarios where extirpation or quasi-extirpation were at least about as likely as not to occur (at least 33% probability).

Table 8. Guidance from the Intergovernmental Panel on Climate Change about how to describe uncertainty (Mastrandrea et al. 2011, p. 680).

Term	Likelihood of the Outcome
Virtually certain	99-100% probability
Very likely	90-100% probability
Likely	66-100% probability
About as likely as not	33-66% probability
Unlikely	0-33% probability
Very unlikely	0-10% probability
Exceptional unlikely	0-1% probability

5.2.1 Southern Mississippi – East Analysis Unit

Alligator snapping turtle abundances in the Southern Mississippi – East Analysis Unit were predicted to decline over the next 50 years in all scenarios (Figure 36). Predicted declines were more rapid the higher the threat level and were slightly mediated by conservation actions (mean λ = 0.85, 0.81, and 0.78 respectively for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, and mean λ = 0.87, 0.85, and 0.82 respectively for

Decreased Threat +, Expert-Elicited Threat +, and Increased Threat + scenarios; Appendix E Table E6). Compared to initial abundances, after the first 10 years of the simulation, mean abundance was predicted to decline by 76-82% under decreased threats scenarios, 83-88% under expert-elicited threats scenarios, and 87-92% under increased threats scenarios. Halfway through the simulation, after 25 years, mean abundance was predicted to decline by 95-100% compared to initial abundance across all six scenarios (See Appendix E for mean abundances at 5-year intervals throughout the entire 50-year simulation).

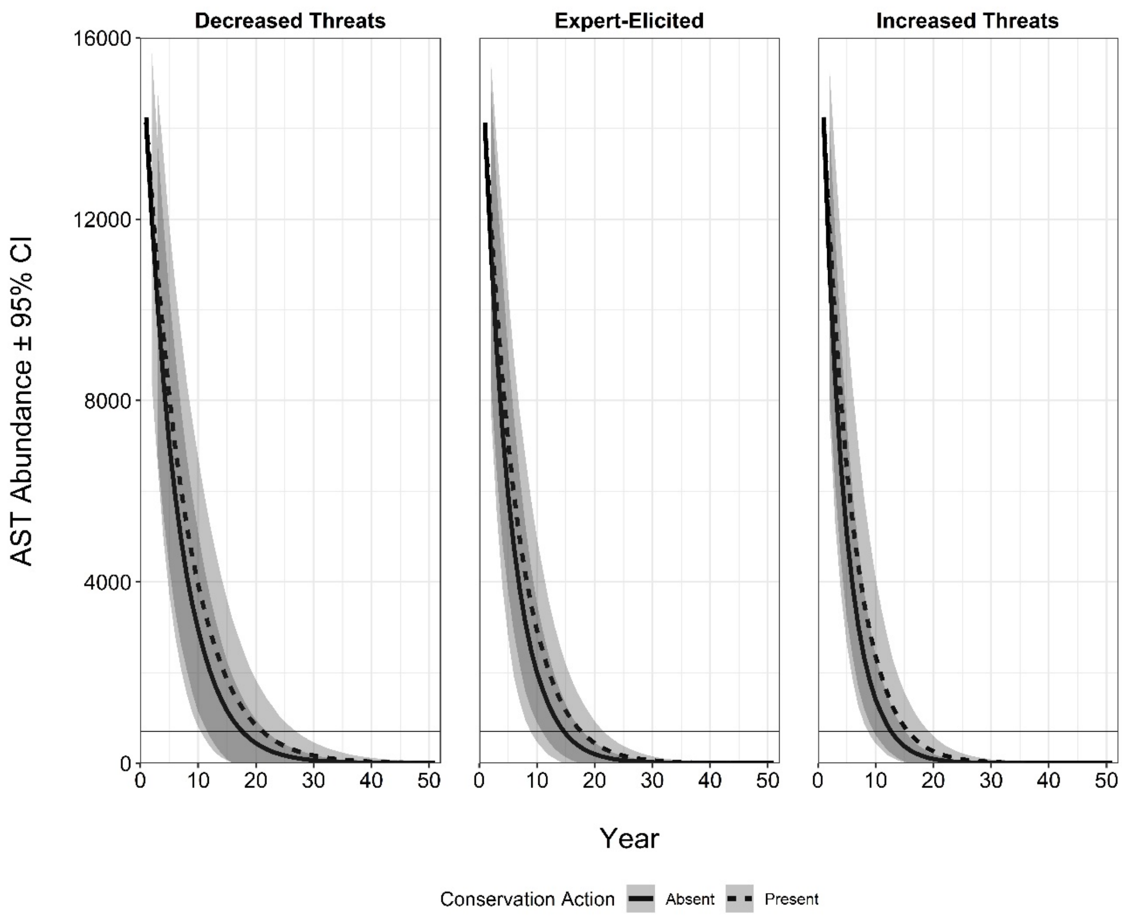


Figure 36. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Southern Mississippi – East Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI). The panels indicate the scenario's threat level: decreased, expert-elicited, or increased. The scenarios with and without conservation actions for each threat level overlap and cannot be distinguished in this figure. The analysis unit-specific quasi-extirpation threshold (5% of initial abundance) is shown by the thin horizontal line.

Though abundance declined in all scenarios, the probability of extirpation within 50 years depended heavily on the threat levels and presence or absence of conservation actions. Without conservation, the species was unlikely to be extirpated in this unit within 50 years under the Decreased Threat scenario, likely to be extirpated under the Expert-Elicited Threat scenario, and very likely to become extirpated under the Increased Threat scenario. With conservation, the species was exceptionally unlikely to be extirpated under the Decreased

Threat + scenario, very unlikely to be extirpated under the Expert-Elicited Threat + scenario, and about as likely as not to be extirpated under the Increased Threat + scenario. In scenarios where extirpation was at least as likely as not to occur, extirpation occurred on average after 41-47 years (Table 9). While the likelihood that the species will become completely extinct varied by scenario, quasi-extirpation where abundances fell below 5% of current levels was virtually certain in all scenarios. Predicted time to quasi-extirpation averaged 18-21 years under the decreased threats scenarios, 15-18 years under the expert-elicited threats scenarios, and 13-16 years under the increased threats scenarios, with the lower bound of each range predicted when conservation actions were present.

Table 9. Probability and time to extirpation and quasi-extirpation for alligator snapping turtles in the Southern Mississippi – East Analysis Unit. The six scenarios included three different threat levels (decreased, expert-elicited, and increased), with conservation action absent (TH) or present (TH+). For each scenario, we calculated the probability of extirpation (Prob Ext) and quasi-extirpation (Prob Q-Ext) as the proportion of the 500 replicates in which the total population (adults and juveniles) declined to zero or less than 5% of the starting population size, respectively. For only those replicates in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, (Time to Ext and Time to Q-Ext, respectively). Mean quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all replicates) given in parentheses. An asterisk (*) indicates only a single simulation crossed the threshold, precluding a standard deviation calculation.

Threat Level	*Prob Ext*		*Time to Ext*		*Prob Q-Ext*		*Time to Q-Ext*	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.434	0.058	47.46 ± 3.05	49.45 ± 1.92	1.0	1.0	17.69 ± 2.40	20.9 ± 3.34
			(41,53)	(43, 51)			(11, 29)	(14, 35)
Expert-Elicited	0.950	0.476	43.33 ± 3.97	47.49 ± 2.84	1.0	1.0	14.89 ± 1.75	17.74 ± 2.34
			(32, 51)	(39, 51)			(10, 22)	(12, 26)
Increased	0.998	0.856	38.07 ± 3.37	44.92 ± 3.87	1.0	1.0	12.97 ± 1.39	15.74 ± 1.98
			(30, 49)	(33, 51)			(9, 18)	(11, 25)

5.2.2 Northern Mississippi – East Analysis Unit

Alligator snapping turtle abundances in the Northern Mississippi – East Analysis Unit were predicted to increase for the next decade, but then decline over the next 50 years in all scenarios (Figure 37). Predicted declines were consistent across scenarios mean $\lambda = 0.95$ for all scenarios with and without conservation; Appendix E Table E6). Compared to initial abundances, after the first 10 years of the simulation, mean abundance was predicted to increase by at least 200% across every scenario. By halfway through the simulation after 25 years, mean abundances were predicted to fall but still remain over 32% higher than initial abundances. By the end of the 50-year simulation however, abundances were predicted to decline by 47-51% compared to initial abundances in the scenarios without conservation actions, and 44-48% in the scenarios with conservation actions (See Appendix E for mean abundances at each time step).

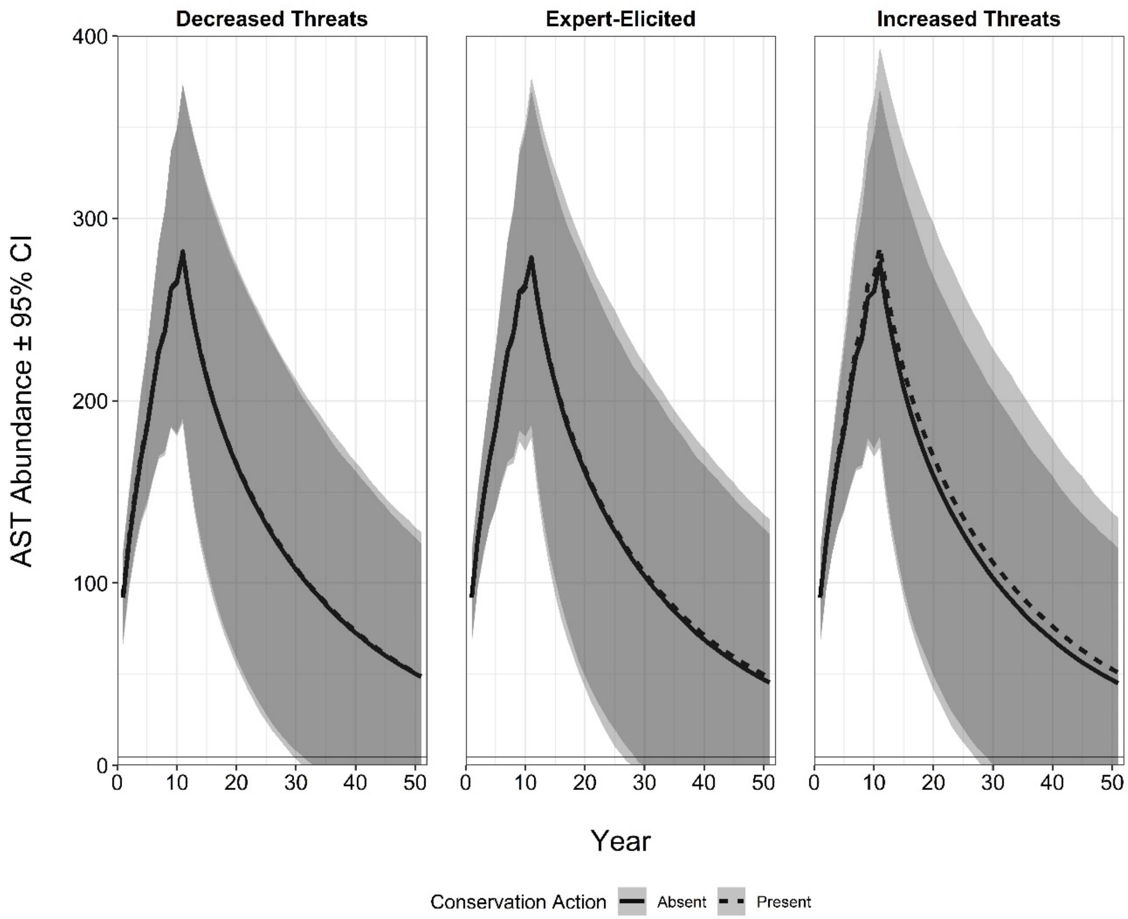


Figure 37. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Northern Mississippi – East Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI). The panels indicate the scenario's threat level: decreased, expert-elicited, or increased. Solid lines represent trajectories with conservation action absent, while dashed lines represent trajectories with conservation actions present. The analysis unit-specific quasi-extirpation threshold (5% of initial abundance) is shown by the thin horizontal line.

Though abundance eventually declined in all scenarios after initial increases, the species was exceptionally unlikely to very unlikely to be extirpated in this unit within 50 years under any modeled scenario (Table 10). Quasi-extirpation was similarly very unlikely to occur in any scenario.

Table 10. Probability and time to extirpation and quasi-extirpation for alligator snapping turtles in the Northern Mississippi – East Analysis Unit. The six scenarios included three different threat levels (decreased, expert-elicited, and increased), with conservation action absent (TH) or present (TH+). For each scenario, we calculated the probability of extirpation (Prob Ext) and quasi-extirpation (Prob Q-Ext) as the proportion of the 500 replicates in which the total population (adults and juveniles) declined to zero or less than 5% of the starting population size, respectively. For only those replicates in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, (Time to Ext and Time to Q-Ext, respectively). Mean quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all replicates) given in parentheses. Dashes (–) indicate that no simulation reached the extirpation or quasi-extirpation threshold, meaning that t_{EX} or t_{QX} were not calculated, whereas an asterisk (*) indicates only a single simulation crossed the threshold, precluding a standard deviation calculation.

Threat Level	*Prob Ext*		*Time to Ext*		*Prob Q-Ext*		*Time to Q-Ext*	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0	0	–	–	0.020	0.038	45.90 ± 4.01 (38, 51)	48.21 ± 2.90 (42, 51)
Expert-Elicited	0	0.002	–	51.00 ± * (51, 51)	0.016	0.036	48.00 ± 4.11 (39, 51)	46.72 ± 3.39 (39, 51)
Increased	0	0	–	–	0.024	0.020	45.42 ± 3.42 (41, 51)	46.60 ± 2.50 (42, 50)

5.2.3 Alabama Analysis Unit

Alligator snapping turtle abundances in the Alabama Analysis Unit were predicted to decline over the next 50 years in all scenarios (Figure 38). Predicted declines were more rapid the higher the threat level and were slightly mediated by conservation actions (mean λ = 0.83, 0.78, and 0.75 respectively for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, and mean λ = 0.86, 0.82, and 0.79 respectively for Decreased Threat +, Expert-Elicited Threat +, and Increased Threat + scenarios; Appendix E Table E6). Compared to initial abundances, after the first 10 years of the simulation, mean abundance was predicted to decline by 75-83% under decreased threat scenarios, 83-90% under expert-elicited threat scenarios, and 88-93% under increased threat scenarios. Halfway through the simulation, after 25 years, mean abundance was predicted to decline by 97-100% compared to initial abundance across all six scenarios, with declines of 100% after 50 years (See for mean abundances at each time step).

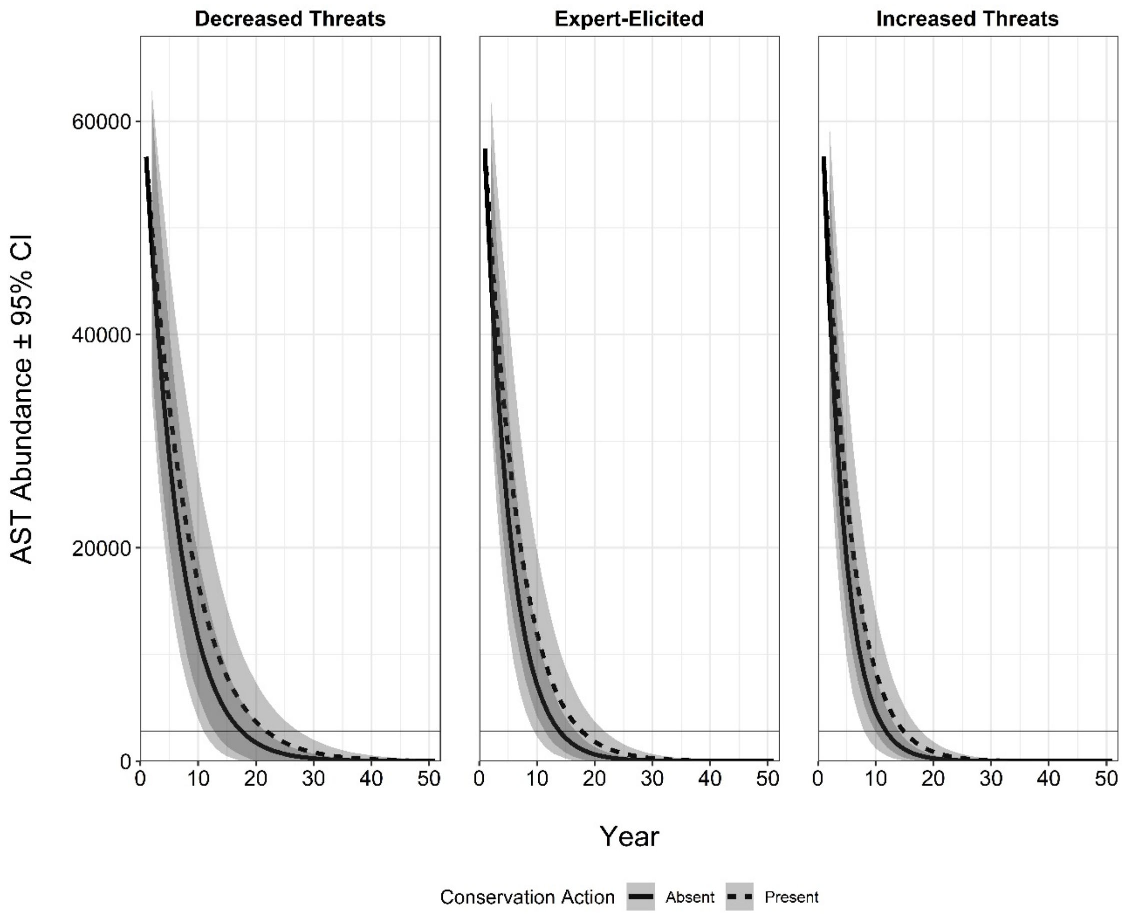


Figure 38. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Alabama Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI). The panels indicate the scenario's threat level: low, moderate, or high. The scenarios with and without conservation actions for each threat level overlap and cannot be distinguished in this figure. The analysis unit-specific quasi-extirpation threshold (5% of initial abundance) is shown by the thin horizontal line.

Though abundance declined in all scenarios, the probability of extirpation within 50 years depended heavily on the threat levels and presence or absence of conservation actions. Without conservation, the species was unlikely to be extirpated in this unit within 50 years under the Decreased Threat scenario, likely to be extirpated under the Expert-Elicited Threat scenario, and virtually certain to become extirpated under the Increased Threat scenario. With conservation, the species was exceptionally unlikely to be extirpated under the Decreased Threat + scenario, unlikely to be extirpated under the Expert-Elicited Threat + scenario, and about as likely as not to be extirpated under the Increased Threat + scenario. In scenarios where extirpation was at least as likely as not to occur, extirpation occurred on average after 40-47 years (Table 11). While the likelihood that the species will become completely extinct varied by scenario, quasi-extirpation where abundances fell below 5% of current levels was virtually certain in all scenarios. Predicted time to quasi-extirpation averaged 18-22 years under the decreased threats scenarios, 14-18 years under the expert-elicited threats scenarios, and 12-15 years under the increased threats scenarios, with the lower bound of each range predicted when conservation actions were present.

Table 11. Probability and time to extirpation and quasi-extirpation for alligator snapping turtles in the Alabama Analysis Unit. The six scenarios included three different threat levels (decreased, expert-elicited, and increased), with conservation action absent (TH) or present (TH+). For each scenario, we calculated the probability of extirpation (Prob Ext) and quasi-extirpation (Prob Q-Ext) as the proportion of the 500 replicates in which the total population (adults and juveniles) declined to zero or less than 5% of the starting population size, respectively. For only those replicates in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, (Time to Ext and Time to Q-Ext, respectively.) Mean quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all replicates) given in parentheses. An asterisk (*) indicates only a single simulation crossed the threshold, precluding a standard deviation calculation.

Threat Level	*Prob Ext*		*Time to Ext*		*Prob Q-Ext*		*Time to Q-Ext*	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.130	0.002	48.91 ± 2.09 (43, 51)	51 ± * (51, 51)	1.0	1.0	17.68 ± 2.27 (12, 29)	22.84 ± 3.20 (14, 33)
Expert-Elicited	0.846	0.114	45.64 ± 3.36 (36, 51)	49.14 ± 2.23 (40, 51)	1.0	1.0	14.20 ± 1.6 (10, 20)	17.91 ± 2.27 (13, 26)
Increased	1.0	0.658	40.19 ± 3.47 (30, 51)	47.21 ± 2.76 (40, 51)	1.0	1.0	12.11 ± 1.35 (8, 16)	15.11 ± 1.72 (12, 23)

5.2.4 Apalachicola Analysis Unit

Alligator snapping turtle abundances in the Apalachicola Analysis Unit were predicted to decline over the next 50 years in all scenarios (Figure 39). Predicted declines were more rapid the higher the threat level and were slightly mediated by conservation actions (mean λ = 0.87, 0.84, and 0.81 respectively for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, and mean λ = 0.90, 0.87, and 0.85 respectively for Decreased Threat +, Expert-Elicited Threat +, and Increased Threat + scenarios; Appendix E Table E6). Compared to initial abundances, after the first 10 years of the simulation, mean abundance was predicted to decline by 55-64% under decreased threats scenarios, 65-74% under expert-elicited threats scenarios, and 72-82% under increased threats scenarios. Halfway through the simulation after 25 years, mean abundance was predicted to decline by 90-99% compared to initial abundance across all six scenarios, and were predicted to decline by 99-100% after 50 years in all scenarios (See Appendix E for mean abundances at each time step).

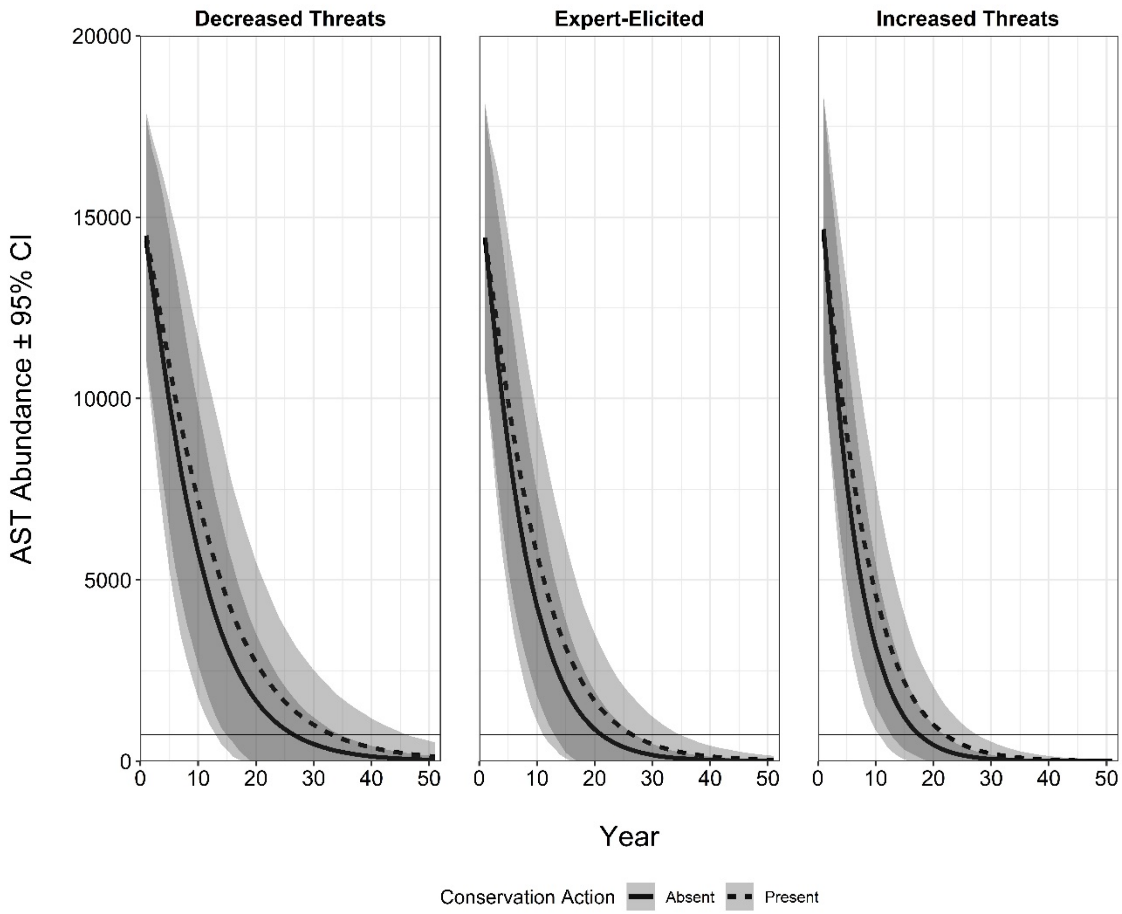


Figure 39. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Apalachicola Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI). The panels indicate the scenario's threat level: low, moderate, or high. The scenarios with and without conservation actions for each threat level overlap and cannot be distinguished in this figure. The analysis unit-specific quasi-extirpation threshold (5% of initial abundance) is shown by the thin horizontal dotted line.

Though abundance declined in all scenarios, the probability of extirpation within 50 years depended heavily on the threat levels and presence or absence of conservation actions. Without conservation, the species was exceptionally unlikely to be extirpated in this unit within 50 years under the Decreased Threat scenario, unlikely to be extirpated under the Expert-Elicited Threat scenario, and likely to become extirpated under the Increased Threat scenario. With conservation, the species was exceptionally unlikely to be extirpated under the Decreased Threat + scenario and the Expert-Elicited Threat + scenario, and very unlikely to be extirpated under the Increased Threat + scenario. In scenarios where extirpation was at least as likely as not to occur, extirpation occurred on average after 47 years (Table 12). While the likelihood that the species will become completely extinct varied by scenario, quasi-extirpation where abundances fell below 5% of current levels was very likely to virtually certain to occur within 50 years in all scenarios. Predicted time to quasi-extirpation was similar across scenarios, averaging 45-48 years depending on the scenario.

Table 12. Probability and time to extirpation and quasi-extirpation for alligator snapping turtles in the Apalachicola Analysis Unit. The six scenarios included three different threat levels (decreased, expert-elicited, and increased), with conservation action absent (TH) or present (TH+). For each scenario, we calculated the probability of extirpation (Prob Ext) and quasi-extirpation (Prob Q-Ext) as the proportion of the 500 replicates in which the total population (adults and juveniles) declined to zero or less than 5% of the starting population size, respectively. For only those replicates in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, (Time to Ext and Time to Q-Ext, respectively.) Mean quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all replicates) given in parentheses. Dashes (–) indicate that no simulation reached the extirpation or quasi-extirpation threshold.

Threat Level	*Prob Ext*		*Time to Ext*		*Prob Q-Ext*		*Time to Q-Ext*	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.004	0	49.5 ± 0.71 (49, 50)	–	0.990	0.980	33.11 ± 6.09 (19, 51)	32.44 ± 6.1 (20, 51)
Expert-Elicited	0.124	0.006	49.02 ± 2.05 (44, 51)	50.67 ± 0.58 (50, 51)	1.0	1.0	26.28 ± 4.65 (16, 47)	32.04 ± 5.79 (18, 51)
Increased	0.660	0.052	46.82 ± 3.15	48.92 ± 1.94	1.0	1.0	21.21 ± 3.25	26.22 ± 4.75

5.2.5 Western, Southern Mississippi – West, and Northern Mississippi – West Analysis Units

The Western, Southern Mississippi – West, and Northern Mississippi – West analysis units were not included in the future simulation modeling because we did not have adequate input data to do so. However, we have no evidence that alligator snapping turtle demographic trends in response to threats in these analysis units would behave dramatically different from the range of analysis units that we did model. While we do not have precise abundance estimates in the future or probabilities of extirpation or quasi-extirpation, it is likely that alligator snapping turtles in these analysis units will decline along similar trajectories as the modeled analysis units, meaning they face a high likelihood of quasi-extirpation within the next 50 years.

5.3 Future Condition Overall Results

In this section we summarize the above analysis unit results to describe the future resilience, redundancy, and representation for alligator snapping turtles.

5.3.1 Future Resilience

Resilience is expected to drastically decline across all analysis units under all scenarios. We modeled scenarios that reflected uncertainty in the impact of threats on alligator snapping turtle demography, and all threat levels (decreased, expert-elicited, and increased) produced mean growth rates (λ) indicating population decline. Predicted abundances were likely to very likely to virtually certain to drop below 5% of current abundances within 50 years under all scenarios in the Southern Mississippi – East, Alabama, and Apalachicola analysis units (Table 13). The only analysis unit for which quasi-extirpation was not consistently likely was the Northern Mississippi – East Analysis Unit. Though the risk of quasi-extirpation was lower in this analysis unit than the others, this was in part an artefact of the way that quasi-extirpation thresholds were defined, as a percentage of the initial abundance. In terms of raw abundance, the Northern Mississippi – East analysis unit was predicted on average to support fewer than 51 female alligator snapping turtles (as we used a female-only demographic model) with or without conservation actions. Thus, even though quasi-extirpation risks were lower than other analysis units, the predicted abundances for this unit still indicate that alligator snapping turtles will become very rare or disappear from this analysis unit.

Time to quasi-extirpation varied across analysis units and scenarios, but in general, the first analysis unit likely to reach the quasi-extirpation threshold was the Alabama Unit (12-22 years), followed by the Southern Mississippi – East Unit (after an average of 14-25 years depending on the scenario), the Apalachicola Unit (21-33 years), and finally the Northern Mississippi – East Unit where quasi-extirpation was not likely.

Table 13. Summary of quasi-extirpation probabilities for all alligator snapping turtle Analysis Units across all six future scenarios.

Analysis Unit	***Conservation Absent***			***Conservation Present***		
Threat Level	*Decreased*	*Expert-Elicited*	*Increased*	*Decreased*	*Expert-Elicited*	*Increased*
Southern Mississippi – East	1.0	1.0	1.0	1.0	1.0	1.0
Northern Mississippi – East	0.02	0.02	0.02	0.04	0.04	0.02
Alabama	1.0	1.0	1.0	1.0	1.0	1.0
Apalachicola	0.99	1.0	1.0	0.98	1.0	1.0

After 50 years, the mean female abundance in any given analysis unit was not predicted to exceed 133 individuals in any scenario (Figure 40). As we did for the current condition, we scaled future predicted abundances (after 25 years and after 50 years of the simulation) to the area of open water in each analysis unit to aid in comparing abundances among units of different sizes (Table 14).

Table 14. Initial and final projected alligator snapping turtle abundances expressed as raw abundances and scaled to 1,000 hectares of open water in each modeled analysis unit. For final abundances, we included in this table only the more optimistic decreased threats scenario (averaged across both conservation scenarios); final abundances for expert-elicited and increased threats scenarios were lower. Note that initial abundances are not equal to those reported in the current conditions section because the initial abundances used in the simulation model a) were generated from 500 draws per scenario/analysis unit combination from a probability distribution that incorporated uncertainty surrounding current abundance, and b) included females only, while current condition abundances included males and females.

Analysis Unit	Initial Mean Abundance	Per 1,000 ha Open Water	25-Year Mean Abundance - Decreased Threats	Per 1,000 ha Open Water	50-Year Mean Abundance - Decreased Threats	Per 1,000 ha Open Water
Alabama	56,648	174.7	1,101	3.4	24	0.1
Apalachicola	14,419	90.1	1,138	7.1	84	0.5
South MS – East	14,188	15.7	476	0.5	17	<0.1
North MS – East	93	0.4	127	0.6	49	0.2

Resilience refers to the ability of populations (or in our case analysis units as we are unable to delineate populations with currently available information) to withstand stochastic disturbances (e.g., demographic, environmental stochasticity). Abundance is central to resilience, as small populations are more vulnerable to perturbations than larger populations. We compiled the best information available about alligator snapping turtles, their demographic rates, and threats, and the resulting simulation model predicted dramatic declines in abundance, and thus resilience, over the next 50 years across all analysis units. Abundances in nearly every analysis unit were predicted to decline by more than 95%, resulting in drastically lowered abilities of alligator snapping turtle populations within analysis units to withstand stochastic events, if alligator snapping turtle populations persist at all.

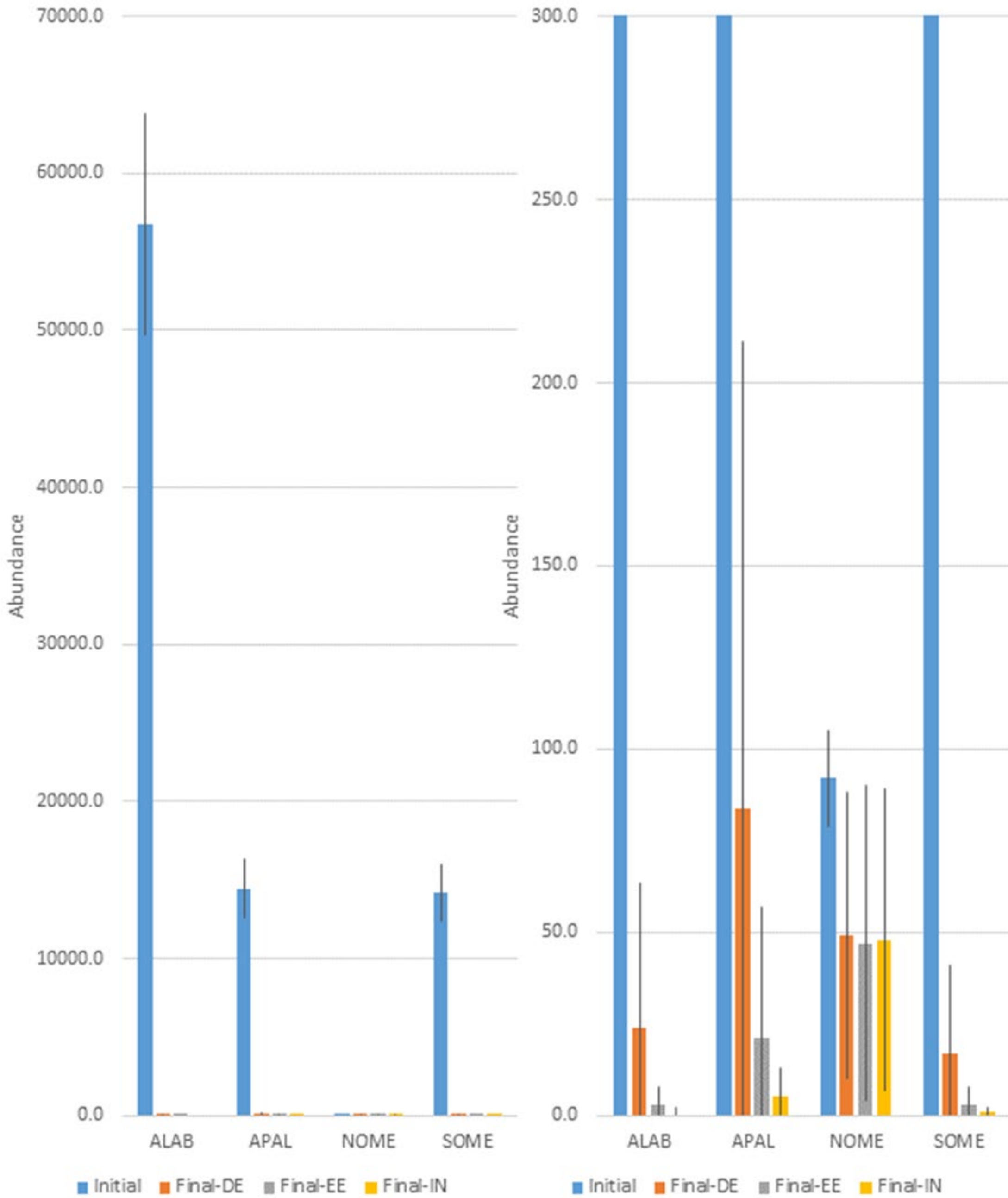


Figure 40. Initial and final projected alligator snapping turtle abundances with standard deviations (zoomed in on right panel). The four modeled analysis units are shown (ALAB = Alabama, APAL = Apalachicola, NOME = Northern Mississippi – East, SOME = Southern Mississippi – East), with initial female-only abundances in blue, and final abundances after 5- years under Decreased (DE), Expert-Elicited (EE), and Increased (IN) threats scenarios. Within each threat level, scenarios with and without conservation actions were averaged together for this figure. Note that initial abundances are not equal to those reported in the current conditions section because the initial abundances used in the simulation model a) were generated from 500 draws per scenario/analysis unit combination from a probability distribution that incorporated uncertainty surrounding current abundance, and b) included females only, while current condition abundances included males and females.

To provide additional clarification regarding model results, model sensitivity analysis, indicated that the population growth rate (lambda) and other model outputs, were most sensitive to changes in adult and juvenile survival parameters (Table E7). In developing the model, modelers used an elasticity analysis rather than a "sensitivity" analysis because the output from an elasticity analyses are more easily interpreted. Elasticity analysis (essentially measuring the percent change in lambda, or any other output metric, relative to percent changes in the input demographic rates (Caswell, 2001)), concluded that even very small changes in the adult survival rate could lead to large changes in predicted lambda and future abundance. Most of the threats that the Core Team identified (hook ingestion, illegal collection, etc.) were factors that affect adult or juvenile survival, and so large changes in population growth and predicted future abundance are expected to occur when those effects are incorporated into the model. For example, experts indicated that hook ingestion was likely to negatively affect adult survival and could cause an up to 8% decline in survival rate (Table E3) in areas where trotline and other fishing was permitted, dropping survival from 95% to 87% (0.95 x (1-0.08)). That one threat alone changes the trajectory of the population from stable or increasing to rapidly declining. Adding additional threats on top of hook ingestion, leads to precipitous predicted declines and very high extinction probability.

5.3.2 Future Representation

Future representation, referring to the ability of the species to adapt to changing environmental conditions over time, is similarly predicted to decline rapidly as alligator snapping turtles in every representative unit decline in abundance to quasi-extirpation or true extirpation (Table 15). The loss of alligator snapping turtles across all representative units would represent losses in genetic diversity (3 broad genetic lineages), life history diversity along a north-south gradient, and finer scale genetic differences among drainages within the larger genetic lineages.

Table 15. Initial and final projected alligator snapping turtle abundances in each representative unit. For final abundances, we included only the more optimistic decreased threats scenario (averaged across both conservation scenarios); final abundances for expert-elicited and increased threats scenarios were lower. Note that initial abundances are not equal to those reported in the current conditions section because the initial abundances used in the simulation model a) were generated from 500 draws per scenario/analysis unit combination from a probability distribution that incorporated uncertainty surrounding current abundance, and b) included females only, while current condition abundances included males and females.

Representative Unit	# Analysis Units Modeled / Total # Analysis Units	Initial Mean Abundance[a]	25-Year Mean Abundance - Decreased Threats	50-Year Mean Abundance - Decreased Threats
Alabama	1 / 1	56,648	1,101	24
Southern MS	1 / 2	14,188	476	17
Western	0 / 1	Not modeled	Not modeled	Not modeled
Apalachicola	1 / 1	14,419	1,138	84
Northern MS	1 / 2	93	127	49
Total	5 / 8	86,510	7,952	838

[a] Initial abundance only shown for those analysis units that were modeled.

5.3.3 Future Redundancy

Future redundancy, or the ability to withstand catastrophic events, for alligator snapping turtles is expected to decline drastically over the next 50 years. Our future simulation model operated at the scale of the analysis unit, and was limited to the units for which we had sufficient data, so we cannot provide precise predictions about which states or counties are most likely to lose or retain alligator snapping turtle biological populations in the future. At the analysis unit scale however, all units were predicted to lose resilience at such a high rate that no redundancy of resilient populations or analysis units is expected to remain across the landscape (See Table 15 above, where each representative unit is equal to one of the 5 modeled analysis units). Where alligator snapping turtles persist in the future, they are predicted to be rare and not found in resilient groupings. Analysis units were predicted to reach quasi-extirpation thresholds in some cases within the next two decades, with more units becoming quasi-extirpated each decade after that. The addition of conservation actions, or different assumptions about the impact of threats on alligator snapping turtle demography altered the time to quasi-extirpation by about a decade at most, typically less. No scenarios resulted in stable or increasing redundancy within representative units or range-wide.

5.4 Summary of Future Conditions and Viability

For the alligator snapping turtle to maintain viability, it needs to have resilient populations that are able to withstand stochastic events and maintain ecological and genetic diversity, which will help preserve the breadth of adaptive capacity of the species. In addition, the populations need to be spread across its range in a way that reduces the chance that a catastrophic event is not likely to lead to the species extinction.

Resilience for all analysis units is expected to decline drastically across all analysis units under all scenarios. We modeled scenarios that reflected uncertainty in the impact of threats on alligator snapping turtle demography, and all scenarios produced mean growth rates indicating population decline. With the exception of the Northern Mississippi – East Unit, all other analysis units were predicted to be quasi-extirpated within 50 years with a probability of over 98%. Though the risk of quasi-extirpation was lower in the Northern Mississippi – East Unit than the others, the predicted abundances for this unit were still low, fewer than 51 female turtles, and still indicate that alligator snapping turtles will become very rare or disappear from this analysis unit.

Time to quasi-extirpation varied across analysis units and scenarios, but in general, the first analysis unit likely to reach the quasi-extirpation threshold was the Alabama Unit (12-22 years), followed by the Southern Mississippi – East Unit (after an average of 14-25 years depending on the scenario), the Apalachicola Unit (21-33 years), and finally the Northern Mississippi – East Unit where quasi-extirpation was not likely.

The Western, Southern Mississippi – West, and Northern Mississippi – West analysis units were not included in the futures simulation modeling because we did not have adequate input data to do so. However, we have no evidence that alligator snapping turtle demographic trends in response to threats in these analysis units would be dramatically different from the range of analysis units that were modeled, therefore, it is likely that alligator snapping turtles in these analysis units will decline along similar trajectories as the modeled analysis units.

Future representation, referring to the ability of the species to adapt to changing environmental conditions over time, is similarly predicted to decline rapidly as alligator snapping turtles in every representative unit decline in abundance to quasi-extirpation or true extirpation. The loss of alligator snapping turtles across all representative units would represent losses in genetic diversity (3 broad genetic lineages), life history diversity along a north-south gradient, and finer scale genetic differences among drainages within the larger genetic lineages.

Future redundancy, or the ability to withstand catastrophic events, for alligator snapping turtles is expected to decline drastically over the next 50 years. Our future simulation model operated at the scale on the analysis unit, so we cannot provide precise predictions about what states or counties are most likely to lose or retain alligator snapping turtles in the future. At the analysis unit scale however, all units were predicted to lose resilience at such a high rate that redundancy is not expected to remain across the landscape. Where alligator snapping turtles persist in the future, they are likely to be rare and not found in resilient groupings. Analysis units were predicted to reach quasi-extirpation thresholds in some cases within the next two decades, with more units becoming quasi-extirpated each decade after that within our 50-year modeling time frame. The addition of conservation actions, or different assumptions about the impact of threats on alligator snapping turtle demography altered the time to quasi-extirpation by about a decade at most, typically less. No scenarios resulted in stable or increasing redundancy.

This concludes our assessment of alligator snapping turtle needs, current condition, and future condition. It is apparent that based on the current state of knowledge, alligator snapping turtles are predicted to decline in abundance and range. However, the current state of knowledge for this species is full of uncertainty. This assessment should be updated as new information becomes available, and in particular can be strengthened with further study into population delineations, abundance and occupancy, variation in demographic rates across the range of the species, the impacts of threats on demography, and prevalence of threats across the landscape.

LITERATURE CITED

Anthony, T., J.D. Riedle, M.B. East, B. Fillmore, and D.B. Ligon. 2015. Monitoring of a reintroduced population of juvenile alligator snapping turtles. Chelonian Conservation and Biology 14:43–48.

Boundy, J., and C. Kennedy. 2006. Trapping survey results for the alligator snapping turtle (*Macrochelys temminckii*) in southeastern Louisiana, with comments on exploitation. Chelonian Conservation and Biology 5:3–9.

Cagle, F. and A. H. Chaney. 1950. Turtle populations in Louisiana. Amer. Midl. Nat. 43(2):383-388.

Carr, J.L., A. Bass, L.B. White, and L. Besenhofer. 2007. Alligator snapping turtle movements and reproduction at Black Bayou Lake National Wildlife Refuge Final Report. University of Louisiana at Monroe Department of Biology & Museum of Natural History. Monroe, La. 76pp.

Carr, J.L., S.M. Holcomb, and M. Ray. 2010. Alligator snapping turtle (*Macrochelys temminckii*) ecology and reproduction at Black Bayou Lake National Wildlife Refuge, Ouachita Parish, Louisiana. Final Report. University of Louisiana at Monroe. 108pp.

Carr, J.L., S.R. Holcomb, and M.J. Ray. 2011. Basking in the alligator snapping turtle, *Macrochelys temminckii* (Testudines: Chelydridae). IRCF Reptiles and Amphibians 18:2–5.

Carr, J. L., A. C. Johnson and A. K. Shook. 2020. Coordinated surveys for alligator snapping turtles [final project report]. University of Louisiana, Monroe, Louisiana. 27 pp.

Carr, J. L. and A. C. Johnson. 2021. Coordinated surveys for alligator snapping turtles in Louisiana - Part II [final project report]. University of Louisiana, Monroe, Louisiana. 36 pp.

Carter, L.M., J.W. Jones, L. Berry, V. Burkett, J.F. Murley, J. Obeysekera, P.J. Schramm, and D. Wear. 2014. Ch. 17: Southeast and the Caribbean. Climate Change Impacts in the United States: The Third National Climate Assessment. J.M. Melillo, T.C. Richmond, and G.W. Yohe, Eds., U.S. Global Change Research Program: 396–417. doi:10.7930/J0NP22CB.

Caswell, H., 2001. Matrix Population Models, Construction, Analysis and Interpretation. Sinauer Associates, Inc. Publishers, Sunderland, MA, USA.

Chaffin, K., T.M. Norton, K. Gilardi, R. Poppenga, J.B. Jensen, P. Moler, C. Cray, E.S. Dierenfeld, T. Chen, M. Oliva, F.C. Origgi, S. Gibbs, L. Mazzaro, and J. Mazet. 2008. Health assessment of free–ranging alligator snapping turtles (*Machrochelys temminckii*) in Georgia and Florida. Journal of Wildlife Diseases 44:670–686.

Congdon, J.D., G.L. Breitenbach, R.C. van Loben Sels, and D.W. Tinkle. 1987. Reproduction and nesting ecology of snapping turtles (*Chelydra serpentina*) in southeastern Michigan. Herpetologica 43:39–54.

Crother, B. I., J. Boundy, F. T. Burbrink, J. A. Campbell, B. I. Crother, K. de Queiroz, D. R. Frost, D. M. Green, R. Highton, J. B. Iverson, F. Kraus, R. W. McDiarmid, J. R. Mendelson III, P. A. Meylan, R. A. Pyron, T. W. Reeder, M. E. Seidel, S. G. Tilley, D. B. Wake. 2000. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Society for the Study of Amphibians and Reptiles. Herpetological Circular 29 (J.J. Moriarty, editor). 82pp.

Dobie, J.L. 1971. Reproduction and growth in the alligator snapping turtle, *Macrochelys temminckii* (Troost). Copeia 1971:645–658.

Ernst, C.H., and R.W. Barbour. 1989. Turtles of the World. Smithsonian Institution Press. Washington, D.C., and London. 313pp.
Ewert, M.A. 1976. Nests, nesting and aerial basking of *Macroclemys* under natural conditions, and comparisons with *Chelydra* (Testudines: Chelydridae). Herpetological 32: 150–156.
Ewert, M.A., and D.R. Jackson. 1994. Nesting ecology of the alligator snapping turtle, *Macroclemys temminckii*, along the Lower Apalachicola River, Florida. Florida Game and Fresh Water Fish Commission. Nongame Wildlife Program Final Report. 45pp+vi. Tallahassee, FL.
Ewert, M.A., D.R. Jackson, and P.E. Moler. 2006. *Macrochelys temminckii* – alligator snapping turtle. *In* Meylan, P.A. (Ed.), Biology and Conservation of Florida Turtles. Chelonian Research Monographs, Lunenburg, Massachusetts, p. 58–71.
Fitzgerald, L.A., and R.E. Nelson. 2011. Thermal biology and temperature-based habitat selection in a large aquatic ectotherm, the alligator snapping turtle, *Macroclemys temminckii*. Journal of Thermal Biology 36:160-166.
Folt, B., and J.C. Godwin. 2013. Status of the alligator snapping turtle (*Macrochelys temminckii*) in South Alabama with comments on its distribution. Chelonian Conservation and Biology 12:211–217.
Folt, B., and C. Guyer. 2015. Evaluating recent taxonomic changes for alligator snapping turtles (Testudines: Chelydridae). Zootaxa 3947(3):447–450.
Folt, B., J.B. Jensen, A. Teare, and D. Rostal. 2016. Establishing reference demography for conservation: a case study of *Macrochelys temminckii* in Spring Creek, Georgia. Herpetological Monographs 30:21–33.
Frazer, N.B., J.W. Gibbons, and T.J. Owens. 1990. Turtle trapping: Preliminary tests of conventional wisdom. American Society of Ichthyologist and Herpetologist 4:1150–1152.
Godwin, J. 2004. Status Survey for the Alligator Snapping Turtle (*Macrochelys temminckii*) at Selected Sites in the Mobile Basin Drainage of Alabama: Year 1. Alabama Natural Heritage Program. Montogomery, Al. 29pp.
Harrel, J.B., C.M. Allen, and S.J. Hebert. 1996. Movement and habitat use of subadult alligator snapping turtles (*Macrochelys temminckii*) in Louisiana. The American Midland Naturalist Journal 135:60–67.
Harrel, J.B., C.M. Allen, and S.J. Hebert. 1997. One year growth of subadult Macroclemys temminckii in a Louisiana Bayou. Herpetological Review 28(3): 128-129.
Holcomb, S.R. 2010. Survivorship and sources of mortality in nests of the alligator snapping turtle, *Macrochelys temminckii*, at Black Bayou Lake National Wildlife Refuge. MS Thesis. University of Louisiana at Monroe. 111pp.
Holcomb, S.R., and J.L. Carr. 2011*a*. Hatchling emergence from naturally incubated alligator snapping turtle (*Macrochelys temminckii*) nests in northern Louisiana. Chelonian Conservation and Biology 10(2):222-227.
Holcomb, S.R., and J.L. Carr. 2011*b*. Infestation of a naturally incubated nest of the alligator snapping turtle (*Macrochelys temminckii*) by the phorid fly *Megaselia scalaris*. The Southwestern Naturalist 56:427–429.
Holcomb, S.R., and J.L. Carr. 2013. Mammalian depredation of artificial alligator snapping turtle (*Macrochelys temminckii* Troost) nests in North Louisiana. Southeastern Naturalist 12:478–491.
Hopkins, B.C., J.D. Willson, and W.A. Hopkins. 2013. Mercury exposure is associated with negative effects on turtle reproduction. Environmental Science and Technology 47:2416–2422.

Howey, C.A.F., and S.A. Dinkelacker. 2009. Habitat selection of the alligator snapping turtle (*Macrochelys temminckii*) in Arkansas. Journal of Herpetology 43:589–596.
Howey, C.A.F., and S.A. Dinkelacker. 2013. Characteristics of a historically harvested alligator snapping turtle (*Machrochelys temminckii*) population. Copeia 2013: 58–63.
Iowa Department of Natural Resources. 2015. Iowa Wildlife Action Plan. Des Moines, Iowa. 375pp.
Iverson, J.B. 1991. Patterns of survivorship in turtles (Order Testudines). Canadian Journal of Zoology 69: 385-391.
Iverson, J.B., C.P. Balgooyen, K.K. Byrd, and K.K. Lyddan. 1993. Latitudinal variation in egg and clutch size in turtles. Canadian Journal of Zoology 71:2448–2461.
Jackson, D.R., and J.B. Jensen. 2003. Macrochelys temminckii (alligator snapping turtle) reproduction. Herpetological Review 34:363-363.
Jackson, C.G., Jr., and A. Ross. 1971. The occurrence of barnacles on alligator snapping turtle, *Macroclemys temminckii* (Troost). Journal of Herpetology 5:188-189.
Jensen, J.B., and W.S. Birkhead. 2003. Distribution and status of the alligator snapping turtle (*Macrochelys temminckii*) in Georgia. Southeastern Naturalist 2:25–34.
Johnson, A.C. 2020. Factors related to occupancy of the alligator snapping turtle (*Macrochelys temminckii*) in Louisiana, with a demographic analysis of a lentic population. M.S. Thesis, University of Louisiana, Monroe, Louisiana. 44 pp.
Kendall, W.J., J.D. Nichols, and J.E. Hines. 1997. Estimating temporary emigration using capture-recapture data with Pollock's robust design. Ecology 78:563–578.
Kendall, B.E. 2019. mpmtools: Tools for building and analyzing matrix population models. R package version 0.2.0, https://github.com/BruceKendall/mpmtools.
Kendall, B.E., Fujiwara, M., Diaz-Lopez, J., Schneider, S., Voigt, J., Wiesner, S., 2019. Persistent problems in the construction of matrix population models. Ecol. Modell. 406, 33–43. https://doi.org/10.1016/j.ecolmodel.2019.03.011
Kessler, E.J., C.A. Phillips, S.R. Ballard, and M.J. Dreslik. 2017. The first record of alligator snapping turtle (Macrochelys temminckii) in Illinois in 30 years. Southeastern Naturalist 16:29-35.
Kessler, E.J., 2020. Assessment of freshwater turtle reintroduction. University of Illinois at Urbana-Champaign.
King, R.L., B.P. Hepler, L.L. Smith, and J.B. Jensen. 2016. The status of Macrochelys temminckii (alligator snapping turtle) in the Flint River, GA, 22 years after the close of commercial harvest. Southeastern Naturalist 15:575–585.
Koons, J. and A. F. Scott. 1993. Summer movements of native and introduced Alligator Snapping Turtles, *Macroclemys temminckii*, in Kentucky Lake. Abstract. ASB Bulletin 40:134.
Louisiana Department of Wildlife and Fisheries (LDWF). 2019*a*. Reptile and amphibian fishing regulations. http://www.wlf.louisiana.gov/fishing/reptile_amphib–regulations. Accessed 8 May 2019.
Louisiana Department of Wildlife and Fisheries (LDWF). 2019*b*. Statistics: Recreational fishing license sales by parish (2013–2017). http://www.wlf.louisiana.gov/licenses/statistics. Accessed 8 May 2019.
Lescher, T.C., J.T. Briggler, and Z. Tang–Martínez. 2013. Relative abundance, population structure, and conservation of alligator snapping turtles (*Macrochelys temminckii*) in Missouri between 1993–1994 and 2009. Chelonian Conservation and Biology 12:163–168.
Ligon, D.B., M. East, and D. Riedle. 2012. The importance of maintaining long–term studies when monitoring long–lived organisms: lessons from a medium–term study.

In: Walde, A. D., D. Riedle, and A. Currylow (Eds.). Program and Abstracts of the Tenth Annual Symposium on the Conservation and Biology of Tortoises and Freshwater Turtles. Tucson, AZ: Turtle Survival Alliance, p. 40.

Mastrandrea, M.D., K.J. Mach, G. Plattner, O. Edenhofer, T.F. Stocker, C.B. Field, K.L. Ebi, and P.R. Matschoss. 2011. The IPCC AR5 guidance note on consistent treatment of uncertainties: a common approach across the working groups. Climatic Change 108:675–691.

Matthews, S.N., L.R. Iverson, M.P. Peters, and A.M. Prasad. 2018. Assessing potential climate change pressures across the conterminous United States: mapping plant hardiness zones, heat zones, growing degree days, and cumulative drought severity throughout this century. RMAP–NRS–9. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 31pp.

Miller, J.L., D.M. Thompson, J. Heywood, and D.B. Ligon. 2014. Nest–site selection among reintroduced *Macrochelys temminckii*. The Southwestern Naturalist 59:188–192.

Mississippi Fish, Wildlife, and Parks (MFWP). 2019. Information on harvest of turtles and other non–game wildlife. https://www.mdwfp.com/fishing–boating/freshwater–commercial/turtle–information/. Accessed 8 May 2019.

Moler, P.E. 1996. Alligator snapping turtle distribution and relative abundance. Final Report, Study Number 7544. Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, USA. 13pp.

Moore, D.B., D.B. Ligon, B.M. Fillmore, and S.F. Fox. 2013. Growth and viability of a translocated population of alligator snapping turtles (*Macrochelys temminckii*). Herpetological Conservation and Biology 8:141–148.

Morris, M.A., and M.J. Sweet. 1985. Size, age, and growth of an alligator snapping turtle, *Macroclemys temminckii*, from Illinois. Transactions of the Illinois State Academy of Science 78:241–245.

Petokas, P.J. and M.M. Alexander. 1980. The nesting of *Chelydra serpentina* in northern New York. Journal of Herpetology 14:239–244.

Pollock, K.H. 1982. A capture-recapture design robust to unequal probability of capture. Journal of Wildlife Management 46:757–760.

Powders, V.N. 1978. Observations on oviposition and natural incubation of eggs of the alligator snapping turtle, *Macroclemys temminckii*, in Georgia. Copeia 1978: 154–156.

Pritchard, P.C.H. 1989. The alligator snapping turtle: Biology and conservation. Milwaukee Public Museum, Milwaukee, WI, USA.

Pryor, S.C., D. Scavia, C. Downer, M. Gaden, L. Iverson, R. Nordstrom, J. Patz, and G.P. Robertson. 2014. Ch. 18: Midwest. Climate Change Impacts in the United States: The Third National Climate Assessment, J. M. Melillo, T. C. Richmond, and G. W. Yohe, Eds., U.S. Global Change Research Program: 418–440. doi:10.7930/J0J1012N.

Reed, R.N., J. Congdon, and J.W. Gibbons. 2002. The alligator snapping turtle [*Macrochelys (Macroclemys) temminckii*]: a review of ecology, life history, and conservation, with demographic analyses of the sustainability of take from wild populations. Report, Division of Scientific Authority, United States Fish and Wildlife Service, Aiken, South Carolina. 17pp.

Rhodin, A. G.J., J.B. Iverson, R. Bour, U. Fritz, A. Georges, H.B. Shaffer, and P.P. van Dijk. 2017. Turtles of the world annotated checklist and atlas of taxonomy, synonymy,

distribution, and conservation status (8th ed.). Turtle taxonomy working group. 292pp.
Riedle, J.D., P.A. Shipman, S.F. Fox, and D.M. Leslie, Jr. 2006. Microhabitat use, home range, and movements of the alligator snapping turtle, *Macrochelys temminckii*, in Oklahoma. The Southwestern Naturalist 51(1):35–40.
Riedle, J.D., D.B. Ligon, and K. Graves. 2008*a*. Distribution and management of alligator snapping turtles, *Macrochelys temminckii*, in Kansas and Oklahoma. Transactions of the Kansas Academy of Science 111:21–28.
Riedle, J.D., P.A. Shipman, S.F. Fox, J.C. Hackler, and D.M. Leslie, Jr. 2008*b*. Population structure of the alligator snapping turtle, *Macrochelys temminckii*, on the western edge of its distribution. Chelonian Conservation and Biology 7:100–104.
Roman, J., S.D. Santhuff, P.E. Moler and B.W. Bowen. 1999. Population structure and cryptic evolutionary units in the alligator snapping turtle. Conservation Biology 13:135–142.
Shafer, M., D. Ojima, J.M. Antle, D. Kluck, R.A. McPherson, S. Petersen, B. Scanlon, and K. Sherman. 2014. Ch. 19: Great Plains. Climate Change Impacts in the United States: The Third National Climate Assessment, J. M. Melillo, T. C. Richmond, and G. W. Yohe, Eds., U.S. Global Change Research Program: 441–461. doi:10.7930/J0D798BC.
Shipman, P.A. 1993. Alligator snapping turtle (*Macroclemys temminckii*) habitat selection, movements, and natural history in southeast Kansas. MS Thesis. Emporia State University. 101pp.
Shipman, P.A., D.R. Edds,, and D. Blex. 1991. Report on the recapture of the alligator snapping turtle (*Macroclemys temminckii*) in Kansas. Kansas Herpetological Society Newsletter 85:8-9.
Shook, A. K. 2020. Anthropogenic and reproductive factors influencing the population biology of an aquatic megafaunal species (*Macrochelys temminckii*, Chelydridae). M.S. Thesis, University of Louisiana, Monroe, Louisiana, 59 pp.
Sloan, K.N., and D. Taylor. 1987. Habitats and movements of adult alligator snapping turtles in Louisiana. Proceedings of the Annual Conference of the Southeast Association of Fish and Wildlife Agencies 41:343–348.
Smith, H.M. 1955. The generic name of the alligator snapper turtle. Herpetologica 11:16.
Spangler, S.J. 2017. Ecology of hatchling alligator snapping turtles (*Macrochelys temminckii*). Missouri State University Graduate Theses. 66pp.
Speirs–Bridge, A., F. Fidler, M. McBride, L. Flander, G. Cumming, and M. Burgman. 2010. Reducing overconfidence in the interval judgments of experts. Risk Analysis 30:512–523.
Steen, D.A., J.P. Gibbs, K.A. Buhlman, J.L. Carr, B.W. Compton, J.D. Congdon, J.S. Doody, J.C. Godwin, K.L. Holcomb, D.R. Jackson, F.J. Janzen, G. Johnson, M.T. Jones, J.T. Lamer, T.A. Langen, M.V. Plummer, J.W. Rowe, R.A. Saumure, J.K. Tucker, and D.S. Wilson. 2012. Terrestrial habitat requirements of nesting freshwater turtles. Biological Conservation 150:121-128.
Steen, D.A., Robinson, O.J., 2017. Estimating freshwater turtle mortality rates and population declines following hook ingestion. Conserv. Biol. 31, 1333–1339. https://doi.org/10.1111/cobi.12926
Thomas, T.M., M.C. Granatosky, J.R. Bourque, K.L. Krysko, P.E. Moler, T. Gamble, E. Suarez, E. Leone, K.M. Enge, and J. Roman. 2014. Taxonomic assessment of Alligator Snapping Turtles (Chelydridae: Macrochelys), with the description of two new species from the southeastern United States. Zootaxa 3786:141–165.

Thompson, D.M., D.B. Ligon, J.C. Patton, and M. Papes. 2016. Effects of life–history requirements on the distribution of a threatened reptile. Conservation Biology 31:427–436.
Trauth, S.E., D.S. Siegel, M.L. McCallum, D.H. Jamieson, and A. Holt. 2016. Long-term monitoring and recovery of a population of alligator snapping turtles, *Macrochelys temminckii* (Testudines: Chelydridae), from a northeastern Arkansas stream. Journal of the Arkansas Academy of Science 70:235-247.
Trauth, S.E., H.W. Robison, and M.V. Plummer. 2004. The amphibians and reptiles of Arkansas. University of Arkansas Press.
Trauth, S.E., J.D. Wilhide, and A. Holt. 1998. Population structure and movement patterns of alligator snapping turtles (*Macroclemys temminckii*) in Northeastern Arkansas. Chelonian Conservation and Biology 3:64–70.
Tucker, A.D., and K.N. Sloan. 1997. Growth and reproductive estimates from Alligator Snapping Turtles, *Macroclemys temminckii*, taken by commercial harvest in Louisiana. Chelonian Conservation and Biology 2:587–592.
Turtle Taxonomy Working Group (Rhodin, A.G.J., J.B. Iverson, R.F.U. Bour, A. Georges, H.B. Shaffer, and P.P. van Dijk. 2017. Turtles of the World: Annotated Checklist and Atlas of Taxonomy, Synonymy, Distribution and Conservation Status (8th Ed.). *In:* Rhodin, A.G.J., J.B. Iverson, P.P. van Dijk, R.A. Saumure, K.A. Buhlmann, P.C.H. Pritchard, and R.A. Mittermeier (Eds.). Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group. Chelonian Research Monographs 7:1–292.
U.S. Air Force (USAF). 2017. Integrated natural resource management plan for Barksdale Air Force Base Bossier City, Louisiana. 141pp.
U.S. Air Force (USAF). 2013. Integrated natural resource management plan for Little Rock Air Force Base Jacksonville, Arkansas. 144pp.
U.S. Air Force (USAF). 2014. Integrated natural resource management plan for Moody Air Force Base, Georgia. 597pp.
U.S. Army National Guard (USANG). 2018. Integrated natural resource management plan for Robinson Maneuver Training Center Little Rock, Arkansas. 329pp.
U.S. Army National Guard (USANG). 2018. Integrated natural resource management plan for Ft. Chaffee, Arkansas. 154pp.
U.S. Navy (USN). 2012. Integrated natural resource management plan for Naval Air Station Joint Reserve Base New Orleans, Louisiana. Prepared by Gulf South Research Corporation for Department of the Navy, Naval Facilities Engineering Command Southeast Jacksonville, Florida. 414pp.
U.S. Army (USA). 2018. Integrated natural resource management plan for Red River Army Depot, Texas. Prepared by Tetra Tech, Inc. USACE, Mobile District Mobile, AL. 254pp.
U.S. Army (USA). 2015. Integrated natural resource management plan for Ft. Benning, Georgia. Directorate of Public Works, Environmental Management Division, Fort Benning, Georgia. 1044pp.
U.S. Fish and Wildlife Service (USFWS). 2005. Inclusion of alligator snapping turtle (*Macroclemys* [=*Macrochelys*] *temminckii*) and all species of map turtle (*Graptemys* spp.) in Appendix III to the Convention on International Trade in Endangered Species of Wild Fauna and Flora. Federal Register 70:74700–74712.
U.S. Fish and Wildlife Service (USFWS). 2016. USFWS Species Status Assessment Framework: An integrated analytical framework for conservation. Version 3.4, dated August 2016. 21pp.

U.S. Fish and Wildlife Service (USFWS). 2018. Report from the Law Enforcement Management Information System (LEMIS). Department of Interior dated 11/20/2018.
Webb, R. G. 1995. The date of publication of Gary's *Catalogue of Shield Reptiles*. Chelonian Conservation and Biology 1(4):322-323.
Yang, L., S. Jin, P. Danielson, C. Homer, L. Gass, A. Case, C. Costello, J. Dewitz, J. Fry, M. Funk, B. Grannemann, M. Rigge, and G. Xian. 2018. A new generation of the United States National Land Cover Database: Requirements, research priorities, design, and implementation strategies. ISPRS Journal of Photogrammetry and Remote Sensing 146:108–123.

APPENDIX A – Alligator Snapping Turtle Suitable Habitat

Spatial analysis of the Alligator Snapping Turtle range was performed to determine the extent of suitable habitats available and the amount of lands in conservation.

The lands in conservation analysis was accomplished using the USGS Protected Areas Database (PAD-US, https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap/science/protected-areas) as the baseline dataset. It was compared for accuracy against the U.S. Forest Service land ownership data (https://data.fs.usda.gov/geodata/), the U.S. Fish & Wildlife Service Cadastral Data (https://www.fws.gov/gis/index.html) and other in-house datasets. Spatial accuracy and analysis were performed for all datasets using ESRI ArcGIS Pro 2.4.1. Acre summaries were calculated for each Analysis Unit and presented into federal, state, local and private ownership categories.

Suitable habitats were determined using the 2016 National Land Cover Data (https://www.mrlc.gov/). Three landcover classes were identified as suitable habitat; emergent herbaceuous wetlands, open water and woody wetlands. Analysis units were buffered to clip data past unit boundaries, land cover data was converted from raster to vector for accurate acreage calculations then data were intersected/clipped to individual analysis units for acreage summaries.

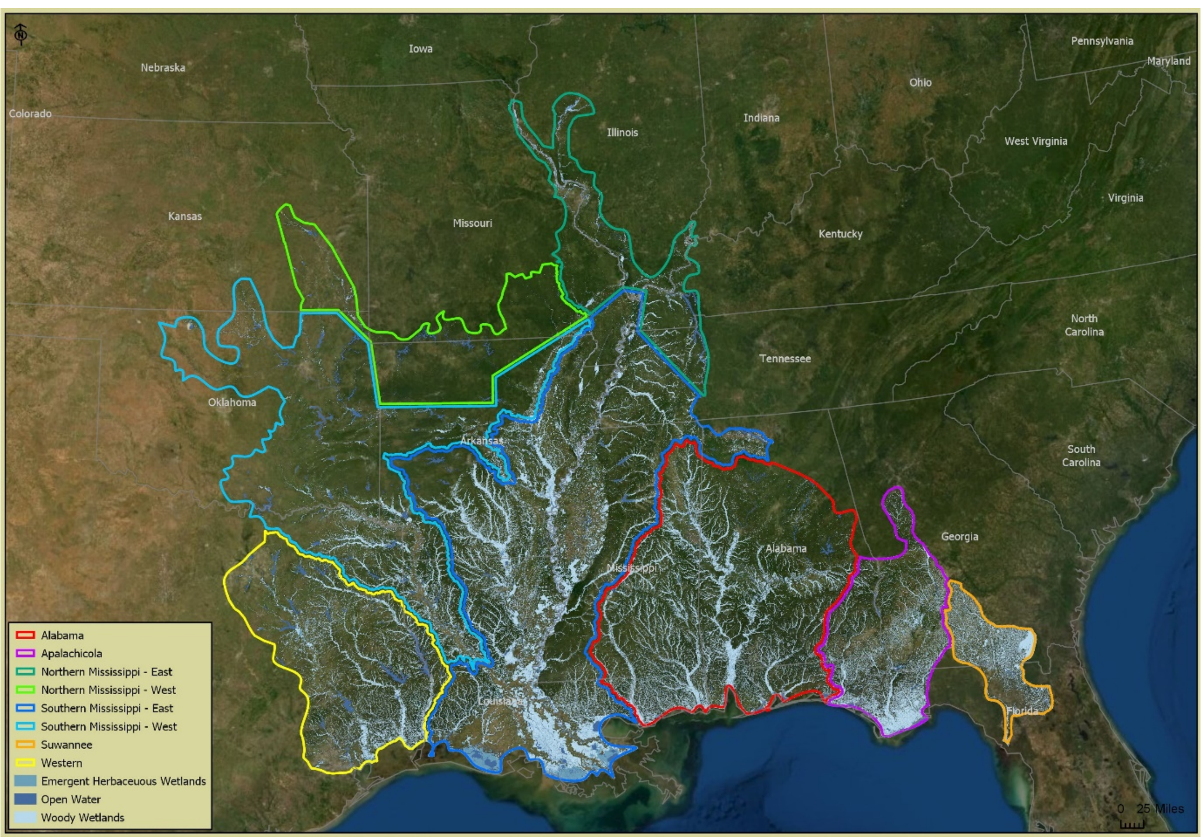


Figure A1. Suitable alligator snapping turtle habitat within the range of the species.

Table A1. Acres of suitable alligator snapping turtle suitable habitat within the range of the species.

Analysis Unit / Acres	Emergent Herbaceuous Wetlands	Open Water	Woody Wetlands	Total Acres	Analysis Unit Acres	Percentage of Unit is Suitable Habitat
Analysis Unit 1 Western	246,468	895,656	2,808,280	3,950,405	23,992,931	16.46%
Analysis Unit 2 Southern Mississippi - West	208,468	1,228,429	2,194,695	3,631,593	43,222,816	8.40%
Analysis Unit 3 Southern Mississippi - East	1,745,297	2,235,897	10,647,081	14,628,274	61,306,892	23.86%
Analysis Unit 4 Alabama	419,289	801,026	6,330,556	7,550,871	41,285,934	18.29%
Analysis Unit 5 Apalachicola	136,807	395,198	3,053,156	3,585,161	14,980,602	23.93%
Analysis Unit 6 Suwannee	62,981	64,890	1,620,961	1,748,832	5,934,668	29.47%
Analysis Unit 7 Northern Mississippi - West	12,722	264,274	73,857	350,854	16,268,981	2.16%
Analysis Unit 8 Northern Mississippi - East	105,292	528,647	642,874	1,276,813	14,376,441	8.88%
Total:	**2,937,325**	**6,414,018**	**27,371,460**	**36,722,803**	**221,369,267**	**16.59%**

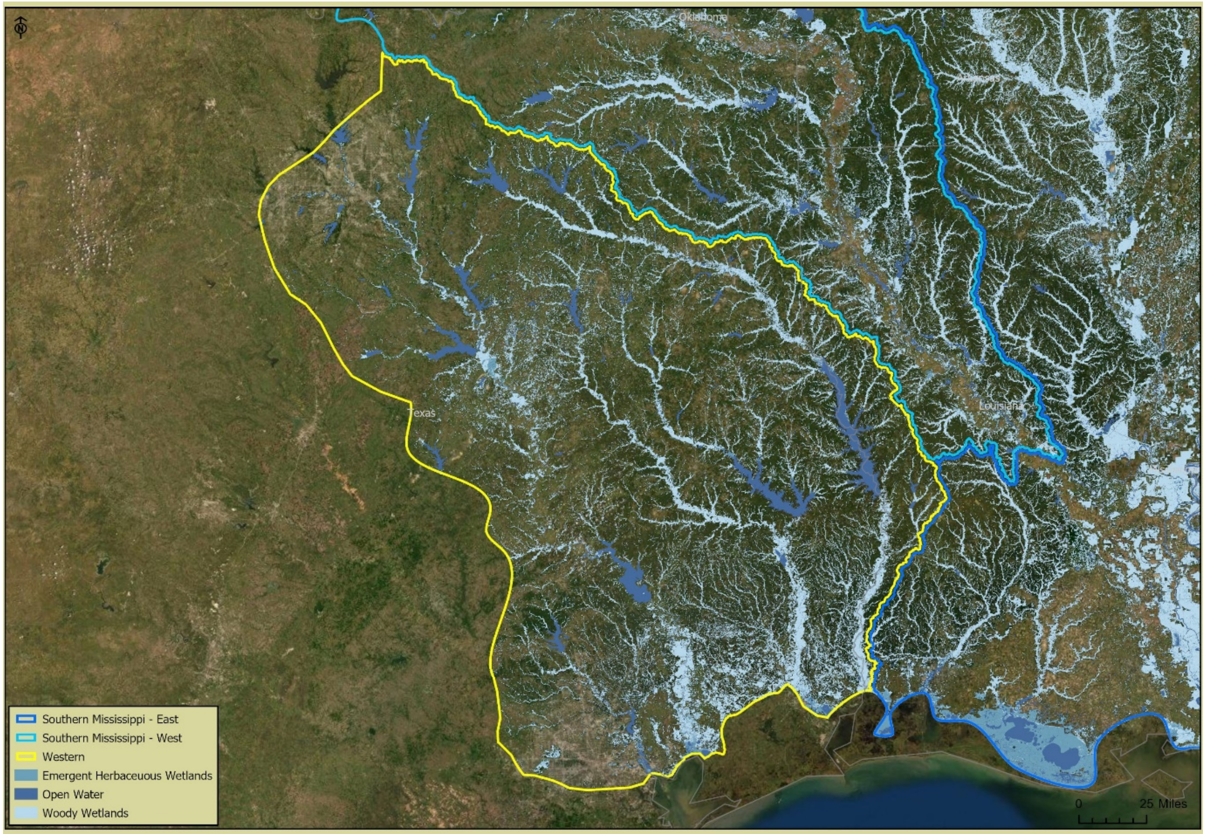


Figure A2. Suitable alligator snapping turtle habitat within the Western Unit.

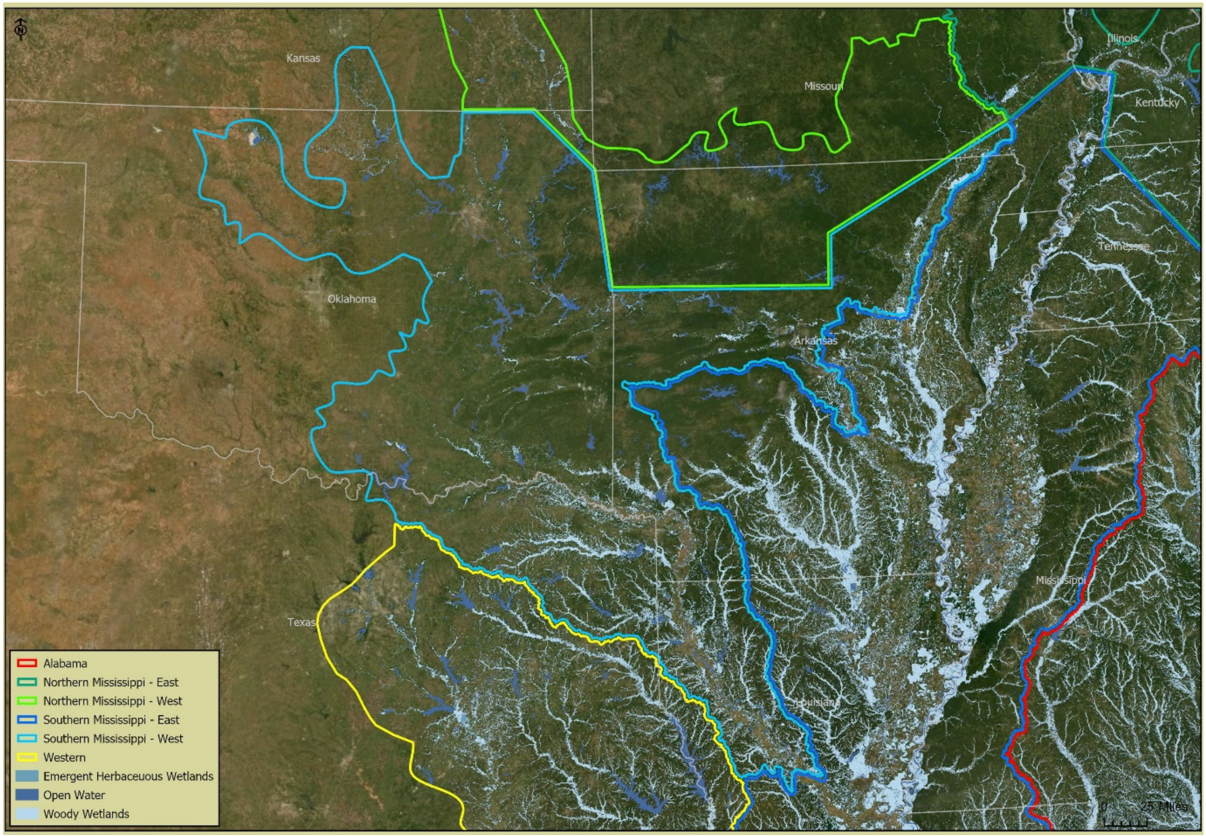


Figure A3. Suitable alligator snapping turtle habitat within the Southern Mississippi – West Unit.

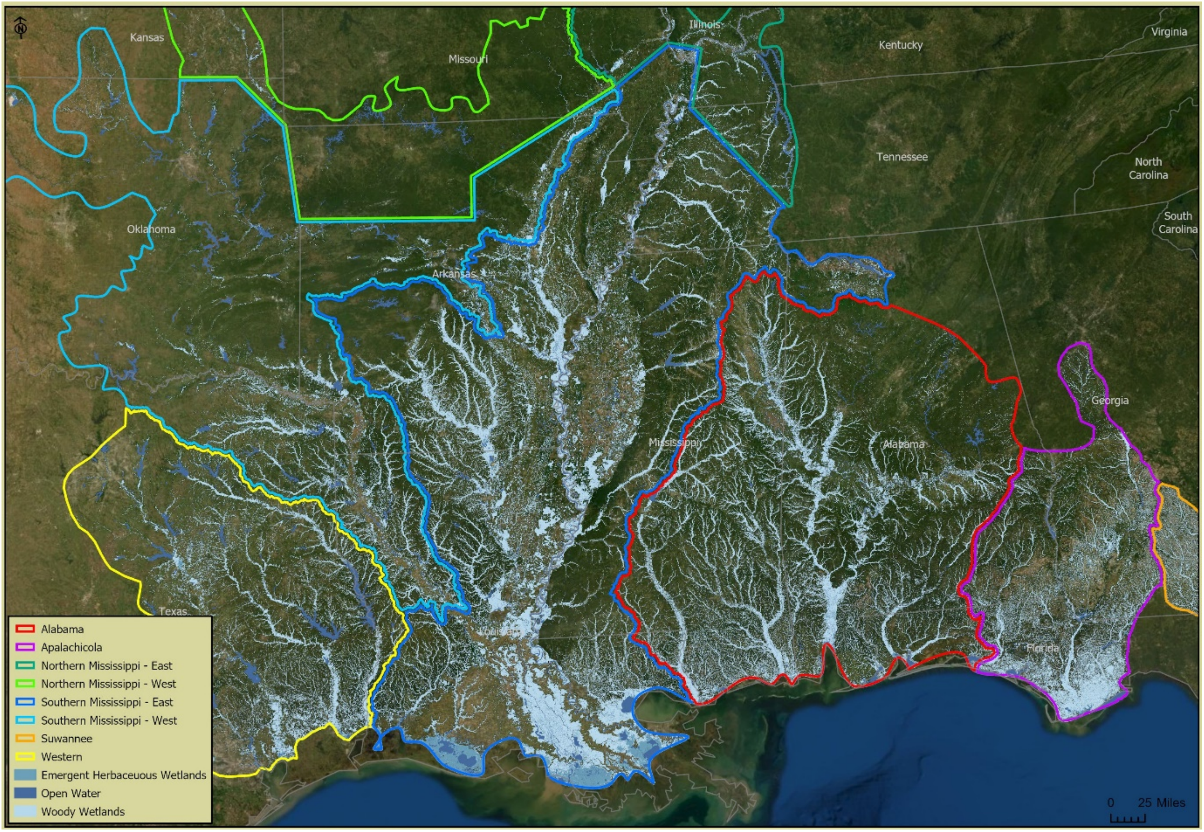


Figure A4. Suitable alligator snapping turtle habitat within the Southern Mississippi – East Unit.

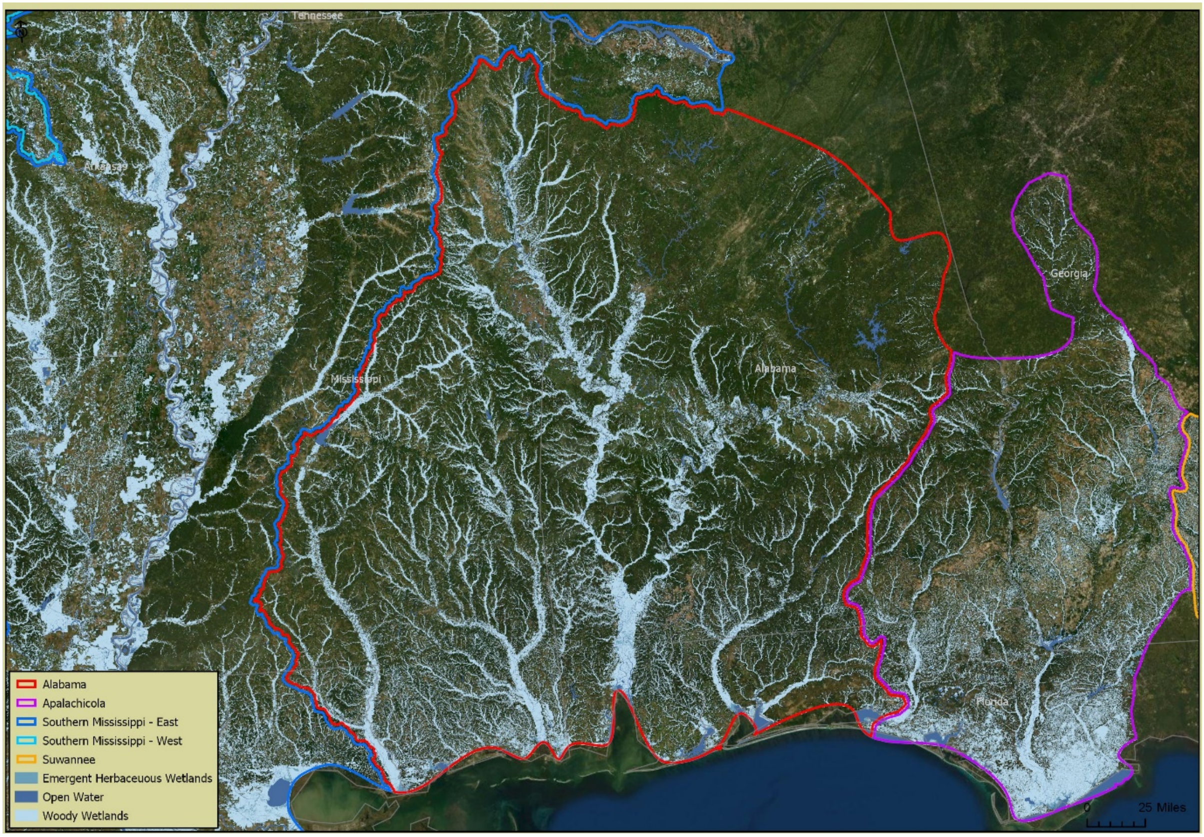


Figure A5. Suitable alligator snapping turtle habitat within the Alabama Unit.

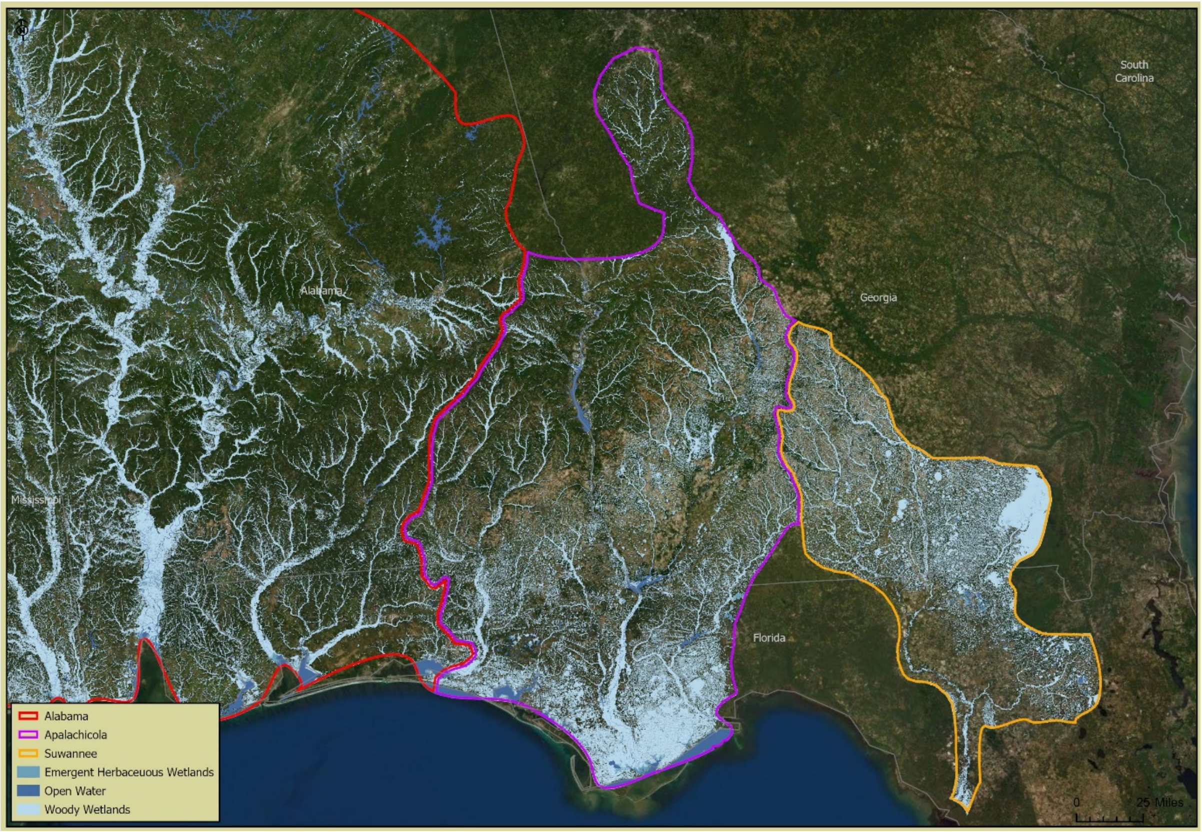


Figure A6. Suitable alligator snapping turtle habitat within the Apalachicola Unit.

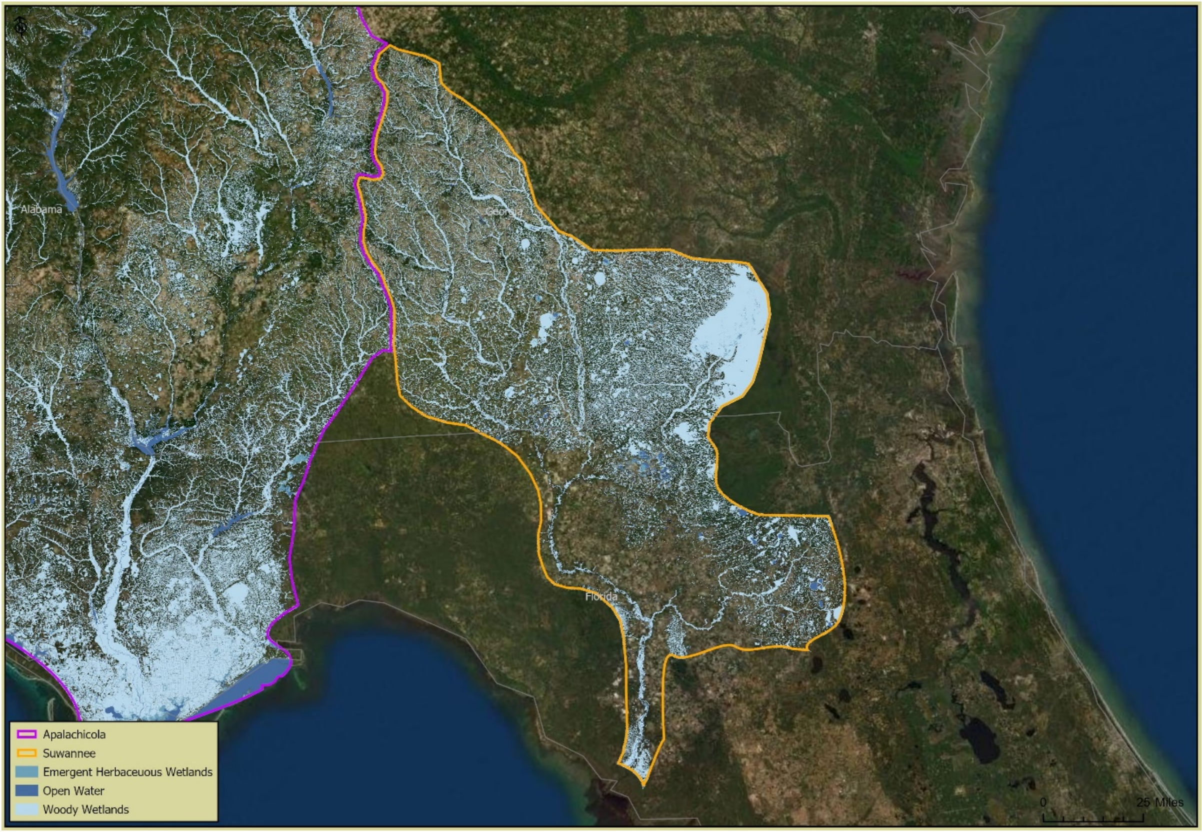


Figure A7. Suitable alligator snapping turtle habitat within the Suwannee Unit.

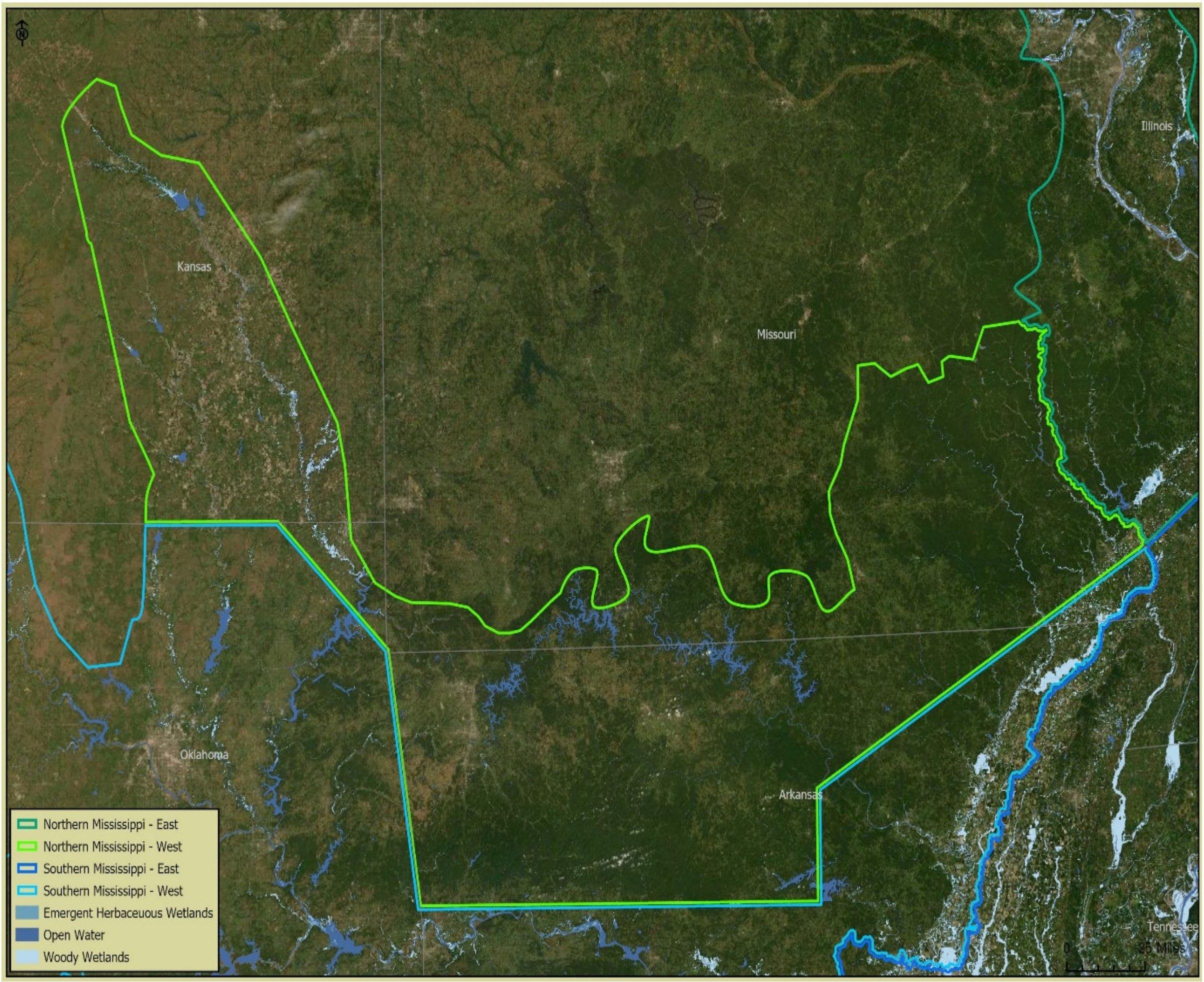


Figure A8. Suitable alligator snapping turtle habitat within the Northern Mississippi – West Unit.

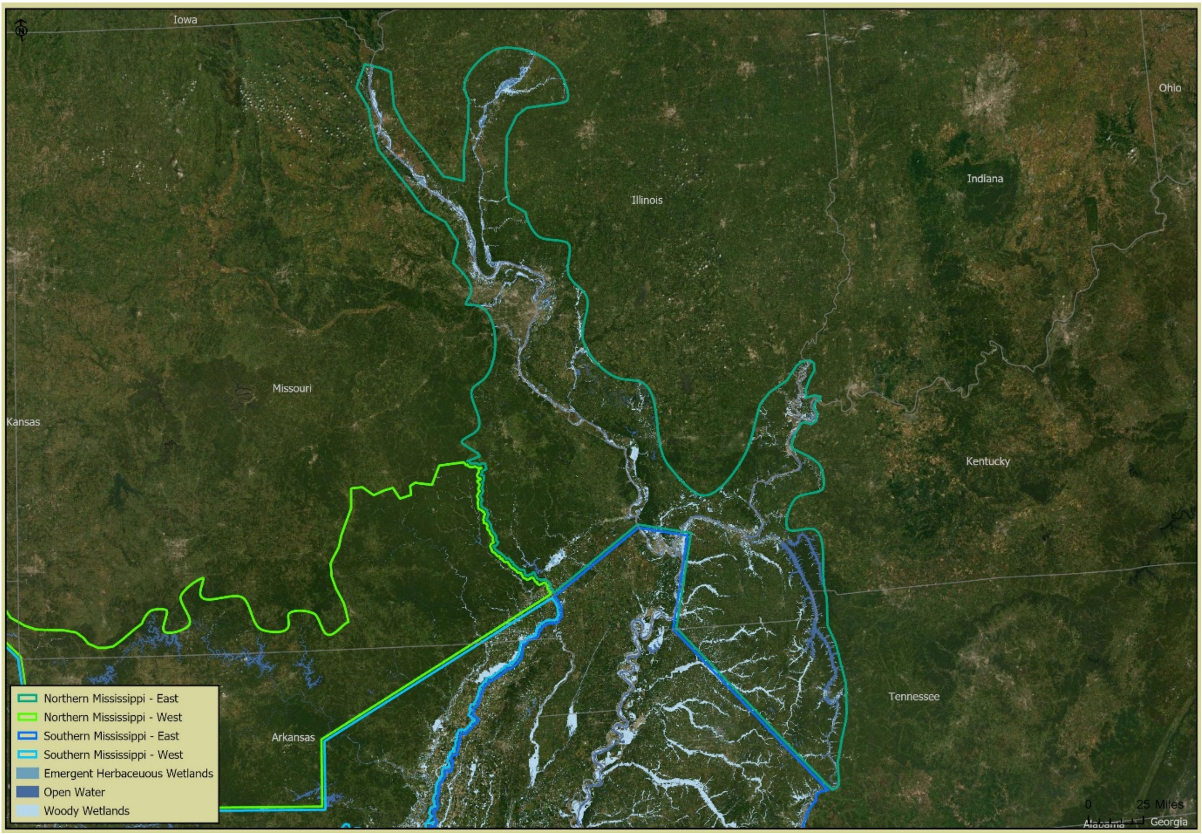


Figure A9. Suitable alligator snapping turtle habitat within the Northern Mississippi – East Unit.

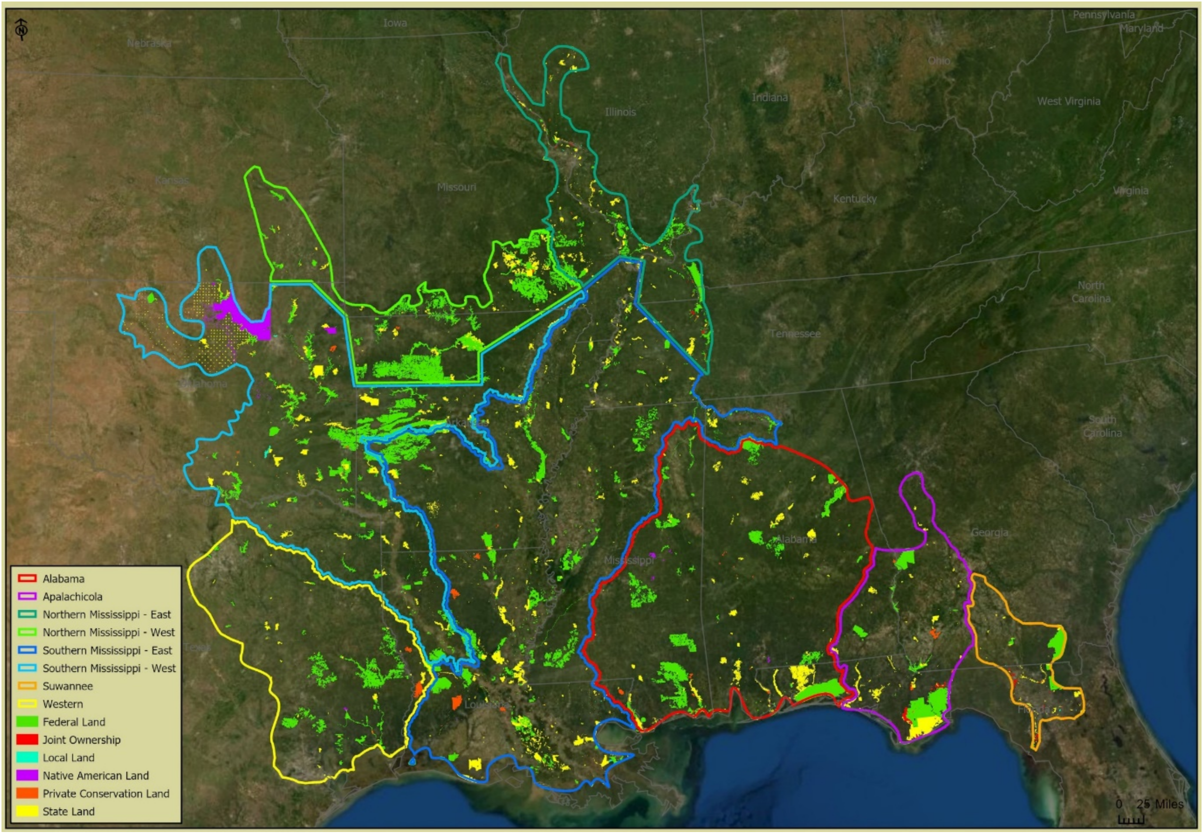


Figure A10. Lands in conservation within the range of the alligator snapping turtle.

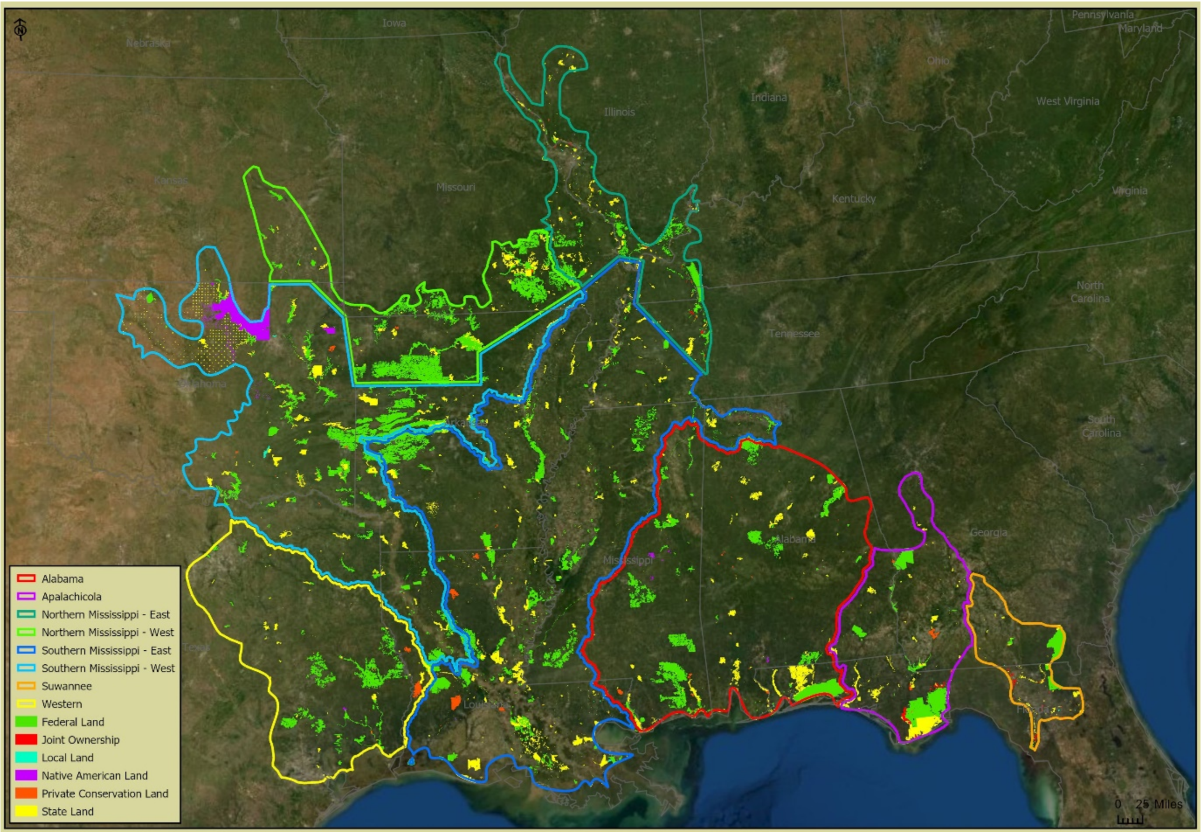


Figure A11. Suitable alligator snapping turtle habitat within conservation lands.

Table A2. Acres of suitable alligator snapping turtle habitat within conservation areas.

All Analysis Units	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Total Suitable Habitat of All Analysis Units	1,284,239	855,336	26,019	102,795	1,139,674	23,556	173	54,524	112,768	5,699	81,705	2,237,757	15,429	72,944	**6,012,620**
All Analysis Units Lands in Conservation Acres	1,646,065	8,020,683	55,198	343,551	1,719,610	85,656	206	927,482	1,118,451	13,442	136,082	4,847,231	36,677	354,524	**19,304,858**
Percentage Suitable Habitat of All Conservation Lands	**78.02%**	**10.66%**	**47.14%**	**29.92%**	**66.28%**	**27.50%**	**83.77%**	**5.88%**	**10.08%**	**42.40%**	**60.04%**	**46.17%**	**42.07%**	**20.58%**	**31.15%**

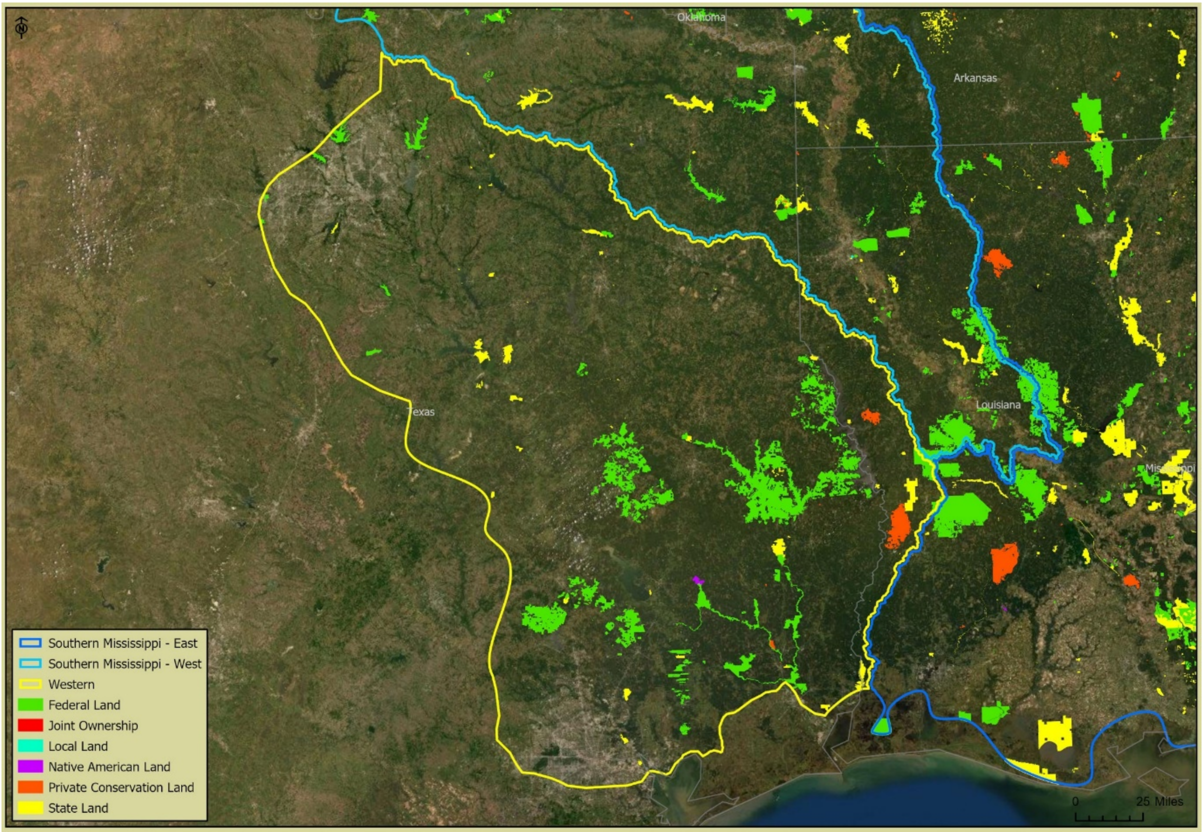


Figure A12. Lands in conservation within the range of the alligator snapping turtle within the Western Unit.

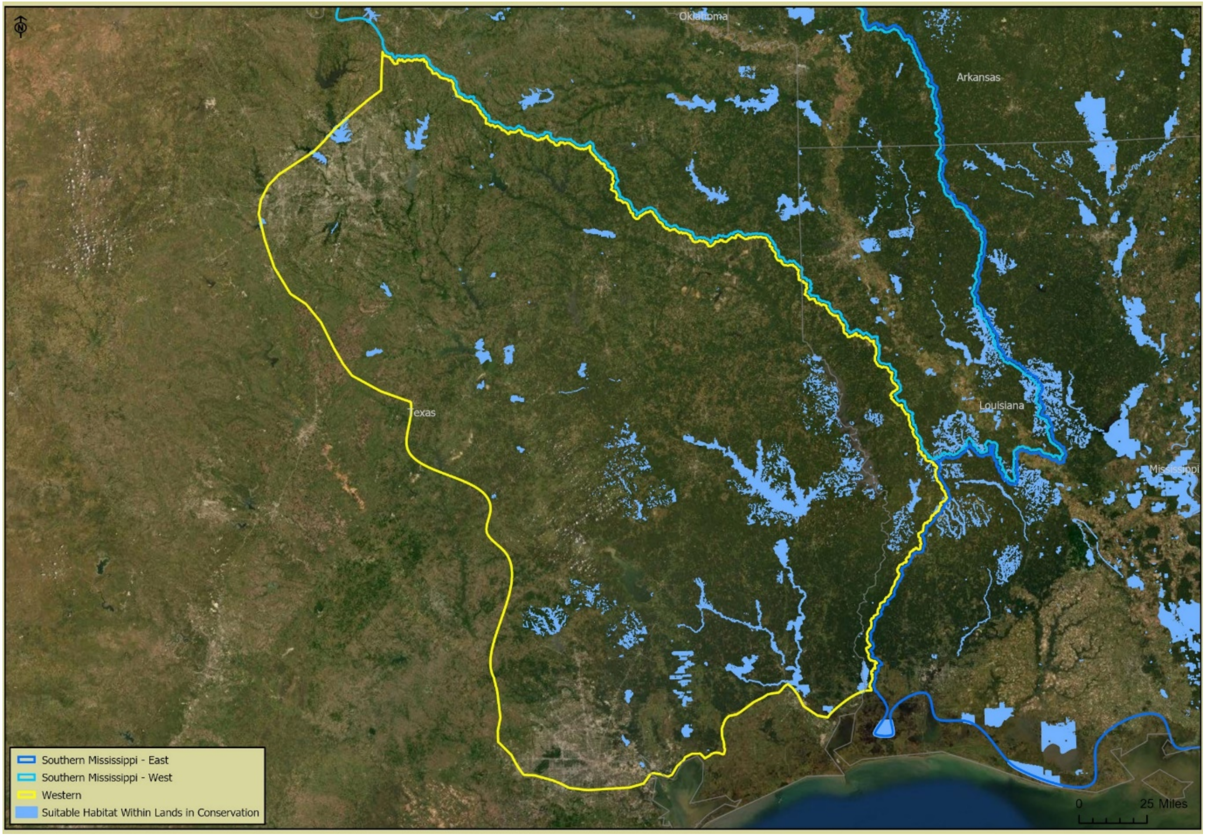


Figure A13. Suitable alligator snapping turtle habitat within conservation lands within the Western Unit.

Table A3. Acres of suitable alligator snapping turtle habitat within conservation areas – Western Unit.

Analysis Unit 1 Western	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	1,678	2,435	0	2,045	7,449	0	0	0	10	0	0	6,162	0	54	**19,833**
Open Water Acres	596	15,857	0	4,026	140,465	0	0	7	32	0	0	15,390	0	101	**176,475**
Woody Wetlands Acres	30,593	61,814	0	62,565	2,536	0	0	1,372	1,801	0	0	61,992	0	5,358	**228,033**
Total Suitable Habitat Acres	32,868	80,106	0	68,637	150,450	0	0	1,379	1,843	0	0	83,544	0	5,514	**424,341**
Analysis Unit in Conservation Acres	38,371	644,353	0	112,269	155,958	0	0	4,477	40,648	0	0	129,297	0	73,270	**1,198,643**
Percentage of Conservation Lands are Suitable Habitat	**85.66%**	**12.43%**	**0.00%**	**61.14%**	**96.47%**	**0.00%**	**0.00%**	**30.81%**	**4.53%**	**0.00%**	**0.00%**	**64.61%**	**0.00%**	**7.52%**	**35.40%**

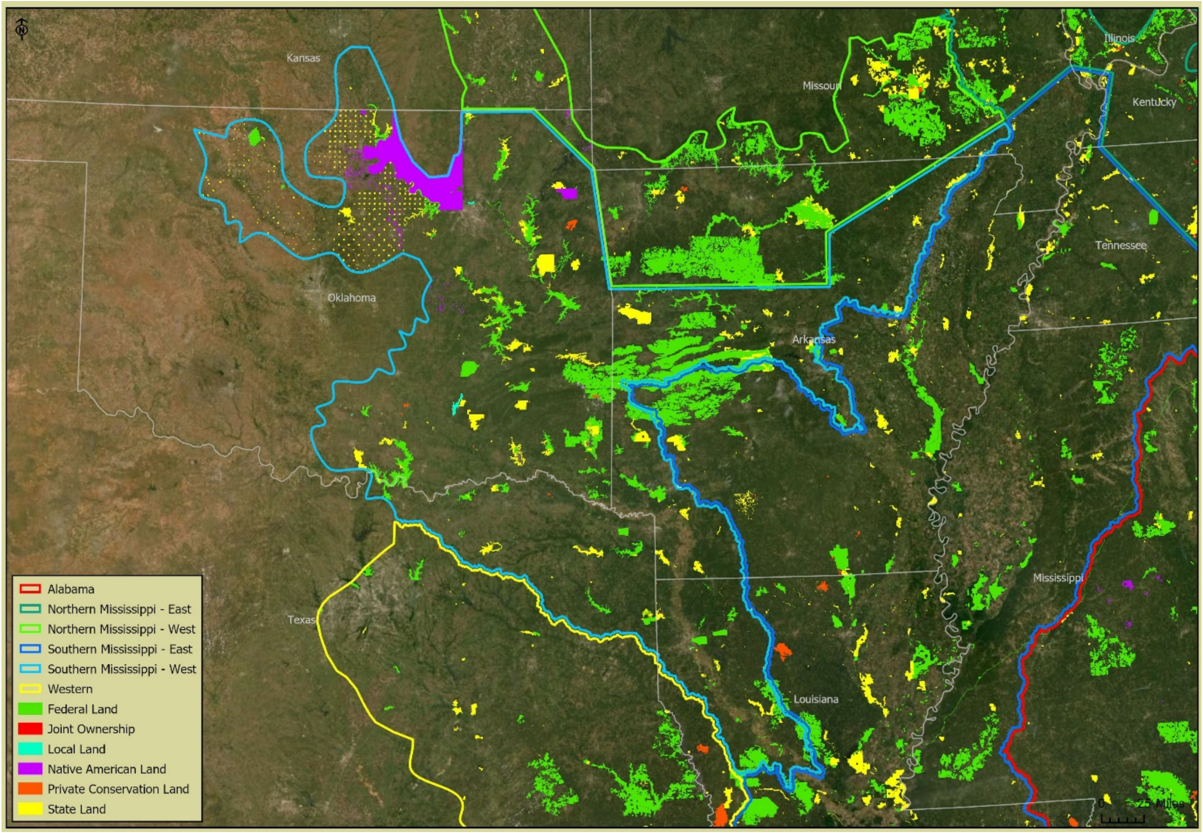


Figure A14. Lands in conservation within the range of the alligator snapping turtle in the Southern Mississippi – West Unit.

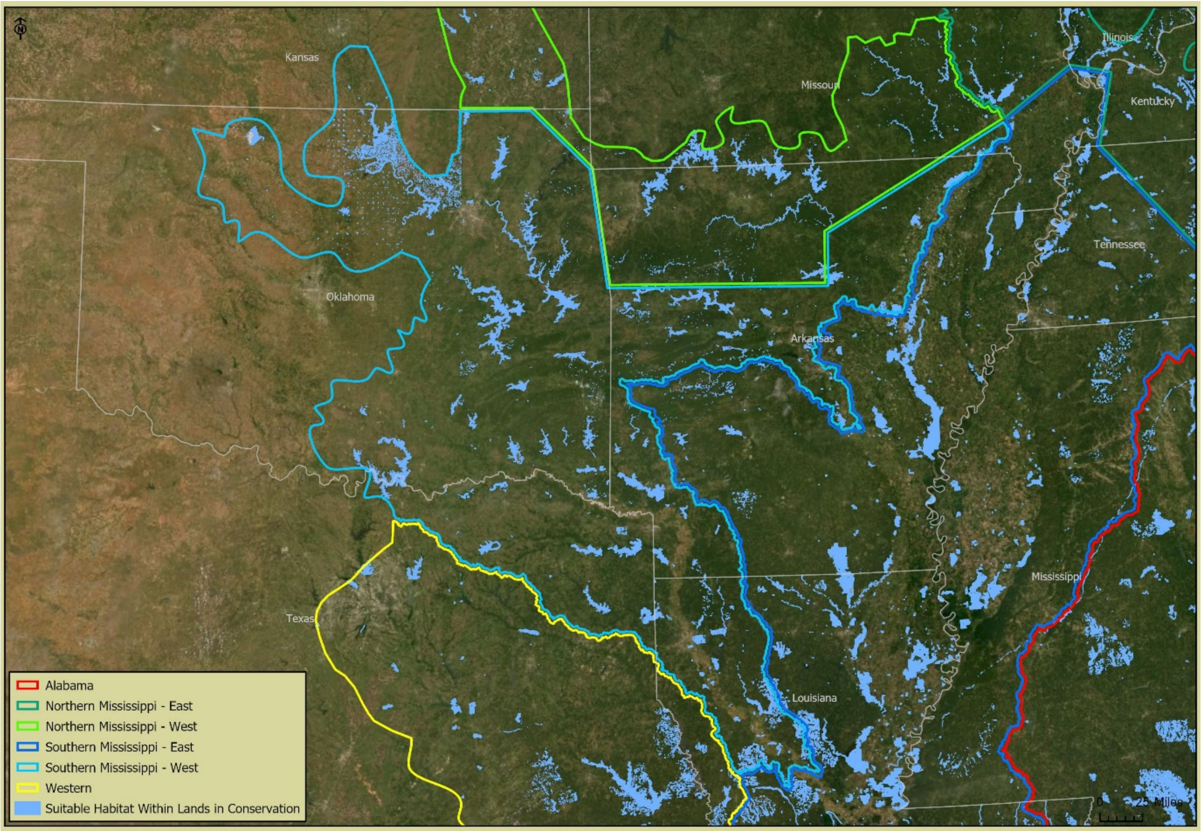


Figure A15. Suitable alligator snapping turtle habitat on conservation lands within the Southern Mississippi – West Unit.

Table A4. Acres of suitable alligator snapping turtle habitat within conservation areas Southern Mississippi – West Unit.

Analysis Unit 2 Southern Mississippi - West	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	11,087	1,864	1,139	17	20,949	0	0	3,608	650	0	0	16,604	114	169	**56,201**
Open Water Acres	26,897	3,435	166	2,439	455,208	0	0	32,164	1,928	0	0	66,345	6,119	503	**595,204**
Woody Wetlands Acres	52,273	48,459	2,251	14	41,668	0	0	9,454	16,651	0	0	169,466	952	2,111	**343,300**
Total Suitable Habitat Acres	90,257	53,758	3,556	2,469	517,825	0	0	45,226	19,230	0	0	252,415	7,185	2,783	**994,705**
Analysis Unit in Conservation Acres	179,486	1,525,242	5,232	10,157	810,026	0	0	885,913	155,631	0	0	1,040,411	19,384	27,173	**4,658,655**
Percentage of Conservation Lands are Suitable Habitat	**50.29%**	**3.52%**	**67.96%**	**24.31%**	**63.93%**	**0.00%**	**0.00%**	**5.11%**	**12.36%**	**0.00%**	**0.00%**	**24.26%**	**37.07%**	**10.24%**	**21.35%**

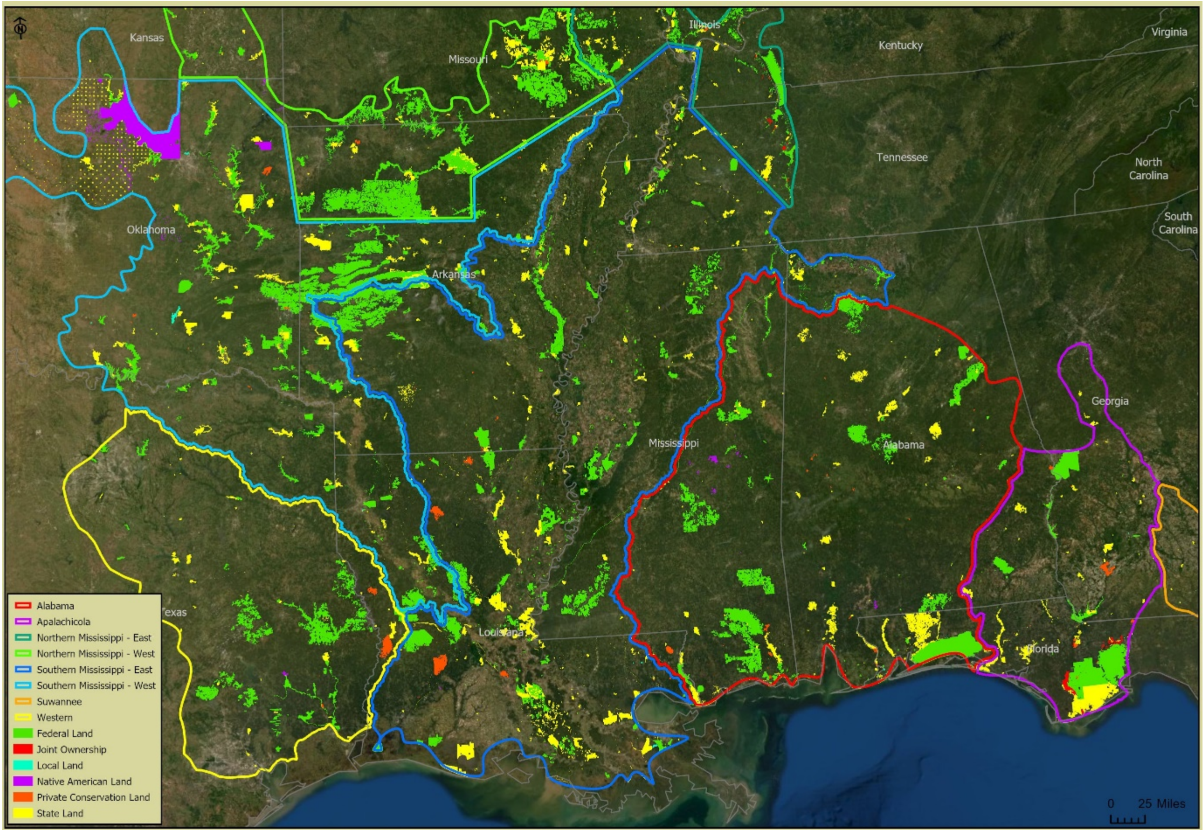


Figure A16. Lands in conservation within the range of the alligator snapping turtle in the Southern Mississippi – East Unit.

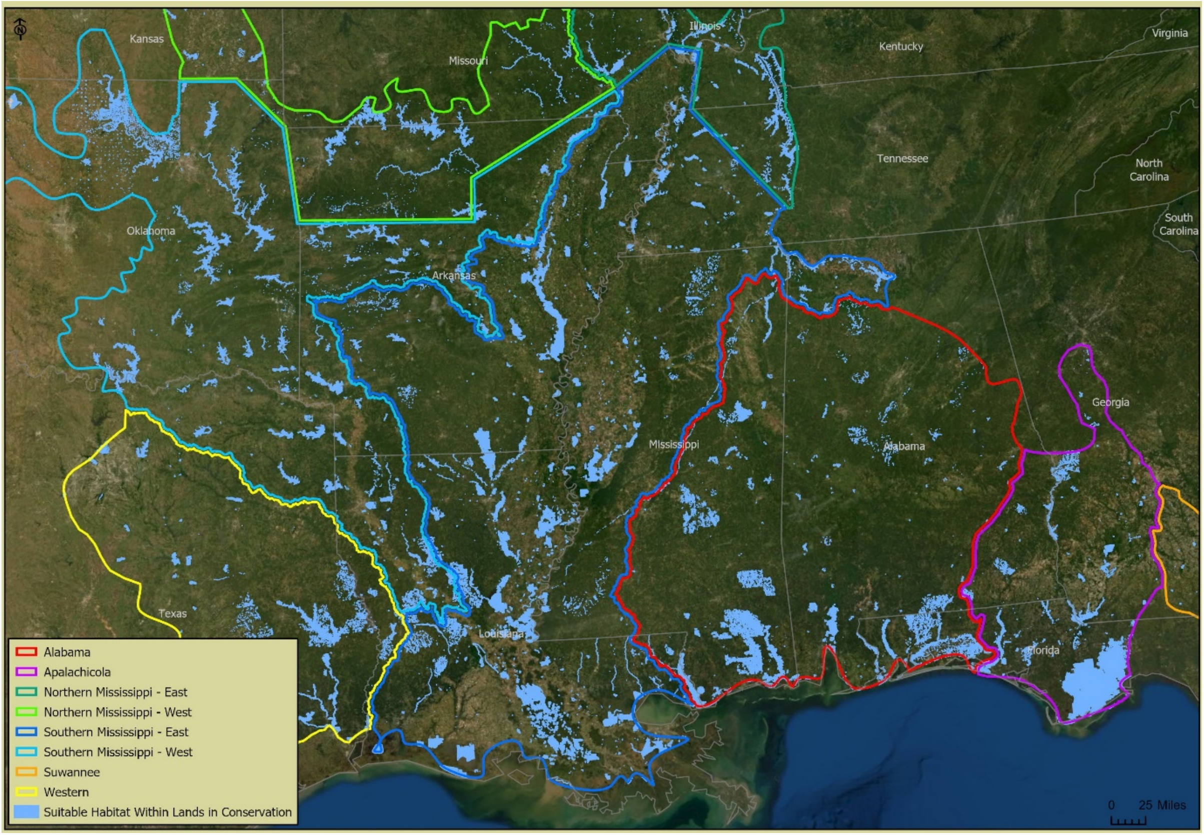


Figure A17. Suitable alligator snapping turtle habitat on conservation lands within the Southern Mississippi – East Unit.

Table A5. Acres of suitable alligator snapping turtle habitat within conservation areas Southern Mississippi – East Unit.

Analysis Unit 3 Southern Mississippi – East	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	92,192	1,033	3,427	13,908	15,631	663	0	3	756	0	4	167,900	3,004	269	**298,788**
Open Water Acres	63,210	5,598	1,382	1,503	27,420	3,195	0	22	345	0	64	97,617	587	39	**200,982**
Woody Wetlands Acres	575,662	122,620	17,655	6,216	125,504	7,534	0	311	8,138	0	362	765,522	58	32,616	**1,662,197**
Total Suitable Habitat Acres	731,064	129,251	22,464	21,626	168,555	11,392	0	336	9,239	0	430	1,031,039	3,649	32,924	**2,161,968**
Analysis Unit in Conservation Acres	897,109	1,474,414	49,966	27,905	209,757	49,028	0	1,159	157,134	0	833	1,401,089	4,659	130,066	**4,403,119**
Percentage of Conservation Lands are Suitable Habitat	**81.49%**	**8.77%**	**44.96%**	**77.50%**	**80.36%**	**23.23%**	**0.00%**	**28.98%**	**5.88%**	**0.00%**	**51.62%**	**73.59%**	**78.32%**	**25.31%**	**49.10%**

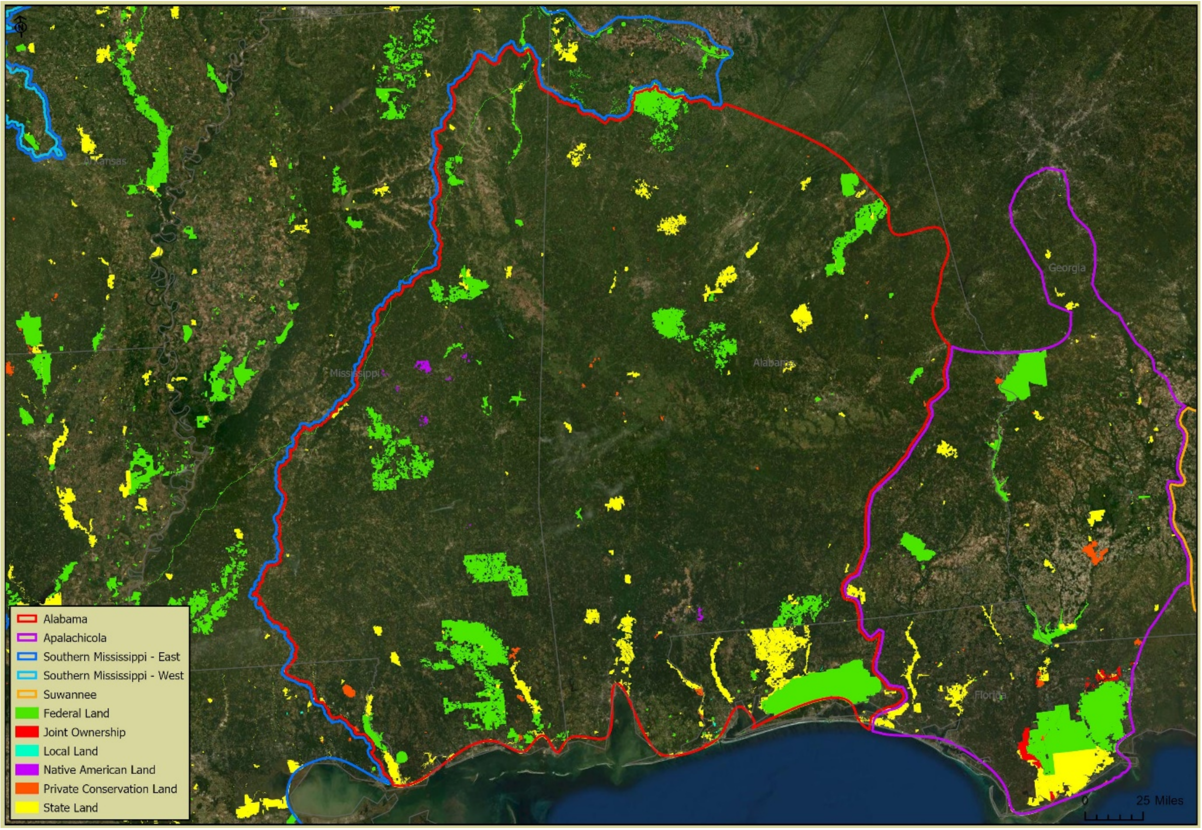


Figure A18. Lands in conservation within the range of the alligator snapping turtle in the Alabama Unit.

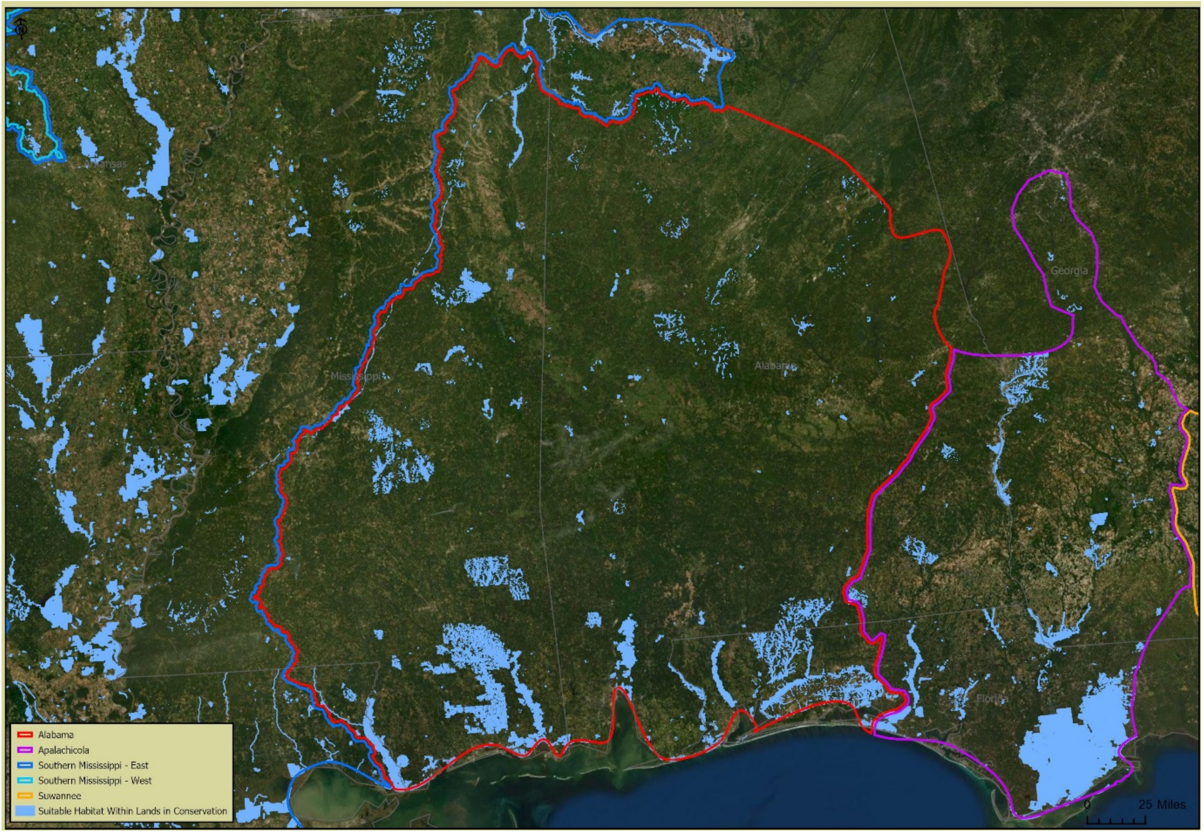


Figure A19. Suitable alligator snapping turtle habitat on conservation lands within the Alabama Unit.

Table A6. Acres of suitable alligator snapping turtle habitat within conservation areas Alabama Unit.

Analysis Unit 4 Alabama	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	4,054	755	0	618	2,719	0	0	471	1,230	170	64	44,837	85	1,009	**56,013**
Open Water Acres	2,151	3,532	0	2,736	1,639	0	0	293	1,016	265	23	18,583	19	843	**31,100**
Woody Wetlands Acres	80,363	188,523	0	218	41,004	0	0	6,410	53,401	5,264	180	262,332	1,688	16,777	**656,160**
Total Suitable Habitat Acres	86,568	192,810	0	3,572	45,361	0	0	7,174	55,647	5,699	267	325,752	1,792	18,629	**743,272**
Analysis Unit in Conservation Acres	121,412	1,350,433	0	6,898	59,728	0	0	25,912	493,449	13,442	650	808,607	2,641	49,143	**2,932,315**
Percentage of Conservation Lands are Suitable Habitat	**71.30%**	**14.28%**	**0.00%**	**51.78%**	**75.95%**	**0.00%**	**0.00%**	**27.68%**	**11.28%**	**42.40%**	**41.04%**	**40.29%**	**67.87%**	**37.91%**	**25.35%**

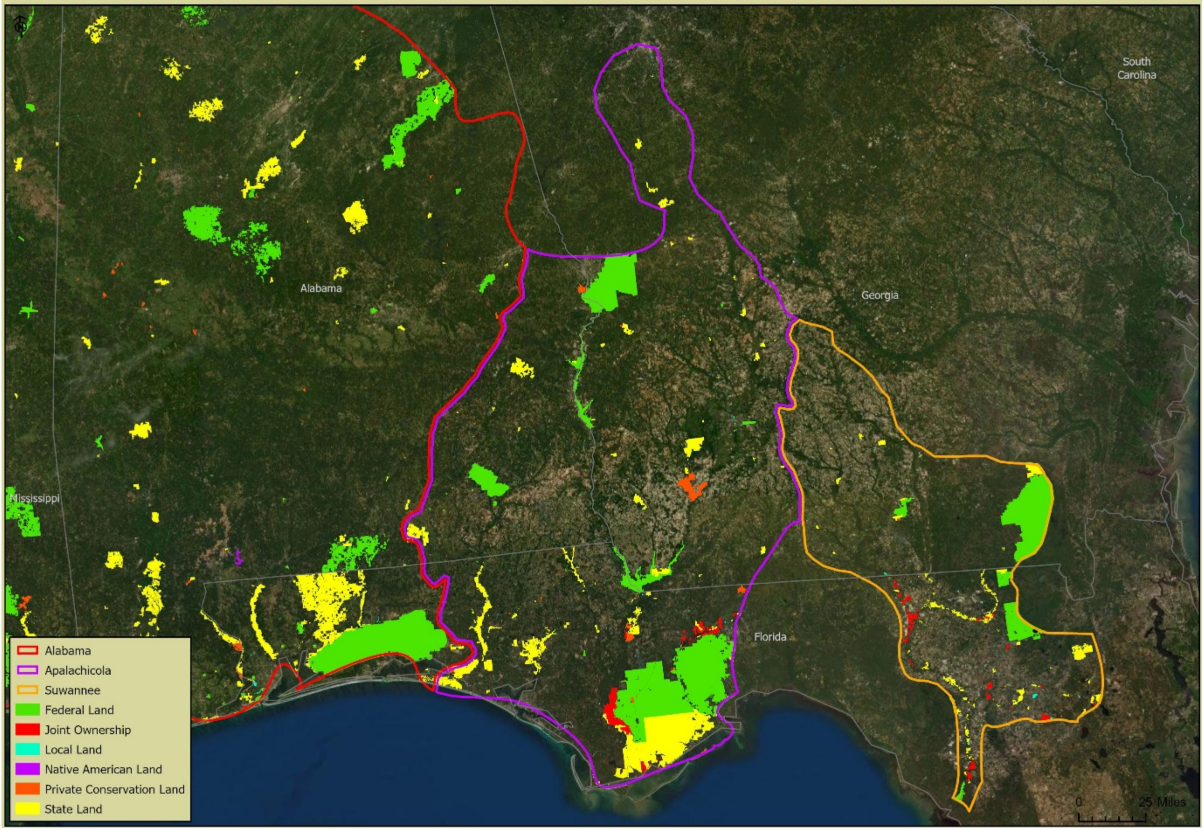


Figure A20. Lands in conservation within the range of the alligator snapping turtle in the Apalachicola Unit.

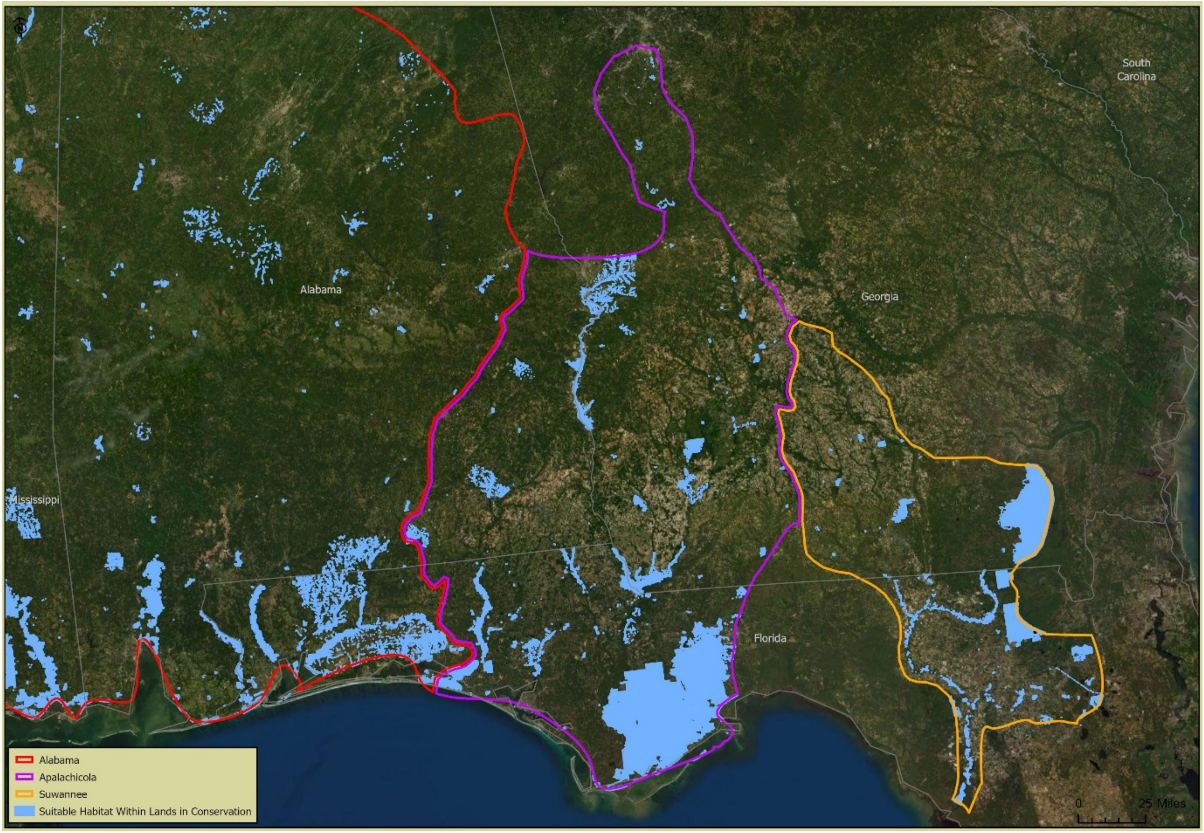


Figure A21. Suitable alligator snapping turtle habitat on conservation lands within the Apalachicola Unit.

Table A7. Acres of suitable alligator snapping turtle habitat within conservation areas in the Apalachicola Unit.

Analysis Unit 5 Apalachicola	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	2,432	3,092	0	0	1,425	0	33	0	452	0	2,915	18,243	22	1,545	**30,159**
Open Water Acres	4,900	1,331	0	0	42,788	0	9	0	2,750	0	4,047	8,115	49	815	**64,805**
Woody Wetlands Acres	7,689	330,249	0	35	17,723	0	130	0	20,294	0	44,193	346,093	757	5,826	**772,989**
Total Suitable Habitat Acres	15,021	334,672	0	35	61,937	0	173	0	23,496	0	51,155	372,451	827	8,186	**867,953**
Analysis Unit in Conservation Acres	21,748	569,605	0	593	83,026	0	206	0	247,319	0	64,386	558,043	1,927	49,647	**1,596,500**
Percentage of Conservation Lands are Suitable Habitat	**69.07%**	**58.76%**	**0.00%**	**5.91%**	**74.60%**	**0.00%**	**83.77%**	**0.00%**	**9.50%**	**0.00%**	**79.45%**	**66.74%**	**42.92%**	**16.49%**	**54.37%**

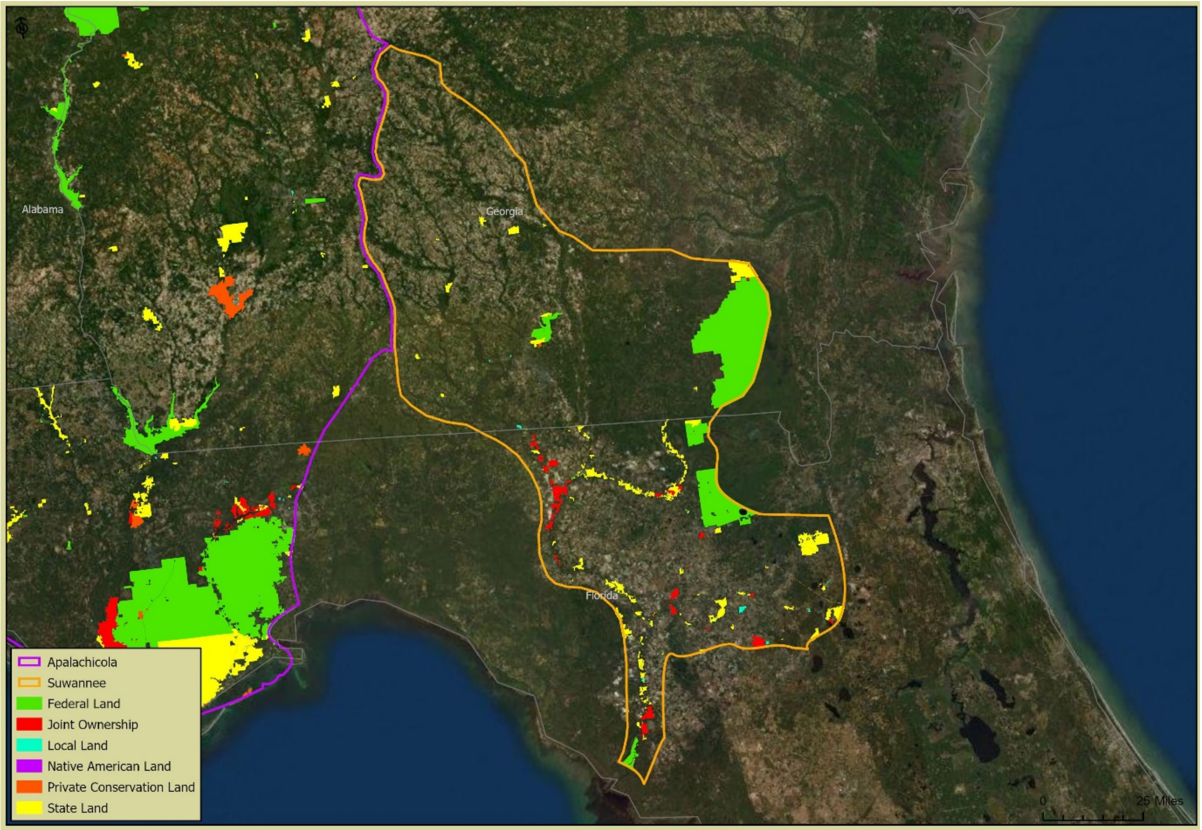


Figure A22. Lands in conservation within the range of the alligator snapping turtle in the Suwannee Unit.

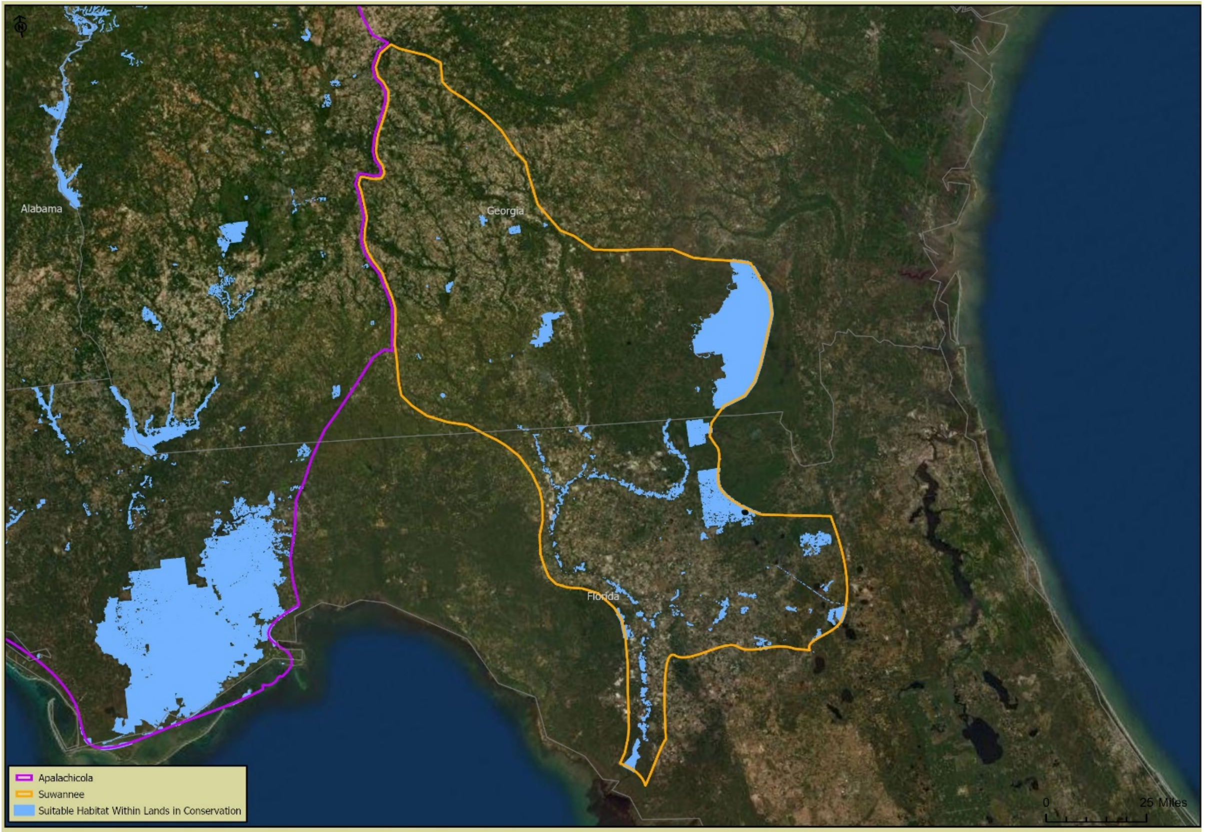


Figure A23. Suitable alligator snapping turtle habitat on conservation lands within the Suwannee Unit.

Table A8. Acres of suitable alligator snapping turtle habitat within conservation areas in the Suwannee Unit.

Analysis Unit 6 Suwannee	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	5,861	127	0	0	158	0	0	0	54	0	234	1,029	7	3	**7,473**
Open Water Acres	933	144	0	0	13	0	0	0	29	0	485	857	13	2	**2,477**
Woody Wetlands Acres	230,271	32,650	0	0	3,751	0	0	0	2,295	0	7,708	52,283	1,118	225	**330,301**
Total Suitable Habitat Acres	237,065	32,921	0	0	3,922	0	0	0	2,379	0	8,427	54,169	1,138	230	**340,251**
Analysis Unit in Conservation Acres	248,181	86,470	0	0	5,596	0	0	0	4,731	0	38,533	116,352	3,270	571	**503,704**
Percentage of Conservation Lands are Suitable Habitat	**95.52%**	**38.07%**	**0.00%**	**0.00%**	**70.09%**	**0.00%**	**0.00%**	**0.00%**	**50.27%**	**0.00%**	**21.87%**	**46.56%**	**34.79%**	**40.32%**	**67.55%**

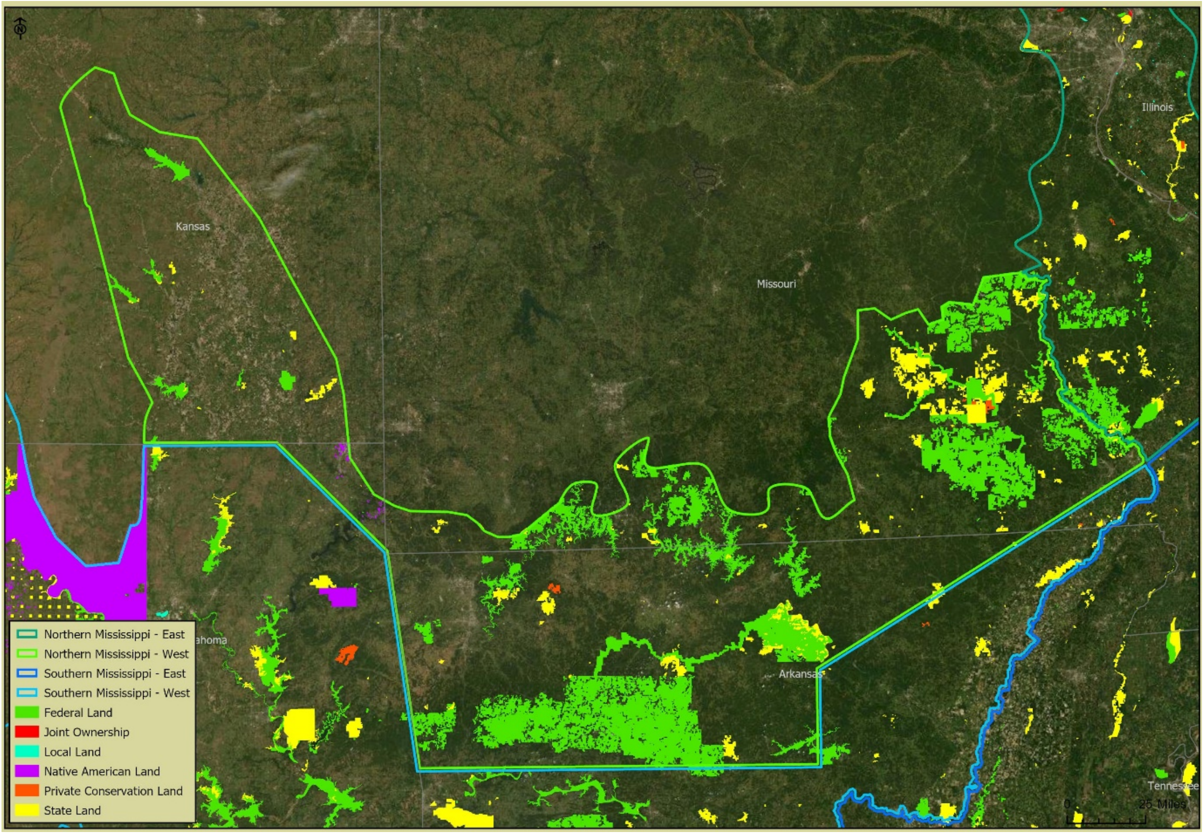


Figure A24. Lands in conservation within the range of the alligator snapping turtle in the Northern Mississippi – West Unit.

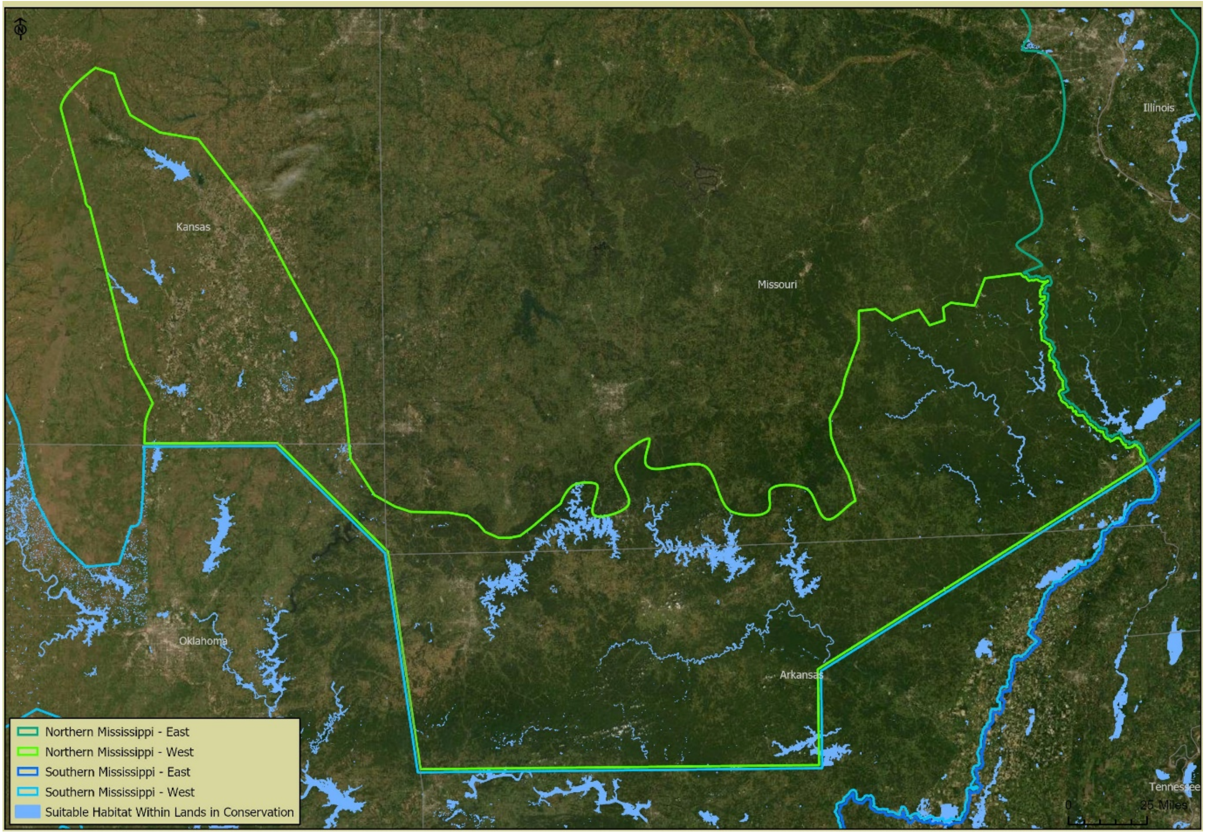


Figure A25. Suitable alligator snapping turtle habitat on conservation lands within the Northern Mississippi – West Unit.

Table A9. Acres of suitable alligator snapping turtle habitat within conservation areas in the Northern Mississippi – West Unit.

Analysis Unit 7 Northern Mississippi – West	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	2,306	96	0	237	2,076	0	0	61	17	0	0	584	0	11	**5,388**
Open Water Acres	3,030	1,411	0	4,038	171,966	0	0	129	139	0	4	4,563	0	43	**185,321**
Woody Wetlands Acres	3,213	1,438	0	2,151	3,815	0	0	218	23	0	0	5,945	0	33	**16,837**
Total Suitable Habitat Acres	8,549	2,945	0	6,426	177,857	0	0	408	178	0	4	11,092	0	87	**207,546**
Analysis Unit in Conservation Acres	18,838	1,817,394	0	185,078	349,307	0	0	10,021	13,915	0	25	411,923	0	12,400	**2,818,901**
Percentage of Conservation Lands are Suitable Habitat	**45.38%**	**0.16%**	**0.00%**	**3.47%**	**50.92%**	**0.00%**	**0.00%**	**4.07%**	**1.28%**	**0.00%**	**16.16%**	**2.69%**	**0.00%**	**0.70%**	**7.36%**

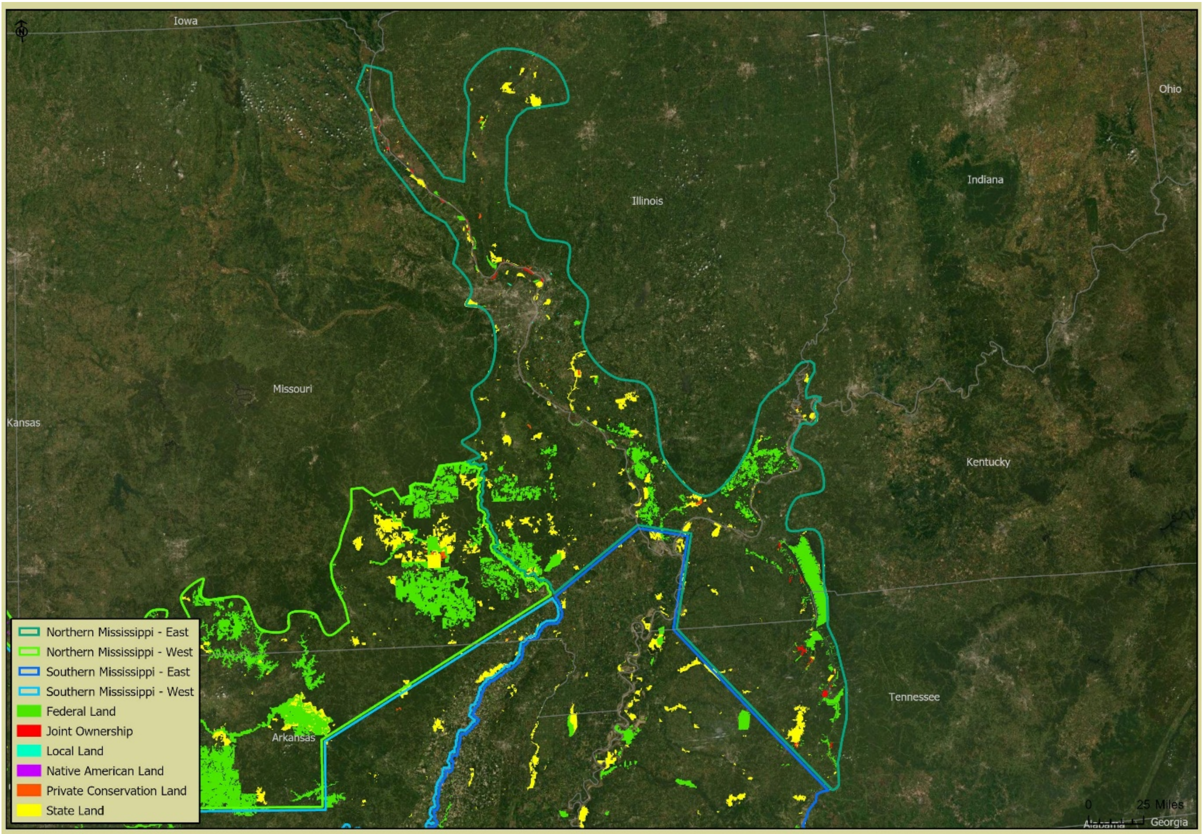


Figure A26. Lands in conservation within the range of the alligator snapping turtle in the Northern Mississippi – East Unit.

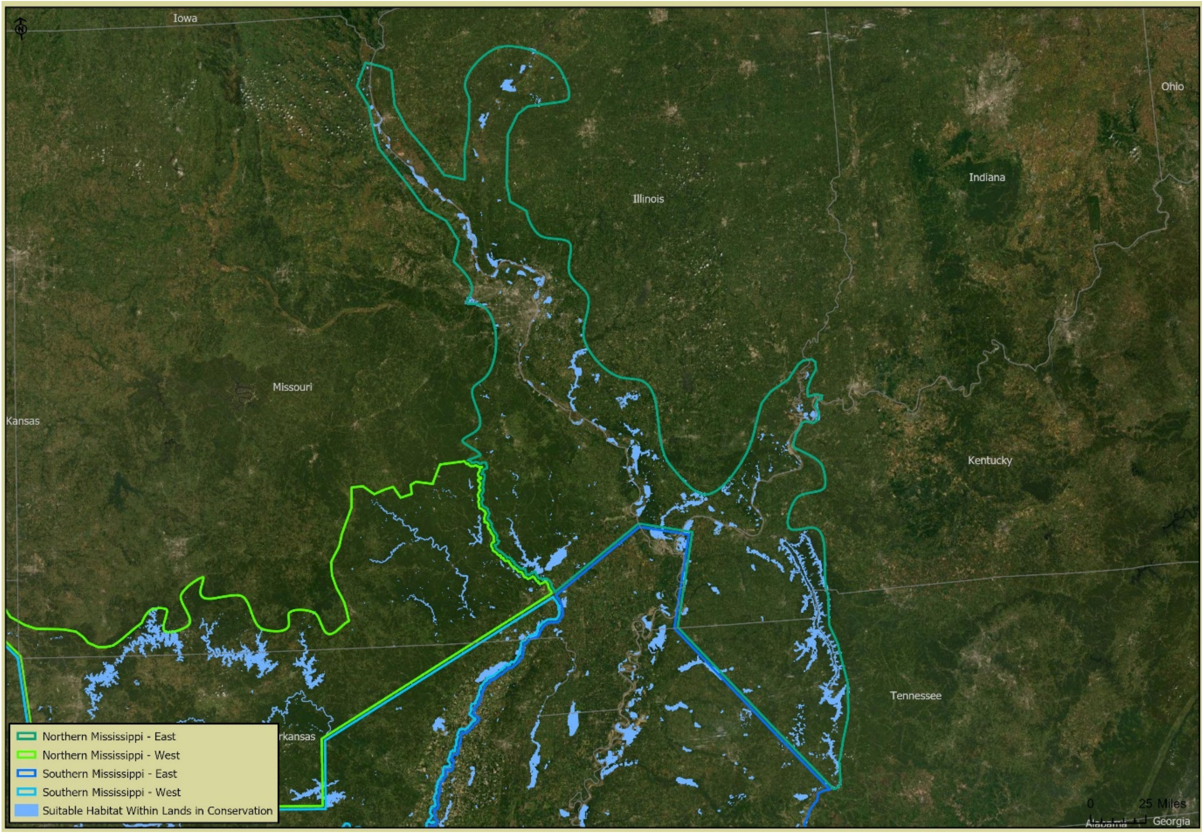


Figure A27. Suitable alligator snapping turtle habitat on conservation lands within the Northern Mississippi – East Unit.

Table A10. Acres of suitable alligator snapping turtle habitat within conservation areas in the Northern Mississippi – East Unit.

Analysis Unit 8 Northern Mississippi – East	FWS NWR	USDA USFS	USDA NRCS	NPS	USACE	TVA	BLM	Native American Lands	Military Lands	NASA	Joint Ownership	State	Local	Private	Total Acres
Emergent Herbaceous Wetlands Acres	13,158	2,318	0	2	832	455	0	0	8	0	3,506	16,504	43	649	**37,475**
Open Water Acres	28,229	6,087	0	16	8,633	1,718	0	0	295	0	1,787	26,740	340	2,480	**76,324**
Woody Wetlands Acres	41,460	20,469	0	14	4,301	9,992	0	0	454	0	16,130	64,049	455	1,462	**158,786**
Total Suitable Habitat Acres	82,847	28,874	0	31	13,766	12,165	0	0	756	0	21,422	107,294	838	4,591	**272,584**
Analysis Unit in Conservation Acres	120,920	552,772	0	651	46,212	36,628	0	0	5,624	0	31,655	381,509	4,796	12,254	**1,193,021**
Percentage of Conservation Lands are Suitable Habitat	**68.51%**	**5.22%**	**0.00%**	**4.73%**	**29.79%**	**33.21%**	**0.00%**	**0.00%**	**13.45%**	**0.00%**	**67.67%**	**28.12%**	**17.47%**	**37.47%**	**22.85%**

APPENDIX B- Alligator Snapping Turtle Harvest Prohibitions

Table B1. Year commercial and recreational harvest of alligator snapping turtles was prohibited by state.

State	Year Commercial Harvest Prohibited	Year Personal Harvest Prohibited	Notes
Alabama	2012	2012	
Arkansas	1994	1994	
Florida	2009	2009	
Georgia	1992	1992	
Illinois	1994	1994	
Indiana	1994	1994	
Iowa	1987	1987	Extremely rare, so not likely to have ever been harvested
Kansas	Unsure	Unsure	Listed as a species in need of conservation in 1975
Kentucky	1975	2012	
Louisiana	2004	Still allowed	Personal harvest with proper license restricted to one per day, per person, per vehicle/vessel, no size limit.
Mississippi	1991	Still allowed	Personal harvest with proper license restricted to one per year with minimum carapace length of 24 inches
Missouri	1980	1980	
Oklahoma	Never Allowed	1992	
Tennessee	1991	1991	
Texas	1993	1993	

APPENDIX C - Expert Elicitation Questionnaire

These questions have been informed by your responses to the first round of questions and the webinar many of you attended on March 19 (Link to recording, which provides explanation of why we are asking the types of questions that follow: https://tamu-cs.webex.com/tamu-cs/ldr.php?RCID=c9b7af365357aa8170c30115fd889843 *).*

Questions are divided into three sections, 1) questions about density range-wide, 2) questions about specific analysis units, and 3) questions about influencing factors range-wide. For analysis-unit-specific questions, please answer the questions for those analysis units (one or multiple) with which you have experience/expertise. If you cannot answer a particular question, please write a brief note about the particular difficulty (e.g., not applicable in my area). <u>Please record your responses in the attached excel sheet, not in this word document.</u>

For some stress factors we have adequate information from previous studies to inform demographic models for the SSA. For several factors however, either literature is lacking or the risk is variable by geographic area, so we are hoping to infer from your collective experience the likely exposure to and demographic effect of these factors on the species. (If you are aware of literature or unpublished reports that contain this information, please send them along). We recognize that these questions may not be easy to answer, but your insights informed by experience will result in a more informed analysis. Please note, even if you aren't sure of the answer, we designed each as a series of questions to capture that uncertainty, and uncertain information is more useful to us than no information at all. In addition, your answers will be combined with those of others provided for your analysis unit giving us the collective understanding of both estimates and uncertainty around them, so each answer you can provide is helpful. Thank you for your time and effort in completing these questions.

Section 1: Range-Wide Density Questions

1) Do you believe densities differ across the entire range of alligator snapping turtles (AST)? For example, are densities higher in the west, east, or central portion of the range? What about from southern areas to northern areas?

2) Do densities differ by habitat type (e.g. oxbows, lakes, streams, rivers), and how? List the habitat types you are familiar with in order from highest AST density to lowest AST density.

3) Are there any conditions (e.g., habitat, stressors [e.g., harvest]) that correlate with densities? What are the correlated factors and how do they relate to density?

Section 2: Analysis Unit-Specific Questions

If you have expertise/experience with more than one analysis unit, please copy the Excel sheet associated with these questions and answer separately for each. For example, if you are answering for 2 analysis units, you will have 2 copies of the analysis unit sheet in the Excel response document. Analysis unit maps can be found in the map document attached in the email with these questions.

4) Abundance estimates:
    a. What do you estimate is the lowest likely number of AST within this analysis unit?
    b. What do you estimate is the highest likely number of AST within this analysis unit?

    c. What do you think the most likely estimate for number of AST is within this analysis unit?
    d. How confident are you that your interval lowest to highest (a and b above) captures the actual number of AST within this analysis unit? Please enter a number between 50% and 100% (Here and for all subsequent questions of this type, if you are less than 50% confident that the actual number falls within the interval, please widen the interval).
    e. Please describe how you arrived at your estimates (e.g., estimated #/km in rivers and #/unit of area in open water).

5) Is incidental hooking of AST on trot and limb lines from recreational fishing occurring in this Analysis Unit? If yes:
    a. What do you think the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of incidental hooking in X% of the occupied area in this analysis unit).
    b. What do you think the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
    c. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
    d. How confident are you that your interval lowest to highest (a and b above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

6) Is commercial fishing occurring in this Analysis Unit? If yes:
    a. What do you think the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of commercial fishing in X% of the occupied area in this analysis unit).
    b. What do you think the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
    c. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
    d. How confident are you that your interval lowest to highest (a and b above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

7) Is legal collection or harvest of AST occurring in this Analysis Unit? If yes:
    a. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of legal collection or harvest in X% of the occupied area in this analysis unit).
    b. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
    c. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.

d. How confident are you that your interval lowest to highest (a and b above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

8) Is illegal collection or harvest (i.e., poaching) of AST occurring in this Analysis Unit? If yes:
   a. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of illegal collection in X% of the occupied area in this analysis unit).
   b. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
   c. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
   d. How confident are you that your interval lowest to highest (a and b above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

9) Is nest predation by subsidized <u>or</u> non-native nest predators (e.g., *Sus scrofa*, *Procyon lotor*, *Solenopsis invicta*) occurring in this Analysis Unit? If yes:
   a. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of subsidized non-native nest predators in XX% of the occupied area in this analysis unit).
   b. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
   c. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
   d. How confident are you that your interval lowest to highest (a and b above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

10) Are conservation measures being taken in this Analysis Unit? If yes:
   a. What types of conservation measures are occurring within the analysis unit?

For each major type of conservation measure listed above, please answer the following questions.

   b. Have any of these measures been shown to affect demographic rates of the species? If so, how?
   c. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area. If multiple conservation actions are listed, please provide a separate estimate of spatial extent for each.
   d. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area. If multiple conservation actions are listed, please provide a separate estimate of spatial extent for each.

e. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area. If multiple conservation actions are listed, please provide a separate estimate of spatial extent for each.
f. How confident are you that your interval lowest to highest (c and d above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%. If multiple conservation actions are listed, please provide a separate estimate of spatial extent for each.

11) Are any mechanisms (e.g., dredging, sedimentation, etc.) contributing to habitat loss in this Analysis Unit?

a. What mechanisms are occurring?
b. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of habitat loss in X% of the occupied area in this analysis unit).
c. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
d. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area.
e. How confident are you that your interval lowest to highest (b and c above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%.

12) Are there additional significant threats impacting the species that have not been characterized above?
a. Describe the threat/threats here.

For each significant threat listed above, please answer the following questions.
b. What do you estimate the smallest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area (e.g., AST are exposed to the threat of habitat loss in X% of the occupied area in this analysis unit). If multiple threats are listed, please provide a separate estimate of spatial extent for each.
c. What do you estimate the largest spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area. If multiple threats are listed, please provide a separate estimate of spatial extent for each.
d. What do you think the most likely estimate of the actual spatial extent of affected occupied area is within this analysis unit? Answer in terms of the percentage of occupied area. If multiple threats are listed, please provide a separate estimate of spatial extent for each.
e. How confident are you that your interval lowest to highest (b and c above) captures the actual spatial extent of affected area? Please enter a number between 50% and 100%. If multiple threats are listed, please provide a separate estimate of your confidence in your estimates for each.

f. Does the threat/s have an effect on survival at the analysis unit scale of any particular life stage? If so, which life stage (i.e., nest survival, hatchling survival, juvenile survival, adult survival)?

g. What do you estimate is the lowest likely change in survival of this life stage as a result of this factor/s?
h. What do you estimate is the highest likely change in survival of this life stage as a result of this factor/s?
i. What do you think the most likely change in survival of this life stage as a result of this factor/s?
j. How confident are you that your interval lowest to highest (g and h above) captures the actual change in this life stage's survival for affected areas? Please enter a number between 50% and 100%.

13) Please list the primary factors (e.g., threats or conservation activities from the above questions known or believed to affect population demographic rates to a measurable degree at the analysis unit scale) occurring within this analysis unit in order of importance below **from most important to least important** (i.e. highest impact on demography to lowest impact). Please indicate the direction of the effect (positive or negative) in your response next to each factor.

Section 3: Range-Wide Influencing Factor Questions:

Note: For any question involving % survival – please indicate positive or negative change (e.g., -5%, +5%) for clarity. For the following questions, we define hatchlings as individuals aged 0-1 year that have emerged from the nest, juveniles as individuals > 1 year of age that have not yet reached sexual maturity, and adults as those that have reached sexual maturity. Nest survival refers to the survival of eggs to hatching.

14) Have any diseases been identified as impacting AST? If not, is there any reason to believe they are particularly at risk from disease impacts?

15) Have you predicted or observed vulnerability to or responses to climate change or drought? Can you provide any data or information on this vulnerability for the analysis?

16) In areas with commercial fishing are AST caught as bycatch? If yes:
a. What do you estimate is the lowest likely change in adult survival as a result of this factor?
b. What do you estimate is the highest likely change in adult survival as a result of this factor?
c. What is your best estimate of the change in adult survival resulting from this factor?
d. How confident are you that your interval lowest to highest (a and b above) captures the actual change in adult survival for affected areas? Please enter a number between 50% and 100%.
e. What do you estimate is the lowest likely change in juvenile survival as a result of this factor?
f. What do you estimate is the highest likely change in juvenile survival as a result of this factor?
g. What is your best estimate of the change in juvenile survival resulting from this factor?

h. How confident are you that your interval lowest to highest (e and f above) captures the actual change in juvenile survival for affected areas? Please enter a number between 50% and 100%.
i. What do you estimate is the lowest likely change in hatchling survival as a result of this factor?
j. What do you estimate is the highest likely change in hatchling survival as a result of this factor?
k. What is your best estimate of the change in hatchling survival resulting from this factor?
l. How confident are you that your interval lowest to highest (i and j above) captures the actual change in hatchling survival for affected areas? Please enter a number between 50% and 100%.

17) In areas with recreational fishing by trot lines and limb lines are AST caught as bycatch? If yes:
    a. What do you estimate is the lowest likely change in adult survival as a result of this factor?
    b. What do you estimate is the highest likely change in adult survival as a result of this factor?
    c. What is your best estimate of the change in adult survival resulting from this factor?
    d. How confident are you that your interval lowest to highest (a and b above) captures the actual change in adult survival for affected areas? Please enter a number between 50% and 100%.
    e. What do you estimate is the lowest likely change in juvenile survival as a result of this factor?
    f. What do you estimate is the highest likely change in juvenile survival as a result of this factor?
    g. What is your best estimate of the change in juvenile survival resulting from this factor?
    h. How confident are you that your interval lowest to highest (e and f above) captures the actual change in juvenile survival for affected areas? Please enter a number between 50% and 100%.
    i. What do you estimate is the lowest likely change in hatchling survival as a result of this factor?
    j. What do you estimate is the highest likely change in hatchling survival as a result of this factor?
    k. What is your best estimate of the change in hatchling survival resulting from this factor?
    l. How confident are you that your interval lowest to highest (i and j above) captures the actual change in hatchling survival for affected areas? Please enter a number between 50% and 100%.

18) If AST are released alive after being caught on a trot line or limb line are they at risk of adverse impacts associated with hook ingestion? If yes:
    a. What proportion of individuals released from a trot line or limb line do you think have ingested the fish hook?
    b. What do you estimate is the lowest likely change in adult survival as a result of this factor?

c. What do you estimate is the highest likely change in adult survival as a result of this factor?
d. What is your best estimate of the change in adult survival resulting from this factor?
e. How confident are you that your interval lowest to highest (a and b above) captures the actual change in adult survival for affected areas? Please enter a number between 50% and 100%.
f. What do you estimate is the lowest likely change in juvenile survival as a result of this factor?
g. What do you estimate is the highest likely change in juvenile survival as a result of this factor?
h. What is your best estimate of the change in juvenile survival resulting from this factor?
i. How confident are you that your interval lowest to highest (f and g above) captures the actual change in juvenile survival for affected areas? Please enter a number between 50% and 100%.

19) In areas with legal collection or harvest:
a. What do you estimate is the lowest likely change in adult survival as a result of this factor?
b. What do you estimate is the highest likely change in adult survival as a result of this factor?
c. What is your best estimate of the change in adult survival resulting from this factor?
d. How confident are you that your interval lowest to highest (a and b above) captures the actual change in adult survival for affected areas? Please enter a number between 50% and 100%.
e. What do you estimate is the lowest likely change in juvenile survival as a result of this factor?
f. What do you estimate is the highest likely change in juvenile survival as a result of this factor?
g. What is your best estimate of the change in juvenile survival resulting from this factor?
h. How confident are you that your interval lowest to highest (e and f above) captures the actual change in juvenile survival for affected areas? Please enter a number between 50% and 100%
i. What do you estimate is the lowest likely change in hatchling survival (survival to hatching) as a result of this factor?
j. What do you estimate is the highest likely change in hatchling survival as a result of this factor?
k. What is your best estimate of the change in hatchling survival resulting from this factor survival as a result of this factor?
l. How confident are you that your interval lowest to highest (i and j above) captures the actual change in hatchling survival for affected areas? Please enter a number between 50% and 100%.

20) In areas with illegal collection or harvest (i.e., poaching):
a. What do you estimate is the lowest likely change in adult survival as a result of this factor?

b. What do you estimate is the highest likely change in adult survival as a result of this factor?
c. What is your best estimate of the change in adult survival resulting from this factor as?
d. How confident are you that your interval lowest to highest (b and c above) captures the actual change in adult survival for affected areas? Please enter a number between 50% and 100%.
e. What do you estimate is the lowest likely change in juvenile survival as a result of this factor?
f. What do you estimate is the highest likely change in juvenile survival as a result of this factor?
g. What is your best estimate of the change in juvenile survival resulting from this factor?
h. How confident are you that your interval lowest to highest (b and c above) captures the actual change in juvenile survival for affected areas? Please enter a number between 50% and 100%.
i. What do you estimate is the lowest likely change in hatchling survival as a result of this factor?
j. What do you estimate is the highest likely change in hatchling survival as a result of this factor?
k. What is your best estimate of the change in hatchling survival resulting from this factor?
l. How confident are you that your interval lowest to highest (b and c above) captures the actual change in hatchling survival for affected areas? Please enter a number between 50% and 100%.
m. What do you estimate is the lowest likely change in nest survival (i.e., survival of eggs to hatching in the wild) as a result of this factor?
n. What is the highest likely change in nest survival as a result of this factor?
o. What is your best estimate of the change in nest survival resulting from this factor?
p. How confident are you that your interval lowest to highest (i and j above) captures the actual change in nest survival for affected areas? Please enter a number between 50% and 100%.

21) In areas with nest predation by subsidized non-native nest predators (e.g., *Sus scrofa*, *Procyon lotor*, *Solenopsis invicta*):
    a. What do you estimate is the lowest likely change in nest survival (survival of eggs to hatching; at a population scale, not the scale of a single nest) as a result of this factor?
    b. What do you estimate is the highest likely change in nest survival as a result of this factor?
    c. What is your best estimate of the change in nest survival resulting from this factor?
    d. How confident are you that your interval lowest to highest (a and b above) captures the actual change in nest survival for affected areas? Please enter a number between 50% and 100%).

APPENDIX D - Current and Historical Range by State and County

By state, alligator snapping turtles were historically found in 14 states: Alabama, Arkansas, Florida, Georgia, Illinois, Indiana, Kansas, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas.

Currently, the species is known to occur in: Alabama, Arkansas, Florida, Georgia, Illinois, Kentucky, Louisiana, Missouri, Mississippi, Oklahoma, Tennessee, and Texas. This list includes all of the historically occupied states with the exception of Indiana and Kansas, where it is unknown whether the species still persists. In Indiana, alligator snapping turtles have been detected from eDNA in the water, but presence has not been confirmed by trapping. In Kansas, the species has not been detected since a 1991 record in Montgomery County.

Table D1. Current and historical occupied status for counties within the alligator snapping turtle range. See Table 2 within the SSA for definitions of Occupied, Not Occupied, and Unknown. Counties that do not currently and did not historically support alligator snapping turtles are not shown.

State-County	Current	Historical	Last Record	Notes
AL-Autauga	Unknown	Occupied	-	
AL-Baldwin	Occupied	Occupied	2018	
AL-Barbour	Unknown	Occupied	-	
AL-Bibb	Unknown	Occupied	-	
AL-Blount	Occupied	Occupied	2010	
AL-Bullock	Unknown	Occupied	-	
AL-Butler	Unknown	Occupied	-	
AL-Calhoun	Unknown	Occupied	-	
AL-Chambers	Unknown	Occupied	-	
AL-Cherokee	Unknown	Occupied	-	
AL-Chilton	Unknown	Occupied	-	
AL-Choctaw	Unknown	Occupied	-	
AL-Clarke	Unknown	Occupied	1997	
AL-Clay	Unknown	Occupied	-	
AL-Cleburne	Unknown	Occupied	-	
AL-Coffee	Unknown	Occupied	-	
AL-Colbert	Unknown	Occupied	-	
AL-Conecuh	Unknown	Occupied	-	
AL-Coosa	Unknown	Occupied	1978	
AL-Covington	Unknown	Occupied	1996	
AL-Crenshaw	Unknown	Occupied	1996	
AL-Cullman	Occupied	Occupied	2017	
AL-Dale	Unknown	Occupied	-	
AL-Dallas	Unknown	Occupied	-	
AL-DeKalb	Unknown	Occupied	-	
AL-Elmore	Occupied	Occupied	2013	
AL-Escambia	Unknown	Occupied	2001	
AL-Etowah	Unknown	Occupied	-	
AL-Fayette	Unknown	Occupied	-	
AL-Franklin	Unknown	Occupied	-	
AL-Geneva	Unknown	Occupied	-	
AL-Greene	Unknown	Occupied	-	
AL-Hale	Occupied	Occupied	2017	
AL-Henry	Occupied	Occupied	2012	
AL-Houston	Unknown	Occupied	1992	

State-County	Current	Historical	Last Record	Notes
AL-Jackson	Unknown	Occupied	-	
AL-Jefferson	Unknown	Occupied	-	
AL-Lamar	Unknown	Occupied	-	
AL-Lauderdale	Unknown	Occupied	1980	
AL-Lawrence	Unknown	Occupied	-	
AL-Lee	Unknown	Occupied	1968	
AL-Limestone	Unknown	Occupied	-	
AL-Lowndes	Unknown	Occupied	-	
AL-Macon	Unknown	Occupied	1969	
AL-Madison	Unknown	Occupied	-	
AL-Marengo	Unknown	Occupied	-	
AL-Marion	Unknown	Occupied	-	
AL-Marshall	Unknown	Occupied	-	
AL-Mobile	Occupied	Occupied	2017	
AL-Monroe	Occupied	Occupied	2009	
AL-Montgomery	Unknown	Occupied	1998	
AL-Morgan	Unknown	Occupied	-	
AL-Perry	Occupied	Occupied	2015	
AL-Pickens	Unknown	Occupied	-	
AL-Pike	Unknown	Occupied	-	
AL-Randolph	Unknown	Occupied	-	
AL-Russell	Unknown	Occupied	1973	
AL-Shelby	Unknown	Occupied	1966	
AL-St. Clair	Unknown	Occupied	1914	
AL-Sumter	Unknown	Occupied	-	
AL-Talladega	Unknown	Occupied	-	
AL-Tallapoosa	Unknown	Occupied	-	
AL-Tuscaloosa	Unknown	Occupied	1975	
AL-Walker	Unknown	Occupied	1985	
AL-Washington	Occupied	Occupied	2017	
AL-Wilcox	Occupied	Occupied	2008	
AL-Winston	Unknown	Occupied	-	
AR-Arkansas	Occupied	Occupied	1995	
AR-Ashley	Occupied	Occupied	1995	
AR-Baxter	Unknown	Occupied	-	
AR-Benton	Unknown	Occupied	-	
AR-Boone	Unknown	Occupied	-	
AR-Bradley	Occupied	Occupied	2010	
AR-Calhoun	Occupied	Occupied	1995	
AR-Carroll	Unknown	Occupied	-	
AR-Chicot	Occupied	Occupied	1995	
AR-Clark	Occupied	Occupied	2009	
AR-Clay	Occupied	Occupied	1995	
AR-Cleburne	Occupied	Occupied	1995	
AR-Cleveland	Occupied	Occupied	1995	
AR-Columbia	Occupied	Occupied	-	
AR-Conway	Occupied	Occupied	2006	
AR-Craighead	Occupied	Occupied	1995	
AR-Crawford	Occupied	Occupied	1995	
AR-Crittenden	Occupied	Occupied	1995	
AR-Cross	Occupied	Occupied	1995	
AR-Dallas	Occupied	Occupied	-	
AR-Desha	Occupied	Occupied	1995	
AR-Drew	Occupied	Occupied	1995	

State-County	Current	Historical	Last Record	Notes
AR-Faulkner	Occupied	Occupied	2008	
AR-Franklin	Occupied	Occupied	1995	
AR-Fulton	Occupied	Occupied	1995	
AR-Garland	Occupied	Occupied	1995	
AR-Grant	Occupied	Occupied	1995	
AR-Greene	Occupied	Occupied	1995	
AR-Hempstead	Occupied	Occupied	2018	
AR-Hot Spring	Occupied	Occupied	2009	
AR-Howard	Occupied	Occupied	-	
AR-Independence	Occupied	Occupied	2015	
AR-Izard	Occupied	Occupied	1995	
AR-Jackson	Occupied	Occupied	1995	
AR-Jefferson	Occupied	Occupied	2018	
AR-Johnson	Occupied	Occupied	1995	
AR-Lafayette	Occupied	Occupied	1995	
AR-Lawrence	Occupied	Occupied	1995	
AR-Lee	Occupied	Occupied	1995	
AR-Lincoln	Occupied	Occupied	1995	
AR-Little River	Occupied	Occupied	2017	
AR-Logan	Occupied	Occupied	1995	
AR-Lonoke	Occupied	Occupied	1995	
AR-Madison	Unknown	Occupied	-	
AR-Marion	Occupied	Occupied	2010	
AR-Miller	Occupied	Occupied	2017	
AR-Mississippi	Occupied	Occupied	1995	
AR-Monroe	Occupied	Occupied	1995	
AR-Montgomery	Occupied	Occupied	-	
AR-Nevada	Occupied	Occupied	1995	
AR-Newton	Occupied	Occupied	2010	
AR-Ouachita	Occupied	Occupied	1995	
AR-Perry	Occupied	Occupied	1995	
AR-Phillips	Occupied	Occupied	1995	
AR-Pike	Occupied	Occupied	2016	
AR-Poinsett	Occupied	Occupied	1995	
AR-Polk	Occupied	Occupied	-	
AR-Pope	Occupied	Occupied	1995	
AR-Prairie	Occupied	Occupied	1995	
AR-Pulaski	Occupied	Occupied	1995	
AR-Randolph	Occupied	Occupied	2009	
AR-Saline	Occupied	Occupied	2005	
AR-Scott	Occupied	Occupied	-	
AR-Searcy	Occupied	Occupied	2010	
AR-Sebastian	Occupied	Occupied	1995	
AR-Sevier	Occupied	Occupied	-	
AR-Sharp	Occupied	Occupied	1995	
AR-St. Francis	Occupied	Occupied	1995	
AR-Stone	Occupied	Occupied	1995	
AR-Union	Occupied	Occupied	1995	
AR-Van Buren	Occupied	Occupied	-	
AR-Washington	Unknown	Occupied	-	
AR-White	Occupied	Occupied	1995	
AR-Woodruff	Occupied	Occupied	1995	
AR-Yell	Occupied	Occupied	1995	
FL-Alachua	Occupied	Occupied	2012	

State-County	Current	Historical	Last Record	Notes
FL-Bay	Occupied	Occupied	2018	
FL-Bradford	Occupied	Occupied	2011	
FL-Calhoun	Occupied	Occupied	2018	
FL-Columbia	Occupied	Occupied	2012	
FL-Dixie	Occupied	Occupied	2014	
FL-Escambia	Occupied	Occupied	2018	
FL-Franklin	Occupied	Occupied	2019	
FL-Gadsden	Occupied	Occupied	2018	
FL-Gilchrist	Occupied	Occupied	2014	
FL-Gulf	Occupied	Occupied	2018	
FL-Hamilton	Occupied	Occupied	2017	
FL-Holmes	Occupied	Occupied	2018	
FL-Jackson	Occupied	Occupied	2019	
FL-Jefferson	Unknown	Unknown	-	2000 newspaper photo purportedly from Aucilla River, but trapping has been unsuccessful in this likely distribution gap
FL-Lafayette	Occupied	Occupied	2014	
FL-Leon	Occupied	Occupied	2018	
FL-Levy	Occupied	Occupied	2014	
FL-Liberty	Occupied	Occupied	2018	
FL-Madison	Occupied	Occupied	2012	
FL-Marion	Unknown	Not Occupied	-	2 museum records from the Ocklawaha River in 1916 and 1955, but the species is not thought to occur in St. Johns River drainage, may be introduced here
FL-Okaloosa	Occupied	Occupied	2018	
FL-Santa Rosa	Occupied	Occupied	2018	
FL-Suwannee	Occupied	Occupied	2014	
FL-Union	Occupied	Occupied	2011	
FL-Wakulla	Occupied	Occupied	2018	
FL-Walton	Occupied	Occupied	2018	
FL-Washington	Occupied	Occupied	2018	
GA-Atkinson	Occupied	Occupied	2018	
GA-Baker	Occupied	Occupied	2017	
GA-Ben Hill	Unknown	Unknown	-	
GA-Berrien	Occupied	Occupied	2018	
GA-Brooks	Occupied	Occupied	2018	
GA-Calhoun	Unknown	Unknown	-	
GA-Chattahoochee	Occupied	Occupied	2010	
GA-Clay	Occupied	Occupied	2003	
GA-Clayton	Occupied	Occupied	2011	
GA-Clinch	Unknown	Occupied	-	
GA-Colquitt	Occupied	Occupied	2018	
GA-Cook	Occupied	Occupied	1998	
GA-Coweta	Occupied	Occupied	2010	
GA-Crawford	Occupied	Occupied	2014	
GA-Crisp	Occupied	Occupied	1989	
GA-Decatur	Occupied	Occupied	2014	
GA-Dooly	Occupied	Occupied	2014	
GA-Dougherty	Occupied	Occupied	2014	
GA-Early	Occupied	Occupied	2001	
GA-Echols	Occupied	Occupied	2018	
GA-Fayette	Occupied	Occupied	2011	
GA-Fulton	Unknown	Not Occupied	-	

State-County	Current	Historical	Last Record	Notes
GA-Grady	Occupied	Occupied	1997	
GA-Irwin	Occupied	Occupied	2017	
GA-Lanier	Occupied	Occupied	1997	
GA-Lee	Occupied	Occupied	2014	
GA-Lowndes	Occupied	Occupied	2018	
GA-Macon	Occupied	Occupied	2014	
GA-Marion	Occupied	Occupied	1996	
GA-Meriwether	Occupied	Occupied	2005	
GA-Miller	Occupied	Occupied	2000	
GA-Mitchell	Occupied	Occupied	2014	
GA-Muscogee	Occupied	Occupied	1997	
GA-Peach	Occupied	Occupied	2014	
GA-Pike	Occupied	Occupied	2005	
GA-Quitman	Occupied	Occupied	2001	
GA-Randolph	Unknown	Not Occupied	-	
GA-Schley	Unknown	Not Occupied	-	
GA-Seminole	Occupied	Occupied	2001	
GA-Spalding	Occupied	Occupied	2011	
GA-Stewart	Occupied	Occupied	2004	
GA-Sumter	Occupied	Occupied	2014	
GA-Talbot	Unknown	Unknown	-	
GA-Taylor	Occupied	Occupied	2014	
GA-Terrell	Unknown	Unknown	-	
GA-Thomas	Occupied	Occupied	2006	
GA-Tift	Unknown	Unknown	-	
GA-Turner	Unknown	Not Occupied	-	
GA-Upson	Occupied	Occupied	2014	
GA-Ware	Unknown	Not Occupied	-	
GA-Webster	Unknown	Unknown	-	
GA-Wilcox	Unknown	Not Occupied	-	
GA-Worth	Occupied	Occupied	2014	
IL-Adams	Unknown	Occupied	1892	
IL-Alexander	Unknown	Occupied	1907	
IL-Calhoun	Unknown	Occupied	1954	
IL-Jackson	Unknown	Occupied	1960	
IL-Jersey	Unknown	Occupied	1961	
IL-Mason	Unknown	Occupied	1961	
IL-Massac	Unknown	Occupied	1937	
IL-Peoria	Unknown	Occupied	1976	
IL-Randolph	Unknown	Occupied	1937	
IL-Rock Island	Unknown	Occupied	1950	
IL-Union	Occupied	Occupied	2014	
IL-Wabash	Unknown	Occupied	1887	
IL-White	Unknown	Occupied	1892	
IN-Gibson	Unknown	Unknown	2017	positive eDNA
IN-Jackson	Occupied	Unknown	2012	
IN-Morgan	Unknown	Occupied	1991	
IN-Pike	Unknown	Unknown	2017	positive eDNA
IN-Posey	Unknown	Occupied	1938	
KS-Allen	Not Occupied	Unknown	-	
KS-Anderson	Not Occupied	Unknown	-	
KS-Butler	Not Occupied	Occupied	1912	
KS-Chase	Not Occupied	Unknown	-	
KS-Chautauqua	Unknown	Unknown	-	

State-County	Current	Historical	Last Record	Notes
KS-Cherokee	Unknown	Occupied	1895	
KS-Coffey	Not Occupied	Unknown	-	
KS-Cowley	Not Occupied	Occupied	1958	
KS-Labette	Unknown	Occupied	1938	
KS-Lyon	Unknown	Occupied	1967	
KS-Marion	Not Occupied	Occupied	1912	
KS-Montgomery	Unknown	Occupied	1991	
KS-Morris	Unknown	Unknown	-	
KS-Neosho	Unknown	Occupied	1911	
KS-Sumner	Unknown	Unknown	-	
KS-Wilson	Not Occupied	Unknown	-	
KS-Woodson	Not Occupied	Unknown	-	
KY-Allen	Unknown	Unknown	-	
KY-Ballard	Unknown	Occupied	1998	
KY-Barren	Unknown	Unknown	-	
KY-Breckinridge	Unknown	Unknown	-	
KY-Butler	Unknown	Unknown	-	
KY-Caldwell	Occupied	Unknown	2003	
KY-Calloway	Occupied	Occupied	2004	
KY-Carlisle	Unknown	Occupied	1979	
KY-Christian	Unknown	Unknown	-	
KY-Crittenden	Unknown	Unknown	-	
KY-Daviess	Unknown	Unknown	-	
KY-Edmonson	Unknown	Unknown	-	
KY-Fulton	Unknown	Occupied	1975	
KY-Graves	Unknown	Unknown	-	
KY-Grayson	Unknown	Unknown	-	
KY-Hancock	Unknown	Unknown	-	
KY-Hardin	Unknown	Unknown	-	
KY-Hart	Unknown	Unknown	-	
KY-Henderson	Unknown	Unknown	-	
KY-Hickman	Occupied	Unknown	2002	
KY-Hopkins	Unknown	Unknown	-	
KY-Jefferson	Unknown	Unknown	-	
KY-Livingston	Unknown	Occupied	1994	
KY-Logan	Unknown	Unknown	-	
KY-Lyon	Unknown	Unknown	-	
KY-Marshall	Unknown	Occupied	1969	
KY-McCracken	Unknown	Occupied	1990	
KY-McLean	Unknown	Unknown	-	
KY-Meade	Unknown	Unknown	-	
KY-Monroe	Unknown	Unknown	-	
KY-Muhlenberg	Unknown	Unknown	-	
KY-Ohio	Unknown	Unknown	-	
KY-Simpson	Unknown	Unknown	-	
KY-Todd	Unknown	Unknown	-	
KY-Trigg	Unknown	Unknown	-	
KY-Union	Unknown	Unknown	-	
KY-Warren	Unknown	Unknown	-	
KY-Webster	Unknown	Unknown	-	
LA-Acadia	Occupied	Occupied	2016	
LA-Allen	Occupied	Occupied	2012	
LA-Ascension	Occupied	Occupied	1999	
LA-Assumption	Occupied	Occupied	1998	

State-County	Current	Historical	Last Record	Notes
LA-Avoyelles	Occupied	Occupied	2000	
LA-Beauregard	Occupied	Occupied	2018	
LA-Bienville	Occupied	Occupied	2000	
LA-Bossier	Occupied	Occupied	2014	
LA-Caddo	Occupied	Occupied	2000	
LA-Calcasieu	Occupied	Occupied	2014	
LA-Caldwell	Occupied	Occupied	2013	
LA-Cameron	Unknown	Unknown	-	
LA-Catahoula	Occupied	Occupied	2000	
LA-Claiborne	Occupied	Occupied	-	
LA-Concordia	Occupied	Occupied	1999	
LA-De Soto	Occupied	Occupied	2000	
LA-East Baton Rouge	Occupied	Occupied	2014	
LA-East Carroll	Occupied	Occupied	1947	
LA-East Feliciana	Occupied	Occupied	1994	
LA-Evangeline	Occupied	Occupied	2000	
LA-Franklin	Occupied	Occupied	-	
LA-Grant	Occupied	Occupied	1965	
LA-Iberia	Occupied	Occupied	2014	
LA-Iberville	Occupied	Occupied	1998	
LA-Jackson	Occupied	Occupied	-	
LA-Jefferson	Occupied	Occupied	1962	
LA-Jefferson Davis	Occupied	Occupied	2012	
LA-Lafayette	Occupied	Occupied	2016	
LA-Lafourche	Occupied	Occupied	1950	
LA-LaSalle	Occupied	Occupied	2000	
LA-Lincoln	Occupied	Occupied	-	
LA-Livingston	Occupied	Occupied	2004	
LA-Madison	Occupied	Occupied	-	
LA-Morehouse	Occupied	Occupied	2015	
LA-Natchitoches	Occupied	Occupied	2014	
LA-Orleans	Occupied	Occupied	1950	
LA-Ouachita	Occupied	Occupied	1983	
LA-Plaquemines	Occupied	Occupied	1997	
LA-Pointe Coupee	Occupied	Occupied	1999	
LA-Rapides	Occupied	Occupied	2014	
LA-Red River	Occupied	Occupied	2000	
LA-Richland	Occupied	Occupied	-	
LA-Sabine	Occupied	Occupied	1974	
LA-St. Bernard	Occupied	Occupied	-	
LA-St. Charles	Occupied	Occupied	1997	
LA-St. Helena	Occupied	Occupied	-	
LA-St. James	Occupied	Occupied	1997	
LA-St. John the Baptist	Occupied	Occupied	1997	
LA-St. Landry	Occupied	Occupied	1970	
LA-St. Martin	Occupied	Occupied	2014	
LA-St. Mary	Occupied	Occupied	2014	
LA-St. Tammany	Occupied	Occupied	1997	
LA-Tangipahoa	Occupied	Occupied	2004	
LA-Tensas	Occupied	Occupied	-	
LA-Terrebonne	Occupied	Occupied	1999	
LA-Union	Occupied	Occupied	1950	
LA-Vermilion	Occupied	Occupied	1998	
LA-Vernon	Occupied	Occupied	2007	

State-County	Current	Historical	Last Record	Notes
LA-Washington	Occupied	Occupied	2018	
LA-Webster	Occupied	Occupied	2014	
LA-West Baton Rouge	Occupied	Occupied	-	
LA-West Carroll	Occupied	Occupied	-	
LA-West Feliciana	Occupied	Occupied	1999	
LA-Winn	Occupied	Occupied	2014	
MO-Bollinger	Occupied	Occupied	2013	
MO-Butler	Occupied	Occupied	2010	
MO-Cape Girardeau	Occupied	Unknown	2018	
MO-Carter	Unknown	Unknown	-	
MO-Christian	Unknown	Unknown	-	
MO-Douglas	Occupied	Occupied	2012	
MO-Dunklin	Occupied	Occupied	2010	
MO-Greene	Occupied	Unknown	2008	
MO-Howell	Occupied	Unknown	2017	
MO-Lewis	Not Occupied	Occupied	1965	
MO-Madison	Occupied	Unknown	2018	
MO-Mississippi	Occupied	Occupied	2007	
MO-New Madrid	Unknown	Occupied	1993	
MO-Oregon	Occupied	Unknown	2004	
MO-Ozark	Occupied	Occupied	2008	
MO-Pemiscot	Occupied	Occupied	2009	
MO-Ripley	Occupied	Occupied	2017	
MO-Scott	Unknown	Unknown	-	
MO-Shannon	Occupied	Unknown	2016	
MO-St. Francois	Unknown	Occupied	1948	
MO-St. Louis	Occupied	Unknown	2014	
MO-Stoddard	Occupied	Occupied	2013	
MO-Stone	Occupied	Unknown	2008	
MO-Taney	Occupied	Occupied	2004	
MO-Wayne	Occupied	Occupied	2018	
MS-Adams	Unknown	Unknown	-	
MS-Alcorn	Unknown	Unknown	-	
MS-Amite	Unknown	Unknown	-	
MS-Attala	Occupied	Unknown	2018	
MS-Benton	Unknown	Unknown	-	
MS-Bolivar	Unknown	Unknown	-	
MS-Calhoun	Unknown	Unknown	-	
MS-Carroll	Occupied	Unknown	2000	
MS-Chickasaw	Unknown	Unknown	-	
MS-Choctaw	Unknown	Unknown	-	
MS-Claiborne	Unknown	Unknown	-	
MS-Clarke	Occupied	Unknown	2018	
MS-Clay	Unknown	Unknown	-	
MS-Coahoma	Unknown	Unknown	-	
MS-Copiah	Occupied	Unknown	2018	
MS-Covington	Occupied	Unknown	2017	
MS-DeSoto	Unknown	Unknown	-	
MS-Forrest	Occupied	Occupied	-	
MS-Franklin	Unknown	Unknown	-	
MS-George	Occupied	Unknown	2018	
MS-Greene	Occupied	Occupied	2018	
MS-Grenada	Unknown	Unknown	-	
MS-Hancock	Occupied	Unknown	2018	

State-County	Current	Historical	Last Record	Notes
MS-Harrison	Unknown	Occupied	1991	
MS-Hinds	Occupied	Unknown	2018	
MS-Holmes	Occupied	Unknown	2018	
MS-Humphreys	Unknown	Occupied	1973	
MS-Issaquena	Unknown	Occupied	1977	
MS-Itawamba	Unknown	Unknown	-	
MS-Jackson	Occupied	Unknown	-	
MS-Jasper	Unknown	Unknown	-	
MS-Jefferson	Unknown	Unknown	-	
MS-Jefferson Davis	Unknown	Unknown	-	
MS-Jones	Unknown	Unknown	-	
MS-Kemper	Unknown	Unknown	-	
MS-Lafayette	Unknown	Unknown	-	
MS-Lamar	Unknown	Unknown	-	
MS-Lauderdale	Unknown	Unknown	-	
MS-Lawrence	Occupied	Unknown	2018	
MS-Leake	Occupied	Unknown	2018	
MS-Lee	Unknown	Unknown	-	
MS-Leflore	Occupied	Unknown	2000	
MS-Lincoln	Unknown	Unknown	-	
MS-Lowndes	Unknown	Unknown	-	
MS-Madison	Occupied	Unknown	2018	
MS-Marion	Occupied	Occupied	2018	
MS-Marshall	Unknown	Unknown	-	
MS-Monroe	Unknown	Unknown	-	
MS-Montgomery	Unknown	Unknown	-	
MS-Neshoba	Occupied	Occupied	2018	
MS-Newton	Occupied	Occupied	2016	
MS-Noxubee	Occupied	Occupied	2018	
MS-Oktibbeha	Unknown	Occupied	1992	
MS-Panola	Unknown	Occupied	1992	
MS-Pearl River	Occupied	Unknown	2018	
MS-Perry	Occupied	Occupied	2017	
MS-Pike	Occupied	Unknown	2018	
MS-Pontotoc	Unknown	Unknown	-	
MS-Prentiss	Unknown	Unknown	-	
MS-Quitman	Unknown	Unknown	-	
MS-Rankin	Occupied	Occupied	2018	
MS-Scott	Occupied	Unknown	2018	
MS-Sharkey	Unknown	Unknown	-	
MS-Simpson	Occupied	Unknown	2018	
MS-Smith	Unknown	Unknown	-	
MS-Stone	Occupied	Unknown	2018	
MS-Sunflower	Occupied	Occupied	2018	
MS-Tallahatchie	Occupied	Occupied	2018	
MS-Tate	Unknown	Unknown	-	
MS-Tippah	Unknown	Unknown	-	
MS-Tishomingo	Unknown	Unknown	-	
MS-Tunica	Occupied	Unknown	2009	
MS-Union	Unknown	Unknown	-	
MS-Walthall	Unknown	Unknown	-	
MS-Warren	Unknown	Occupied	1977	
MS-Washington	Occupied	Occupied	2018	
MS-Wayne	Occupied	Unknown	2017	

State-County	Current	Historical	Last Record	Notes
MS-Webster	Unknown	Unknown	-	
MS-Wilkinson	Unknown	Unknown	-	
MS-Winston	Unknown	Unknown	-	
MS-Yalobusha	Unknown	Unknown	-	
MS-Yazoo	Occupied	Occupied	2018	
OK-Adair	Unknown	Unknown	-	
OK-Atoka	Occupied	Occupied	2015	
OK-Bryan	Unknown	Occupied	1960	
OK-Carter	Unknown	Unknown	-	
OK-Cherokee	Unknown	Occupied	1941	
OK-Choctaw	Unknown	Occupied	-	
OK-Coal	Unknown	Unknown	-	
OK-Craig	Unknown	Occupied	1952	
OK-Creek	Unknown	Unknown	-	
OK-Delaware	Unknown	Unknown	-	
OK-Haskell	Occupied	Occupied	2002	
OK-Hughes	Unknown	Unknown	-	
OK-Johnston	Unknown	Occupied	-	
OK-Kay	Unknown	Unknown	-	
OK-Latimer	Unknown	Unknown	-	
OK-Le Flore	Occupied	Occupied	2018	
OK-Marshall	Unknown	Occupied	-	
OK-Mayes	Occupied	Occupied	2018	
OK-McCurtain	Occupied	Occupied	2004	
OK-McIntosh	Occupied	Occupied	2009	
OK-Muskogee	Occupied	Occupied	2010	
OK-Nowata	Unknown	Occupied	-	
OK-Okfuskee	Unknown	Unknown	-	
OK-Okmulgee	Unknown	Occupied	1994	
OK-Osage	Unknown	Occupied	-	
OK-Ottawa	Unknown	Occupied	-	
OK-Pawnee	Not Occupied	Unknown	-	
OK-Pittsburg	Occupied	Occupied	2001	
OK-Pontotoc	Not Occupied	Unknown	-	
OK-Pushmataha	Occupied	Occupied	2004	
OK-Rogers	Unknown	Occupied	1939	
OK-Sequoyah	Occupied	Occupied	2010	
OK-Tulsa	Not Occupied	Occupied	1931	
OK-Wagoner	Unknown	Occupied	1992	
OK-Washington	Unknown	Occupied	1939	
OK-Woods	Not Occupied	Unknown	-	
TN-Benton	Occupied	Occupied	-	
TN-Carroll	Not Occupied	Occupied	-	
TN-Chester	Not Occupied	Occupied	-	
TN-Crockett	Not Occupied	Occupied	-	
TN-Davidson	Occupied	Occupied	2015	
TN-Decatur	Occupied	Occupied	2017	
TN-DeKalb	Occupied	Occupied	2017	
TN-Dyer	Occupied	Occupied	2016	
TN-Fayette	Occupied	Occupied	2018	
TN-Gibson	Not Occupied	Occupied	-	
TN-Hardeman	Not Occupied	Occupied	1970	
TN-Hardin	Not Occupied	Occupied	-	
TN-Haywood	Not Occupied	Occupied	-	

State-County	Current	Historical	Last Record	Notes
TN-Henderson	Not Occupied	Occupied	-	
TN-Henry	Not Occupied	Occupied	1965	
TN-Houston	Occupied	Occupied	2000	
TN-Humphreys	Occupied	Occupied	2017	
TN-Lake	Occupied	Occupied	2018	
TN-Lauderdale	Not Occupied	Occupied	-	
TN-Madison	Not Occupied	Occupied	-	
TN-McNairy	Not Occupied	Occupied	1975	
TN-Montgomery	Not Occupied	Occupied	-	
TN-Obion	Occupied	Occupied	2015	
TN-Perry	Not Occupied	Occupied	1971	
TN-Shelby	Occupied	Occupied	2016	
TN-Stewart	Occupied	Occupied	2017	
TN-Tipton	Occupied	Occupied	2017	
TN-Wayne	Occupied	Occupied	2006	
TN-Wilson	Not Occupied	Occupied	1983	
TX-Anderson	Occupied	Occupied	2014	
TX-Angelina	Occupied	Unknown	2016	
TX-Bowie	Occupied	Unknown	2010	
TX-Camp	Unknown	Unknown	-	
TX-Cass	Occupied	Unknown	2014	
TX-Chambers	Unknown	Unknown	-	
TX-Cherokee	Unknown	Occupied	2013	
TX-Collin	Occupied	Unknown	2002	
TX-Dallas	Unknown	Unknown	-	
TX-Delta	Unknown	Unknown	-	
TX-Fannin	Unknown	Occupied	1993	
TX-Franklin	Unknown	Occupied	1986	
TX-Freestone	Unknown	Occupied	2013	
TX-Grayson	Unknown	Occupied	1993	
TX-Gregg	Unknown	Occupied	2013	
TX-Hardin	Occupied	Occupied	2018	
TX-Harris	Occupied	Occupied	2019	
TX-Harrison	Occupied	Occupied	2015	
TX-Henderson	Occupied	Occupied	2014	
TX-Hopkins	Unknown	Occupied	2013	
TX-Houston	Unknown	Occupied	1986	
TX-Jasper	Occupied	Occupied	2016	
TX-Jefferson	Unknown	Occupied	2013	
TX-Lamar	Unknown	Occupied	1993	
TX-Leon	Occupied	Occupied	2013	
TX-Liberty	Occupied	Occupied	2016	
TX-Madison	Occupied	Unknown	2017	
TX-Marion	Occupied	Occupied	2009	
TX-Montgomery	Occupied	Unknown	2019	
TX-Morris	Unknown	Unknown	-	
TX-Nacogdoches	Occupied	Occupied	2001	
TX-Newton	Occupied	Occupied	2000	
TX-Orange	Unknown	Occupied	2013	
TX-Panola	Occupied	Occupied	2004	
TX-Polk	Unknown	Occupied	2013	
TX-Rains	Unknown	Occupied	1985	
TX-Red River	Unknown	Occupied	2013	
TX-Rockwall	Unknown	Unknown	-	

State-County	Current	Historical	Last Record	Notes
TX-Rusk	Occupied	Occupied	2016	
TX-Sabine	Occupied	Occupied	2000	
TX-San Augustine	Unknown	Unknown	-	
TX-San Jacinto	Occupied	Occupied	2000	
TX-Shelby	Occupied	Occupied	2016	
TX-Smith	Occupied	Occupied	2014	
TX-Tarrant	Occupied	Unknown	2018	
TX-Titus	Unknown	Occupied	2013	
TX-Trinity	Unknown	Unknown	-	
TX-Tyler	Occupied	Occupied	2010	
TX-Upshur	Unknown	Unknown	-	
TX-Van Zandt	Unknown	Unknown	-	
TX-Walker	Occupied	Occupied	2000	
TX-Wood	Occupied	Occupied	2001	

APPENDIX E - Future Condition Model Methods and Results

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OVERVIEW

Here we describe the analytical framework used to evaluate the current and future conditions of alligator snapping turtle (hereafter AST) populations across their range. We constructed a female-only, stage-structured Lefkovitch matrix model to project AST population dynamics over 50 annual timesteps. We used the best available data from the literature to parameterize the projection matrix, and elicited data from taxon experts to quantify stage-specific initial abundance, the spatial extent of threats, and threat-specific percent reductions to survival. To reflect differences among analysis units, we adjusted initial abundance and some demographic parameters within the matrix model based on the proportion of the population within the unit exposed to each threat, including all threat-overlap combinations. To account for potential uncertainty in the effects of each threat, we created six different scenarios, in which a portion of the expert-elicited threat-induced reductions to survival were unaltered, increased, or decreased, and the spatial extent of each threat left the same, or reduced to simulate "conservation actions". We used a stochastic projection model that accounted for parametric uncertainty in the demographic parameters, to predict future conditions of the AST in four of the seven analysis units under the six different scenarios. We then used the model output to predict the probability of extirpation and quasi-extirpation, defined here as the probability that the total AST population declined to less than 5% of the population size in year one of the simulation within an analysis unit.

METHODS

Expert Elicitation

We relied on expert elicitation to fill information gaps needed to project AST population dynamics under alternative scenarios of future conditions. For modeling purposes, we used remote expert elicitation to parameterize stage-specific initial abundance, habitat loss mechanisms, the spatial extent of threats, and threat-specific percent reductions to survival. We conducted a four-point elicitation (Speirs-Bridge et al. 2010, p. 515) of the expert team via e-mail (questions in Appendix C), in which we asked the respondent to provide a minimum, maximum, and mean numerical value, as well as the percent confidence that the true mean was within the minimum and maximum range for quantity-based questions. We applied the same quality control and summarization process to all questions that were pertinent to our modeling efforts. Specifically, we only included responses to individual questions that included at least the first three quantities (minimum, maximum, mean), and assigned a value of 50% to all missing or blank confidence values. Using these responses, we back calculated the distribution that each expert was describing by assuming the minimum and maximum were equivalent to the upper and lower boundaries of a 95% confidence interval around the identified mean value. For each response, we calculated two quantities that described the potential error range: mean (μ) minus the minimum divided by 1.96 (SD1) and maximum minus mean divided by 1.96 (SD2), this essentially reverses the 95% confidence interval calculations (95% C.I. = $\mu \pm 1.96 \times \sigma$). This approach assumes a normal, or bell curve, shape to the distributions which may not be true since for some experts that mean value was closer to the minimum or maximum that in the middle for some quantities. For each question, we then calculated the weighted mean across experts for mean, SD1, and SD2, using the percent confidence quantity as weights. Lastly, we averaged the weighted averages of SDs 1 and 2 to create a single measure of error.

The responses for the Western, Southern Mississippi – West, and Northern Mississippi – West analysis units did not meet the minimum quality control standards for the unit-specific quantities (e.g., initial abundance, spatial extent of threats); therefore, we dropped these units from the modeling framework. The exclusion of these units did not affect the range-wide quantities (e.g., threat-specific reductions to parameters), as all responses that met the quality control standards were included, regardless of the expert's analysis unit affiliation.

Matrix Model Construction

We constructed a female-only, stage-structured Lefkovitch matrix model (Caswell 2001, p. 33) to project alligator snapping turtle (AST) population dynamics over annual timesteps in each analysis unit. We based our model off the peer reviewed and published model in Folt et al. (2016, p. 24) and corrected the model to reflect guidance on the appropriate structure of matrix population models (Kendall et al. 2019, p. 33) and to better support the SSA needs. Our conceptual model of the AST's life cycle (Figure E1) that parameterized the matrix model used a prebreeding census structure with two life stages: Juveniles (J) included individuals ≥1 year-old that had not reached reproductive maturity, whereas Adults (A) included mature, breeding individuals. For each timestep (year), individuals in the juvenile stage could either remain a juvenile with probability P_J or transition to the adult stage (grow) with probability G_J:

$$G_J = \phi_J \times \gamma_J$$

$$P_J = \phi_J \times (1 - \gamma_J)$$

where φ_J is annual juvenile survival and γ_J is the fraction of individuals that reach maturity at the end of the timestep. Upon reaching reproductive maturity, the probability of remaining in the adult stage class (P_A) was equal to adult annual survival φ_A (Figure D1). Given the prebreeding census structure, adults were the only stage class contributing to fecundity (F_A), the number of female offspring produced per adult female in each timestep:

$$F_A = BP \times CS \times \phi_N \times NSC \times FP \times \phi_H \qquad \text{(Eq. 1)}$$

in which *BP* is the proportion of adult females that breed annually and *CS* is clutch size. Nest survival (φ_N) is the proportion of nests in which one egg successfully hatched, whereas nest success (*NSC*) is the proportion of eggs from which a hatchling successfully emerged in surviving nests, *FP* is the proportion of female hatchlings (neonates), and φ_H is the survival rate for hatchlings from nest emergence to one year of age.

Matrix model parameterization. — To parameterize the four elements (P_J, G_J, P_A, F_A) of our projection matrix **A**, we used a combination of demographic parameter estimates elicited from taxon experts, and the literature for AST or closely-related species (e.g., *Chelydra serpentina*). When possible, we selected for demographic parameters from reference populations that had minimal exposure to threats, meaning that their parameter estimates were a closer approximation of the parameter's "true" or "biological" value and more appropriate for perturbation analyses that seek to isolate the effects of threats and stressors. Though we created separate projection matrices for each analysis unit *u* ($\mathbf{A}_u$), all demographic parameters used to calculate the matrix elements were the same across all seven units, with the exception of φ_J. This approach assumed that differences in demographic parameters among the analysis units were driven by unit-specific factors such as climate or exposure to threats (e.g., fishing bycatch).

$$\boldsymbol{A_u} = \begin{bmatrix} P_{J,u} & F_A \\ G_{J,u} & P_A \end{bmatrix}$$

We incorporated stochasticity into our modeling framework by modeling each demographic parameter (summarized in Table E1) as a draw from a statistical distribution based on the parameter's mean (μ) and sampling standard deviation (σ_S; σ hereafter). In our simulation model we partitioned our variance into sampling variance (to model parametric uncertainty) and temporal variability according to the methods described by McGowan et al. (2011, p. 1401) and here we report the mean and sampling standard deviation (square root of the sampling variance) for brevity. For all analysis units except Northern Mississippi – East, we based the φ_J parameter on an apparent survival estimate from a 16-year mark-recapture study of an AST reference population located within Spring Creek, Georgia, USA (0.86; Folt et al. 2016, p. 26). In our model, however, we increased the Folt et al. (2016) juvenile apparent survival estimate by 5% (μ: 0.90, σ: 0.027) to account for potential dispersal (i.e., permanent emigration) of juvenile AST. Juvenile AST are known to move greater distances compared to adults (Riedle et al. 2006, p. 37), though no peer-reviewed estimates of AST natal dispersal rates exist. We applied a different juvenile survival estimate for the Northern Mississippi – East analysis unit, which includes the northern extent of the AST's geographic distribution (Thompson et al. 2016, p. 429), to reflect the effects of cooler temperatures that can increase mortality in juvenile age classes during winter months (Dreslik et al. 2017, p. 22). We used the median annual survival estimate (μ: 0.73, σ: 0.035) for individuals aged 1–16 reported by Dreslik et al. (2017, Table 21, p. 26). The juvenile survival rates reported in Dreslik et al. (2017, p. 22) were estimated from a known-fate analysis, in which dispersal events can be distinguished from mortality, therefore we did not increase the survival estimates, as done for the other units. The age-specific survival estimates were derived by interpolating a decay function between a hatchling survival rate (Bass 2007, entire) and an asymptotic adult survival rate (beginning at 17 years of age) reported by Folt et al. (2016, p. 27). The decay function used age-specific survival data points from head-started AST juveniles released as 2, 3, and 4-year olds in southern Illinois (Dreslik et al. 2017, p. 26).

Juvenile female AST reach sexual maturity (i.e., transition to the adult stage) at 13–21 years of age (Tucker and Sloan 1997, p. 589), for a median juvenile stage duration of 16 years. We derived γ_J, the proportion of individuals transitioning from the juvenile to adult stage in each timestep, using the asymptotic age-within-stage structure (AAS) formula (Kendall et al. 2019, p. 36):

$$\gamma_J = \frac{(\phi_J/\lambda_1)^{T_J-1}}{\sum_{k=0}^{T_J-1}(\phi_J/\lambda_1)^k}$$

where T_J is the mean duration in the juvenile stage (16 years) indexed by *k* years, and λ_1 is the asymptotic growth rate. Specifically, we used the 'make_stage4age_matrix' function in the *mpmtools* package (Kendall 2019, website) within the R statistical program (R Core Team 2019, software) to apply the AAS formula and solve for γ_J. We assumed that sexual maturity was based on age, rather than size, and used the same γ_J value for all analysis units (μ: 0.019, σ: 0.011), despite a negative association between juvenile growth rates and latitude (Dreslik et al. 2017, p. 36). Thus, our analysis assumed that females in northern areas reach sexual maturity at a smaller size, but similar age to females in southern portions of the AST range.

We parameterized adult survival (φ_A) using the estimate reported by Folt et al. (2016, p. 26; μ: 0.95, σ: 0.017) for all analysis units. Studies suggest that not all adult AST females breed every year (Dobie 1971, 650), therefore we set breeding probability (*BP*) within the adult fecundity formula (Eq. 1) to 0.98 (σ: 0.011). Though clutch sizes in turtles are thought to positively vary with latitude (Iverson et al. 1993, p. 2450), existing clutch sizes reported for AST did not adhere to this pattern (Table E2). Therefore, we constructed a weighted mean of clutch sizes reported across the AST's range (Table E2), using the number of nests from each study as weights, and the standard deviation used in Folt et al. (2016; p. 26) to model clutch size (*CS*; μ: 33.2, σ: 10).

We used parameter estimates from an AST nesting study in the lower Apalachicola drainage (Ewert et al. 2006, p. 67) in the Apalachicola analysis unit to model nest survival (φ_N; μ: 0.13, σ: 0.027) and nest success (*NSC*; μ: 0.72, σ: 0.036). Sex in AST is environmentally determined based on incubation temperatures and follows Pattern II in which predominantly produces males at temperatures 24–27°C, and temperatures below or above this range produce mainly females (Ewert et al. 1994, p. 10). No published estimates of wild AST hatchling sex ratios from unperturbed nests exist, though relatively even sex ratios have been reported for *C. serpentina* (0.47; Congdon et al. 1994) and other turtle species (Heppell 1998, p. 369). Therefore, as consistent with previous AST population viability assessments (Folt et al. 2016, p. 25, Dreslik et al. 2017, p. 10), we assumed a 1:1 hatchling sex ratio for the proportion of female hatchlings (*FP*; μ: 0.50, σ: 0.040). Finally, the prebreeding census structure used in our matrix model required that hatchling survival (φ_H) also be included in the fecundity term (Eq. 1), rather than treating hatchlings as a separate stage class (Caswell 2011, p.25). No peer-reviewed estimates of annual AST hatchling survival exist, therefore we used 0.15 (σ: 0.029), which was used in Folt et al. (2016, p. 25), and is based on φ_J estimates of related turtle species (e.g., *C. serpentina*; Congdon et al. 1994, p. 399, Heppell 1998, p. 370 and references therein).

Based on the recommendations of Kendall et al. (2019, p. 33), our resulting matrix model contained extensive structural differences compared to the model published in Folt et al. (2016, p. 24, i.e., the original basis for our model), which we detail here. For the juvenile transition term (γ_J), the Folt et al. (2016, p. 25) model used a simple *1/median duration in the juvenile stage* to approximate the probability of transition between juvenile to adult, which is a common practice in population modeling but that approximation assumes the population is in a stable age distribution, which is not often the case. Moreover, the median juvenile duration term (denominator) in the Folt model was misspecified as 17, which reflects the median age at maturity, rather than the median duration (16), due to the AST's first year of life as a hatchling (neonate) with a different survival rate (φ_H). The Folt model omitted survival (φ_J) from the juvenile growth matrix element (G_J), which assumes a different timestep process than our model used and so we modified that parameter in our model according to the recommendations from Kendall et al. (2019, p. 36). The postbreeding census structure used by Folt et al. (2016, p. 24), requires that adult female survival be included in the adult fecundity formula (Caswell 2001, p. 25), though it was not used in the Folt model. Similarly, the postbreeding structure also requires a juvenile fecundity term be included as a matrix element, to include individuals that transition from the juvenile to adult stage within the timestep (Caswell 2001, p. 25), though Folt et al. (2016, p. 24) set juvenile fecundity to zero. Our model used a prebreeding census structure, in which the final two points are not applicable.

The misspecifications in the Folt model described above are expected to produce opposing biases on the asymptotic growth rate (λ). For example, overestimating duration in the juvenile stage and omitting juvenile fecundity would have biased λ low, whereas omitting juvenile survival from the juvenile growth element and omitting adult survival from the adult fecundity element would have biased λ high. However, the cumulative changes to the baseline Folt et al. (2016, p. 24) model required for a correct specification change the population from stable or increasing by up to 3% annually ($\lambda = 1.03$) as reported in Folt et al. (2015, p. 27) to decreasing by up to 3% annually ($\lambda = 0.97$). Lastly, upon reviewing the code used in Folt et al. (2016; B. Folt, pers. communication) we found an additional error that may have artificially inflated the precision of λ in the stochastic simulation. The function used to generate the lognormal distribution shape and scale parameters for the mean duration in the juvenile stage and clutch size was misspecified, so that the resulting distributions generated draws that underestimated both the intended mean and standard deviations. However, the elasticity analysis results in Folt et al. (2016, p. 28), which were consistent with expected patterns for long-lived species (Stearns 1992, entire), indicate that λ was relatively inelastic to the matrix elements that contained the affected parameters. Though the effects of the lognormal misspecification were minor, the type of error is expected to produced opposing biases on the λ value, and systematically underestimate the standard deviation (i.e., inflate the precision).

Stochastic simulation and parametric uncertainty. — We used the projection matrix $\mathbf{A}_{\boldsymbol{u}}$ in a stochastic simulation framework that contained two nested loops: an inner temporal loop that specified the number of timesteps to project forward (*n*=50 years), and an outer simulation loop that specified the number iterations in which to replicate the temporal loop (*n*=500). Given the paucity of AST demographic parameter estimates in the literature, we incorporated parametric uncertainty into our modeling framework using the methods described by McGowan et al. (2011, p. 1401). Parametric uncertainty, or sampling variance (σ_S^2), reflects the lack of perfect knowledge of the parameter's true value due to population sampling, whereas process (temporal) variance (σ_P^2) is the fluctuation in demographic parameters attributed to demographic or environmental stochasticity (Williams et al. 2002, p. 219, McGowan et al. 2011, p. 1401). No AST study to date has partitioned parameter variance in to sampling and process variance (Morris and Doak 2002, p. 348); therefore parametric uncertainty levels in AST population dynamics remain largely unknown.

The standard deviations (σ) for each of the demographic parameters described in the previous section were used to reflect parametric uncertainty (sampling variation; i.e., $\sigma = \sigma_S$) in the model. For each parameter (except *CS*), we used an iterative approach to identify σ_S^2 and σ_T^2 values that partitioned the total variance (i.e., $\sigma_T^2 = \sigma_S^2 + \sigma_P^2$) along a 2:3 ratio (i.e., 66% of the total variance was assigned to the sampling variance) and produced an average coefficient of variation (*CV*) ≈0.15 for σ_T across all parameters. Specifically we manipulated the *CV*s, which were common across all parameters (*p*), for each of the variance components:

$$\sigma_{Sp} = \sqrt{\mu_p \times (1 - \mu_p)} \times CV_S$$

$$\sigma_{Pp} = \sqrt{\mu_{Sp} \times (1 - \mu_{Sp})} \times CV_P \qquad \text{(Eq. 2)}$$

in which $\sigma_{S,p}$ is a function of a mean estimate of parameter *p* (μ_p; i.e., mean values in Table E1) and the sampling standard deviation's coefficient of variation (CV_S), whereas $\sigma_{P,p}$ is a function of $\sigma_{S,p}$ and the process standard deviation's coefficient of variation (CV_P). In both formulas, *CV* is the percentage of a theoretical maximum variation of a mean estimate for parameter *p* (μ_p); *CV* was held constant across all parameters (*p*), but differed between sampling and process variances. Our iterative process identified 0.08 and 0.002 as the highest possible values for CV_S and CV_P (respectively) that met our criteria, producing a CV_T of 0.117, when averaged across all parameters. Though some of the demographic parameters we used to calculate the $\mathbf{A}_{\boldsymbol{u}}$ matrix elements had existing estimates of σ_T^2 reported in the literature, we opted to generate σ_S^2 and σ_P^2 variance components that adhered to the criteria above to ensure model stability (i.e., avoid sampling negative values from probability distributions) and to treat parameters in a consistent manner. It is a common practice in simulation modeling to apply a coefficient of variation function when empirical estimates of variance are not available. The above formulas are only suitable for proportional parameters, therefore we implemented the desired variance partitioning ratio for clutch size (*CS*) by setting σ_S and σ_P to 10 and 5, respectively. Our decision to partition σ_T^2 along a 2:3 ratio for σ_S^2 and σ_P^2 explicitly assumed that there is greater uncertainty in the true mean parameter value (i.e., parametric uncertainty) rather than the amount of annual variation, which is more conservative, given the dearth of AST demographic parameter estimates.

Following the framework described in McGowan et al. (2011, p. 1402), we used μ and σ_S to generate distributions of the overall mean and variance for each parameter. For the overall mean, we used beta distributions for all survival rates (φ_H, φ_J, φ_A, φ_N), the proportion of juveniles transitioning to adults (γ_J), *BP*, *NSC*, and *FP*— i.e., proportional parameters (μ.p)— whereas *CS* (a whole number) was sampled from a lognormal distribution. For each iteration *i* of the simulation loop, a mean ($\mu.p_i$, CS_i) and process standard deviation (σ_{Pi}) were drawn from the parameter's overall mean and variance distributions:

$$\mu.p_i \sim \text{beta}(\alpha, \beta)$$

$$CS_i \sim \text{lognormal}(x_1, x_2)$$

$$\sigma_{Pi} \sim \text{normal}(\sigma_P, \sigma_P \times 0.05)$$

in which α and β are the beta distribution parameters which describe the shape of the distribution bounded between 0 and 1.0, x_1 and x_2 are the shape and scale parameters of lognormal distribution, for the overall mean distributions. We used a normal distribution (above) for the overall variance, which was used to draw iteration-specific process (temporal) variances (σ^2_{Pi}) to determine the amount of temporal variation in each demographic parameter. We verified before beginning our analysis that the error term of the normal distribution was small enough to avoid generating negative values. The variance parameter of the normal distribution (i.e., the variance of the variance) was set to 5% of the theoretical maximum based on the mean sampling process deviation (σ_P), determined in Eq. 2 ($CV_P = 0.002$). Lastly, the iteration-specific means ($\mu.p_i$, CS_i) and standard deviations (σ_{Pi}) were then used to create iteration-specific distributions from which baseline parameter values were then drawn for each timestep *t* within iteration *i*:

$$\mu.p_{i,t} \sim \text{beta}(\alpha_i, \beta_i)$$

$$CS_{i,t} \sim \text{lognormal}(x_{1i}, x_{2i})$$

This hierarchical simulation structure (i.e., using embedded loops to replicate parameter uncertainty and temporal variability) is widely applied in decision support population viability modeling (McGowan et al. 2011, p. 1402; e.g., McGowan et al. 2017, p. 122).

Future Condition Scenarios

Incorporating threat effects. — The expert team identified six potential threats that were likely to reduce stage-specific survival probabilities (summarized in Table E3): commercial fishing bycatch (BYC; $\varphi_H/\varphi_J/\varphi_A$), recreational fishing bycatch (BYR; φ_J/φ_A), hook ingestion (HKI; φ_J/φ_A), legal collection (CLL; $\varphi_H/\varphi_J/\varphi_A$), illegal collection (CLI, i.e., poaching; $\varphi_H/\varphi_J/\varphi_A$), and subsidized nest predators (SNP; φ_N). The baseline φ_N value that we used (0.13; Table E1) was based on a study in which 40 of 46 nests (87%) were depredated by raccoons (*Procyon lotor;* Ewert et al. 2006, p. 67). Therefore, the SNP threat was meant to reflect additional threats to nest survival, such as depredation of emerging neonates from fire ants (*Solenopsis* spp.).

In the expert elicitation questionnaire, we asked the respondents to provide the following threat-related quantities: percent reduction to a survival parameter attributed to each threat and the spatial extent of each threat within their analysis unit(s) of expertise. Thus, reductions to survival parameters attributed to each threat *a* (θ_a) were assumed to be the same across all analysis units, though the spatial extent of each threat (i.e., the proportion of the population exposed to the threat) was structured to vary among analysis units ($\omega_{a,u}$). For example, ingesting a fishing hook would be expected to produce the same percent reduction in φ_A across the entire range, though the probability that an individual AST encounters the threat would vary among analysis units. As such, we determined that CLL violated this assumption, as regulations for legal AST collection differed among states (LDWF 2019a, MFWP 2019, websites). Therefore, we decided to model the effects of CLL as a reduction in juvenile and adult abundances (see Legal Collection section) that varied across analysis units, rather than a reduction to demographic parameters.

We chose to focus on the potential uncertainty regarding the expert-elicited threat-specific parameter p reductions ($\theta_{a,p}$) and the presence or absence of conservation actions to build alternative future condition scenarios. First, we defined three different "threat levels" by adjusting $\theta_{a,p}$ ±25% relative to the summarized expert elicitation responses: (1) decreased threat; (2) expert-elicited; (3) increased threat. Next, we defined conservation action-absent as $\omega_{a,u}$ and present as reducing $\omega_{a,u}$ by 25%. Using a two-factor design, this generated six different scenarios of decreased threat (DE-), expert-elicited (EE-), or increased threat (IN-), with conservation action absent (TH) or present (TH+): DETH, EETH, INTH, DETH+, EETH+, INTH+. For example, the DETH+ scenario reduced both $\theta_{a,p}$ and $\omega_{a,u}$ by 25%, relative to the summarized expert elicitation quantities for $\theta_{a,p}$ and $\omega_{a,u}$. The only exception to this structure is SNP, in which the expert-elicited $\theta_{SNP,p}$ and $\omega_{SNP,u}$. values were used for all scenarios. We chose to hold the SNP spatial extent ($\omega_{SNP,u}$) constant between the conservation action absent (TH) and present (TH+) based on the established difficulties of controlling fire ant populations to reduce nest depredation. Further, only the means for $\theta_{a,p}$ and $\omega_{a,u}$, and not the standard deviations, were adjusted across the different scenarios.

We then used the means and standard deviations for $\theta_{a,p}$ and $\omega_{a,u}$ to create beta distributions specific to each scenario s within the stochastic simulation framework, in which a different value of $\theta_{p,a,s,i,t}$ and $\omega_{a,u,s,i,t}$ was drawn for each simulation i and timestep t:

$$\theta_{p,a,s,i,t} \sim \text{beta}(\alpha_{a,p,s}, \beta_{a,p,s})$$

$$\omega_{a,u,s,i,t} \sim \text{beta}(\alpha_{a,u,s}, \beta_{a,u,s})$$

Threat-weighted survival estimates. — To reflect spatial heterogeneity in threat occurrence and overlap within each analysis unit, we calculated a weighted average of each survival parameter, based on the probable occurrence and overlap of all possible threat combinations. For each analysis unit and survival parameter combination, the total number of threat combinations is equal to two raised to the power of the number of threats within the analysis unit that affect the survival parameter. For example, SNP and CLI are the only threats that affect φ_N (Table E3), and both occur in the Alabama analysis unit (Table E4). Therefore, φ_N in the Alabama analysis unit has four possible threat combination-specific c survival values ($\varphi_{N,\text{Alabama},c}$): (1) SNP only; (2) CLI only; (3) SNP and CLI; (4) no threats.

Survival for each threat combination c follows the general form:

$$\phi_{p,u,c,s,i,t} = \phi_{p,u,s,i,t} - \left(\phi_{p,u,s,i,t} \times \sum \theta_{p,a,s,i,t}\right) \qquad \text{(Eq. 3)}$$

in which the baseline survival parameter p for analysis unit u in iteration i at timestep t is reduced by the sum of the threat-specific a survival reductions (θ), which are expressed as a percent reduction to survival (Table E3). For combinations in which no threats occur (e.g., c =4 in the above example), θ is set to zero, meaning that the baseline survival probability drawn for survival parameter p in analysis unit u in scenario s iteration i at timestep t is used.

After a survival estimate for each threat combination was calculated, we computed a weighted average of the survivals ($\varphi'_{p,u,s,i,t}$), that was weighted according to the probability of the specific threat combination c occurring ($\delta_{p,u,c,s,i,t}$). We treated each threat that could potentially occur as an independent trial in which the threat was either present with probability ($\omega_{a,u,s,i,t}$) or absent ($1 - \omega_{a,u,s,i,t}$), and then multiplied the threat outcomes (presence or absence) together to calculate the threat combination probability. Extending the previous example for φ_N in the Alabama analysis unit, the CLI only (#2) combination probability would be calculated as follows, using the spatial extent values in Table E4:

$$\delta_{CLI\ only} = \omega_{CLI} \times (1 - \omega_{SNP}) = 0.758 \times (1 - 0.902) = 0.074$$

All threat combinations must sum to one, meaning that in the example above, the survival value associated with the CLI only scenario will have a relatively small influence on the overall weighted nest survival estimate (φ'_N), due to the low threat combination probability value (0.074). Thus, for *c* total threat combinations, the weighted average of survival parameter (φ') *p* in analysis unit *u* in scenario *s* iteration *i* in year *t* is given by:

$$\phi'_{p,u,s,i,t} = \sum_1^c \delta_{c,p,u,s,i,t} \times \phi_{p,u,c,s,i,t}) \qquad \text{(Eq. 4)}$$

using the threat combination specific survival estimates derived in Eq. 3. Finally, the weighted averages of the survival parameters (φ'), as well as the demographic parameters not affected by threats (e.g., γ, *CL*, *BP*) were applied to their respective formulas to populate the projection matrix.

Population Projection

Time Frame. — We selected a 50-year time frame to simulate AST population dynamics because the duration allowed for initial demographic transient dynamics to settle and a population trajectory for each iteration to establish, and reflected a sufficiently-short timescale to remain relevant to decision makers, in the context of environmental conditions and existing threats that we incorporated into the projection model (Table E3). Preliminary simulation modeling indicated that the average time to extinction in our framework was <50 years, meaning that extending the time frame would not have likely influenced population viability assessment (PVA) metrics (described in Population Viability Assessment section), such as time to quasi-extirpation. In other words, the number of time steps in our simulation framework was sufficiently large to avoid underestimating extirpation risk, as determined by the PVA metrics. From a cost-benefit perspective, expanding the number of time steps (>50 years) would have come at a computational cost (longer run time), for little benefit because the same number of iterations would be expected to go extinct compared to our framework that used a 50-year time frame.

Initial abundance and stage distribution. — During the expert elicitation process, we asked all participants to provide an estimate of total AST population size within their analysis unit(s) of expertise, and to clarify which sex or age classes (hatchlings, juveniles, adults) their estimate included. We then combined the responses across experts according to the quality control criteria described earlier. However, with the exception of analysis unit eight, the expert-elicited abundance estimates included hatchlings, which were not included as a stage class in our matrix model due to the prebreeding census structure. For the purposes of initializing abundance in units 1–7, we re-formulated our projection model to reflect a prebreeding census structure with three stages (hatchlings, juveniles, adults) and multiplied the proportion of hatchlings at stable stage by the expert elicited total abundance estimates, to obtain the expected initial abundance of juveniles and adults only (IA_u). We initialized the starting population for each analysis unit assuming that the population was in a stable stage distribution ($\mathbf{ssd}_u$), the corresponding eigenvector of the dominant eigenvalue of the projection matrix $\mathbf{A}_u$.

Next, we created a series of stochastic variables to generate stage-specific abundances at time t=1, that were unique to each analysis unit *u*, scenario *s*, and iteration *i* combination. First, we converted IA_u to a Poisson-distributed stochastic variable ($N_{u,s,i}$) that was multiplied by an initial stage distribution ($\mathbf{isd}_{u,s,i}$) generated from a Dirichlet distribution to convert $N_{u,s,i}$ back to stage-specific abundances. We parameterized the Dirichlet distribution using the unit-specific stable stage distribution ($\mathbf{ssd}_{u}$) multiplied by 10, to reduce the amount of variation.

$$N_{u,s,i} \sim \text{Poisson}(\text{IA}_{u,s,i})$$

$$\boldsymbol{isd}_{u,s,i} \sim Dirichlet(10 \times \overrightarrow{\boldsymbol{ssd}_{u}})$$

All of the expert-elicited initial abundance estimates included both males and females, whereas our model was females-only. Therefore, we generated two samples of initial stage-specific sex ratios ($\mathbf{isr}_{j,u,s,i}$), one for each stage class j, from a normal distribution. We specified the distribution with a mean of 0.45 based on observed sex ratios in juveniles and adults from a reference population (Folt et al. 2016, p. 26) and a standard deviation that was assumed to be 20% of the theoretical maximum.

$$\mathbf{isr}_{j,u,s,i} \sim \text{normal}(0.45, 0.45 \times (1-0.45) \times 0.20)$$

$$\begin{bmatrix} n_{J,u,s,i,1} \\ n_{A,u,s,i,1} \end{bmatrix} = N_{u,s,i} \times \boldsymbol{isd}_{u,s,i} \times \boldsymbol{isr}_{us,i}$$

Finally, we multiplied the three stochastic quantities to generate stage-specific initial abundances (t=1) for all analysis unit, scenario, and iteration combinations ($n_{j,u,s,i,1}$).

Our modeling framework incorporated three additional effects believed to influence AST demography: habitat loss, legal collection, and head start releases. Unlike the threat-specific parameter reductions, these effects were held consistent across all future condition scenarios, though they were subject to stochastic variation among iterations and timesteps. The first two effects were applied directly to the stage-specific abundance vector $\mathbf{n}_{u,s,i,t}$, before it was multiplied by the projection matrix to project to the next timestep, whereas the effect of habitat loss was incorporated into the adult fecundity element in the projection matrix, but was contingent upon total abundance for t>1.

Legal Collection. — The expert-elicitation process generated stage-specific reductions in survivals attributed to legal collection that were not specific to individual analysis units (Table E3). After reviewing the responses from experts, we suspected that some of the respondents may have interpreted the question at the analysis unit-level, rather than range-wide. Therefore, based on the potential inconsistencies, we decided to simulate the effects of legal collection on AST populations by an annual deduction of abundance within each unit so that we could better capture dynamics among analysis units. Currently, only Louisiana and Mississippi allow legal collection of AST. We did not incorporate the effects of the Mississippi harvest program because carapace length (>61 cm) restrictions functionally exclude females (MFWP 2019, website), which generally do not exceed 50 cm (Folt et al. 2016, p. 24). Whereas in Louisiana, current regulations allow for any angler with a freshwater fishing license to take one AST of any size per day (LDWF 2019b, website). Within our modeling framework, we restricted the effects of legal collection to the two remaining analysis units that overlapped geographically with Louisiana: Southern Mississippi – East and Alabama.

No data are available from LDWF or other sources regarding legal AST collection, therefore, we relied upon annual freshwater fishing license and specialty permit sales for wire traps and hoop nets (often used to catch turtles) from 2012–2017 as an index of take (LDWF 2019b, website). We used several stochastic variables to generate an initial random number of AST to be collected each year (ANG), that was further refined based on population size and composition. First, we modeled the annual number of freshwater fishing licenses (FL) as a normally distributed variable, according to the mean and standard deviation of the LDWF data:

$$FL_{u,s,i,t} \sim \text{normal}(392771, 28970)$$

Next, we derived the proportion of individuals (anglers) that purchased wire trap or hoop net permits, relative to freshwater fishing licenses based on the average across years: 0.0094 ± 0.005. We rounded the annual proportion anglers that purchased permits for either trap type (*PT*: 0.010 ± 0.014) and modeled it as a beta distributed stochastic variable. We increased *PT* to account for the fact that some anglers may take more than one AST per year, and that anglers are permitted to deploy up to five traps of a single type at a time. We also scaled the amount of AST to be collected based on the proportion of Louisiana that overlapped with each analysis unit—Southern Mississippi – East (0.695) and Alabama (0.019)— and multiplied the three quantities:

$$ANG_{u,s,i,t} = FL_{u,s,i,t} \times PT_{u,s,i,t} \times OV_u \times 0.50 \qquad \text{(Eq.5)}$$

The OV_u adjustment, roughly, assumed that freshwater fishing license sales have an even spatial distribution in Louisiana. Lastly, we added a sex ratio adjustment (0.50) to account for the fact that not all anglers will catch females. Though this assumption is likely violated, attempting to spatially refine this quantity is likely of limited utility, as individuals may fish or set traps in parishes outside of where they bought their license.

The random number of AST to be legally collected at each timestep within all analysis unit and scenario combinations ($ANG_{u,s,i,t}$) was generated outside of the model's looping structure. Within the model itself, we generated a stage-specific legal collection vector **cll** that was informed by other parameters. First, we limited the legal take of AST based on the proportion of the analysis unit that overlapped with Louisiana (*LA*): Southern Mississippi – East (0.316) and Alabama (0.013). Note that the purpose of *OV* in Eq. 5 was to limit the randomly generated AST collection based on fishing license sales in Louisiana, whereas the purpose of *LA* was to limit the proportion of the population within the analysis unit exposed to legal collection. Like *OV*, the *LA* adjustment assumed that AST were evenly distributed in space within the analysis unit. While this assumption is likely violated, it is difficult to refine the *LA* values in the absence of a detailed GIS analysis that could estimate AST densities within each of the analysis units based on habitat types.

After reducing the randomly generated AST harvest based on *HT* and LA_u, we further scaled the annual take based on the proportion of total AST (*N*, i.e., both stage classes) currently in the analysis unit *u* at time *t* relative to the population size in iteration *i* at *t*=1:

$$\frac{N_{u,s,i,t}}{N_{u,s,i,1}}$$

The proportion of AST relative to starting population size adjusted for "catchability", in that the number of AST captured is expected to positively vary with population size. Finally, to produce stage-specific legal collection quantities ($c_{j,u,s,i,t}$) within the **cll** vector, we assumed that stage classes were harvested (approximately) in proportion to their occurrence in the population, denoted by the vector on the far right of the below equation:

$$\begin{bmatrix} c_{J,u,s,i,t} \\ c_{A,u,s,i,t} \end{bmatrix} = ANG_{u,s,i,t} \times LA_u \times \frac{N_{u,s,i,t}}{N_{u,s,i,1}} \times \begin{bmatrix} n_{J,u,s,i,t} + (0.02 \times N_{u,s,i,t}) \big/ N_{u,s,i,t} \\ n_{A,u,s,i,t} - (0.02 \times N_{u,s,i,t}) \big/ N_{u,s,i,t} \end{bmatrix}$$

We increased collection of juveniles by 2% (relative to their proportion in the population) and correspondingly, reduced harvest of adults by the same amount, to account for potential harvest of hatchlings. Due to the pre-breeding census structure, the model does not produce hatchling abundance estimates in which a legal collection function could be applied. Therefore, we opted to instead account for potential collection of hatchlings by increasing the relative proportion of juvenile collection.

Head Start Releases. — Several states within the AST's distribution have initiated head start release programs, in which AST are raised for several years in captivity and then released into the wild population as juveniles (Dreslik et al. 2017, p. 13). Similarly, states also opportunistically release adult AST confiscated from illegal activities (e.g., poaching) into wild populations, when available. We included the juvenile and adult releases within the model, though only for the first ten timesteps within an iteration, to avoid having AST population persistence be contingent on head start activities (i.e., conservation-dependent). We parameterized the releases in the model based on statistics from Illinois described in Dreslik et al. (2017; juveniles: ~30 individuals/year, adults: ~12, p. 13). The mean number of releases did not vary among analysis units or scenarios, but because of the uncertainty and variability in the simulations, the specific value drawn for each year in each unit in each replicate varied. Specifically, for the first ten timesteps (t<11) of each iteration, the number of released juveniles ($h_{J,i,t}$) and adults ($h_{A,i,t}$) were drawn from Poisson distributions and placed in the **hsd** vector:

$$h_J \sim \text{Poisson}(30)$$

$$h_J \sim \text{Poisson}(12)$$

$$\boldsymbol{hsd}_{u,s,i,t} = \begin{bmatrix} h_{J,u,s,i,t} \\ h_{A,u,s,i,t} \end{bmatrix}$$

whereas $\mathbf{hsd}_{u,s,i,>10}$ contained all 0s beyond the first ten timesteps after the releases ceased. For the baseline model, we ran two scenarios— one that included releases of adults and juveniles and one in which no releases occurred.

Given the uncertainty regarding the number of harvested individuals, we ran a "no legal collection scenario for the two affected analysis units for comparative purposes. All results reflect the presence of legal collections, unless otherwise noted.

Habitat Loss Function. — We asked the expert team to list habitat loss mechanisms within their analysis unit(s) of expertise. After adjusting for spelling, grammar, and linguistic differences among responses (e.g., "desnagging" and "removal of large woody debris" were two answers that reflected the same mechanism), we summarized the number of unique habitat loss mechanisms within each analysis unit and calculated the mean across experts. We imposed a population ceiling (i.e., carrying capacity) that was annually reduced by a habitat loss rate (κ_u), which equaled the mean number of unique threats in the unit, divided by 100. The initial (i.e., t=1) population ceiling ($PC_{u,1}$) was determined based on the summarized expert elicitation values for the maximum number of AST currently within the analysis unit + 25%, after adjusting for sex ratios and hatchlings (as described in the previous section). Thus, the population ceiling ($PC_{u,t}$) for analysis unit u in year t was calculated deterministically:

$$PC_{u,t} = PC_{u,1} \times (1 - \kappa_u)^t \qquad \text{(Eq. 6)}$$

and was not subject to stochastic variation across simulation iterations. To incorporate the effects of habitat loss on AST demography within the model, we included a function that set adult fecundity (F_A) to zero in the projection matrix if AST total abundance (Juveniles and Adults) in year t if the AST total abundance in year t−1 exceeded PC_t.

The population ceiling-contingent adult fecundity value was the last required step to finalize the projection matrix $\mathbf{A}_{u,s,i,s,t}$, which was then multiplied by the stage-specific abundance vector, after it was adjusted for additions through head starts and adult releases ($h_{j,u,s,i,t}$), and reductions through legal collections ($c_{j,u,s,i,t}$):

$$\begin{bmatrix} n_{Ju,s,i,t+1} \\ n_{Au,s,i,t+1} \end{bmatrix} = \left(\begin{bmatrix} n_{Ju,s,i,t} \\ n_{Au,s,i,t} \end{bmatrix} + \begin{bmatrix} h_{Ju,s,i,t} \\ h_{Au,s,i,t} \end{bmatrix} - \begin{bmatrix} c_{Ju,s,i,t} \\ c_{Au,s,i,t} \end{bmatrix} \right) \times \boldsymbol{A}_{u,s,i,t}$$

Finally, our temporal looping structure contained 50 timesteps, meaning that our analysis generated stage-specific abundances for 51 years, as we stored both the initial abundance values (parameterized by expert elicitation data) and the outcome of the final projection.

Baseline model.— For comparative purposes, we simulated AST population dynamics in the absence of threats to reflect baseline (i.e., idealized, reference) conditions, in which the added threats (Tables E2, E3) we included in the future condition scenarios were absent. The baseline model was meant to reflect population dynamics in protected or isolated areas, like the Spring Creek population studied by Folt et al. (2016, p. 23). We used the demographic parameter means and standard deviations listed in Table E2 to populate the projection matrix, as well as the initial abundances provided by experts for each unit. We ran two versions of the baseline model, one that included adult and juvenile releases and one that did not, and neither included the habitat loss function.

Sensitivity Analyses

To identify which model inputs had the largest influence on the model results, we conducted two forms of sensitivity analysis. First, we used the 'eigen.analysis' function in the *popbio* package (Stubben et al. 2016, p. 16) to generate asymptotic population growth rate (λ), elasticities, and stable stage distributions from each of the transition matrices ($\mathbf{A}_{u,s,i,t}$). Elasticity essentially measures the percent change in lambda, or any other output metric, relative to percent changes in the input demographic rates (Caswell, 2001), meaning that proportional variables (e.g., survival) and continuous variables (fecundity) can be directly compared to one another. We performed the same procedure on the baseline deterministic transition matrices for units 1–7 and 8 ($\mathbf{D}_{1\text{-}7}$ and $\mathbf{D}_8$, respectively), that used the baseline demographic parameter estimates in Table E1 to parameterize the matrix elements. Hence, the sensitivity analysis for the baseline model only evaluated a single matrix for each analysis unit group ($\mathbf{D}_{1\text{-}7}$ and $\mathbf{D}_8$) that contained the mean values, whereas up to 500 (*n* simulations) were evaluated for each of the analysis unit and scenario combinations.

We then conducted an additional sensitivity analysis of the model outputs using a regression-based approach to link realized lambda ($N_{u,s,i,t+1} \div N_{u,s,i,t}$) to the stochastically generated threat levels and demographic rates each year. The regression analysis treats the realized lambda as the dependent variable and the stochastically drawn annual values of survival and each threat as independent variable in regression models. The effect and strength of each parameter and threat can be assessed and compared using the regression slope estimates and model selection analysis to identify the most influential effects on population growth.

Population Viability Assessment

We derived a series of summary statistics to evaluate AST population trends and identify potential variation among analysis units and alternative scenarios. Here we define an extirpation event as the total population (juveniles + adults) declining to zero individuals, whereas a decline to less than 5% of the starting population size (t=1) was considered quasi-extirpation. We selected this threshold because it reflected a result of a catastrophic population decline and was similar to values used for previous Species Status Assessments (e.g., 2% and 4% for the Sonoran desert tortoise, USFWS, p. 86). For each analysis unit and scenario combination, we estimated extirpation and quasi-extirpation probabilities (p_{EX}, p_{QX}) by determining proportion of iterations in which the population reached those thresholds. Within the iterations in which the population reached extirpation or quasi-extirpation, we estimated the mean number of years until the population reached the specified criteria (t_{EX}, t_{QX}). Additionally, We performed all analyses in the R statistical program (v.3.5.3, R Core Development Team 2019, software).

RESULTS

Threat Summaries

Summaries of the expert-elicited threat-specific reductions to demographic parameters ($\theta_{p,a}$) and their spatial extents within the analysis units ($\omega_{a,u}$) are summarized in Tables E3 and E4, respectively. Among the threats used in the model (CLL excluded), the effect of SNP on φ_N was the largest overall reduction, followed by CLI on φ_A (Table E3). SNP also generally had the largest spatial extent within the analysis units, followed by CLI (Table E4).

Eigen Analyses and Model Sensitivity

Asymptotic population growth rate.— The asymptotic population growth rates (λ) derived from the projection matrices (Table E6, Figs. E6–9) were less than one, indicating a population decline, for all analysis units and future conditions scenarios. Mean λ for all of the analysis unit and future condition scenario combinations ranged from 0.749 ± 0.038 (SD) for the INTH scenario in ALAB, to 0.899 ± 0.039 for NOME's DETH+ scenario, and averaged 0.86 ± 0.07 across all combinations. Averaging across scenarios within analysis units, λ was highest for NOME (0.952 ± 0.03), followed by APAL (0.856 ± 0.05), SOME (0.830 ± 0.03), and ALAB (0.804 ± 0.04).

These results are consistent with the population declines we detected in the stochastic simulation (Figs. E2–5). We note, however, that the baseline scenario simulations showed mixed evidence of population growth for the non-NOME units— though the baseline population simulations indicated a growing population (Figure E12) the λ derived from the Eigen analysis indicated a population decline (0.988 ± 0.038 SD; Table E6) though the standard deviation overlapped 1 indicating some uncertainty in the trajectory. In contrast, all metrices of population growth indicated a decline in the NOME unit based on the stochastic simulations (Figs. E5, E12) and λ values (0.963 ± 0.030 SD; Table E6) for both the future conditions and baseline scenarios. We note that the baseline mean λ values appearing at the bottom of Table E6 were computed by pooling across the two baseline condition scenarios (releases of juveniles and adults present or absent) within the two groups (non-NOME vs. NOME). The asymptotic lambda, which is based on matrix formulation, is not expected to change among the baseline scenarios because releases were directly added to abundance and did not influence the demographic parameters within the projection matrix.

Sensitivity Analyses.— Life history theory predicts that changes in adult female survival are likely to generate the greatest proportional change in the asymptotic growth rate (λ) of long-lived species (Stearns 1992, entire), like AST. This pattern is reflected in the elasticities of the deterministic matrices ($\mathbf{D}_{\text{SOME, ALAB, APAL}}$ and $\mathbf{D}_{\text{NOME}}$) and NOME (Table E7), in which P_A (adult survival) consistently ranked the highest, followed by P_J (juvenile retention), and identical values for G_J and F_A (juvenile growth and adult fecundity, respectively). In contrast, λ was consistently the most elastic to P_J, followed by P_A, and G_J and F_A elasticities being equal for the SOME and ALAB analysis units, whereas the elasticity patterns observed for APAL were intermediate to those of SOME/ALAB and NOME (Table E7). In general, as survival rates were reduced in our analysis framework due to the increasing threat level (i.e., $\varphi_{DE-} < \varphi_{EE-} < \varphi_{IN-}$), the elasticity of P_J, G_J, and F_A increased, while P_A elasticity decreased (Table E7). This general trend explains the increasing elasticity of P_A from SOME/ALAB, APAL, and NOME due to adult survival also following an increasing pattern ($\varphi_{SOME/ALAB} < \varphi_{APAL} < \varphi_{NOME}$).

This lambda-regression sensitivity analysis concluded that the illegal collection has the greatest effect on population growth, primarily through its reduction to adult survival, as the model containing that term had all of the model weight (Table E10), followed by the spatial extent of illegal collection, and the effects of hook ingestion and recreational bycatch on adult survival (Table E10). Each of these threats are modeled as percent reductions in adult and juvenile survival thus the results of this regression analysis match the eigen elasticity analysis and our expectations for this analysis. Experts believed that illegal collection caused up to a 19.5% reduction in survival (Table E3) and that it affected a minimum of 30% of the population in all regions except Northern Mississippi–East (Table E4). Given the magnitude and spatial extent of this threat, it is not surprising that it has the greatest effect on realized lambda in our model.

Stable stage distribution.— The stable stage distribution (SSD) of the projection matrix reflects the proportion of individuals within each stage class when the realized population growth rate is equal to the asymptotic growth rate. In the deterministic matrices (bottom rows in Table E8), juveniles comprised a larger proportion of the population than juveniles in the SOME, ALAB, and APAL analysis units, whereas the two stage classes were nearly even in NOME (Table E8). The SSD patterns we detected mirrored those of the elasticity analysis in that juveniles comprised a majority of the population in SOME, ALAB, and APAL, whereas adults comprised the majority in NOME (Table E8). In general, the proportion of juveniles in the SSD was positively associated with the increasing threat level (Table E8).

AST Population Viability

The baseline models suggested that in the absence of threats, AST populations were expected to increase in all analysis units, with the exception of NOME (Figure E12). However, we note that the baseline population trajectories for the non-NOME analysis units (SOME, ALAB, APAL) contrast with their corresponding mean asymptotic growth rate. Though the mean asymptotic growth rate indicated a population decline, the standard deviation overlapped one, indicating some uncertainty (Table E6); this contrast and uncertainty is further discussed in the Synthesis section. For the NOME analysis unit, all baseline scenarios indicated a population decline based on the mean total abundance (Figure E12) and asymptotic growth rates (Table E6). In the baseline scenario that included releases, the NOME population increased for the first ten years, and then declined rapidly after releases halted, whereas the no releases scenario declined slowly over time.

In contrast, when threats were introduced to the simulation framework (i.e., the future conditions scenarios), the results showed a vastly different pattern than the baseline scenario. All analysis unit and scenario combinations showed steep declines in abundance (Figs. E2–5). At the stage class level, all units except NOME followed a common pattern in which juveniles initially comprised the majority of the population, but then decline and are eventually outnumbered by adults. This pattern is likely driven by juveniles recruiting into the adult stage class and insufficient adult fecundity values to replace the recruited juveniles. In both deterministic matrices, $\mathbf{D}_{\text{SOME, ALAB, APAL}}$ and $\mathbf{D}_{\text{NOME}}$, each adult female produced 0.23 juvenile females per year (F_{A} in Table E5), meaning that at least four nesting attempts would be needed for replacement. After incorporating the effects of threats on the demographic parameters, all of the mean matrix element values were reduced compared to their deterministic counterparts. The majority of matrix element values were relatively similar among SOME, ALAB, and APAL, as they were derived from the same baseline demographic parameter values, compared with NOME (Table E1). The P_{J} and G_{J} projection matrix elements were generally higher for SOME, ALAB, and APAL, compared to NOME (Table E5), due to the lower baseline juvenile survival value used for NOME (Table E1). However, adult survival (P_{A}) was higher in the NOME unit (0.95 ± 0.01 SD, all scenarios) compared to other three units (0.76 ± 0.01), despite a shared baseline survival rate (Table E1), which is likely driven by the near-absence of BYR and CLI threats in NOME (Table E4).

Extirpation and Quasi-extirpation Probability. — In the main future condition scenario analysis, none of the analysis units exhibited extirpation probabilities (p_{EX}) greater than 0.45 at the decreased threat level (Table E9). Averaging across scenarios within analysis units, p_{EX} was highest for SOME (0.63 ± 0.37), followed by ALAB (0.46 ± 0.43 SD), APAL (0.14 ± 0.26), and lowest for NOME (0.00). Of all analysis unit and scenario combinations, p_{EX} was the highest for ALAB-INTH (1.0) and ≤0.002 for DETH+ ALAB, APAL, and NOME. Among the eight instances in which conservation action was absent (TH columns in Table E9) and p_{EX}>0.01, the average reduction in p_{EX} for the conjugate conservation action scenario was 0.37 ± 0.23. However, among the analysis unit and scenario combinations in which p_{EX}>0.01, the number of years to reach extirpation (t_{EX}) was relatively large with an overall mean of 46.18 ± 3.49 years and ranged from 38.07 ± 3.37 years (SOME-INTH) to 49.45 ± 1.92 (SOME-DETH+, Table E9).

Quasi-extirpation probabilities (p_{QX}) were consistently high (approximately 1.0) across all analysis unit and scenario combinations, with the exception of NOME (Table E9). In non-NOME units, p_{QX} ranged was equal to 1.0 for all analysis unit and scenario combinations with the exception of decreased threats in APAL. Time to quasi-extirpation (t_{QX}) in all non-NOME units averaged 22.28 ± 7.60 (SD) years across all scenarios, whereas t_{QX} ranged from 12.11 ± 1.35 in ALAB-INTH to 33.11 ± 6.09 years in APAL-DETH. Within the NOME unit, multiple measures of extirpation risk (e.g., p_{QX}) did not for the predicted pattern of extirpation or quasi-extirpation being least likely in the DETH+ scenario and highest in INTH. For example, p_{QX} for NOME was lowest for EETH (0.016) and highest for DETH (0.038). This pattern can be explained by examining Table E4, as the threats with reduced spatial extent in conservation action scenarios that occur in NOME (BYR and CLI) have extremely small spatial extents.

In our separate analysis evaluating the effects of legal harvest, we found that while removing legal harvest drastically lowered the probability of extirpation (p_{EX}) in SOME, the remaining metrics were relatively unchanged (Table E11). For example, the time to extirpation or quasi-extirpation was only reduced by 2–3 years, and the probability of quasi-extirpation averaged one across all six scenarios regardless of whether legal collection was present or not (Table E11).

Synthesis

Drivers of AST demographics. — The sensitivity analyses showed a consistent pattern suggesting that population growth is most sensitive to factors that influence adult survival, which is expected for a long-lived species like AST (Stearns 1992, entire). The elasticity analysis indicated that under baseline conditions ("Deterministic" entries in Tables E5–8), conservation interventions to increase adult survival (contained in the P_A matrix element; Table E1) are likely to have the greatest proportional impact on AST population growth (Table E7). Though all six of the future condition scenarios reduced the elasticity of P_A relative to the deterministic matrix (Table E7), P_A remained the most elastic parameter in the majority of analysis unit and future scenario combinations. When adult survival was drastically reduced in the SOME and ALAB units (Figure E10), the elasticity of P_J exceeded (NOME) or was approximately equal to that of P_A (APAL), indicating that conservation interventions to increase juvenile survival, as opposed to adults, may be more effective in population recovery if threat levels are relatively high.

Similarly, the lambda regression approach indicated that the illegal collection impacts on adult survival and its spatial extent has the greatest effect on population growth in our model followed by hook ingestion impacts on adult survival and recreational fishing bycatch impacts on adult survival (Table E10). Experts believed that illegal collection caused up to a 19.5% reduction in survival (Table E3) and that it affected a minimum of 30% of the population in all regions except Northern Mississippi–East (Table E4). Given the magnitude and spatial extent of this threat, it is not surprising that it has the greatest effect on realized lambda in the model.

Within the stochastic simulation framework, we simulated conservation actions as a reduction in a threat's spatial extent (ω_a; Table E4). Based on a comparison of survival rates for all stage classes and scenarios (Figure E10), the conservation actions had increasing effectiveness (i.e., difference between circles and triangles for a given threat level in Fig. E10) with stage class (hatchlings, juveniles, adults). The effectiveness of conservation actions positively varied with threat level, particularly for adults, meaning that the largest improvements to adult survival with conservation action were observed at the high threat level (red points in Fig. E10).

Some experts indicated that habitat loss may be a limiting factor for AST. Based on our simulation that included a declining population ceiling to represent habitat loss (bottom row in Table E4), AST population declines outpaced the habitat loss rate (Figure E11). Meaning that, the AST population size never reached the population ceiling to trigger the density dependent response (F_A= 0). In summary, habitat creation is likely to have less of an impact on population growth compared to enacting conservation actions that could increase adult survival.

Model limitations and uncertainties. — Our model was constructed to predict current and future conditions of the alligator snapping turtle within the Southern Mississippi – East, Alabama, Apalachicola, and Northern Mississippi East analysis units. While this modeling framework was constructed with the intention of informing the Endangered Species Act listing decision, all models have potential inferential limitations due to an imperfect knowledge of the system in question. In this particular case, the limited number of *M. temminckiii* demographic studies required the use of data from closely related species (e.g., *Chelydra* spp.) and expert opinion (obtained through remote elicitation). We addressed these sources of uncertainty in multiple ways within the modeling framework using a combination of established techniques (e.g., stochastic iterations, parametric uncertainty) and newly developed methods (e.g., threat-weighted survivals).

Due to a dearth of demographic studies on *M. Temminckii* and closely-related species (e.g., *M. Suwanniensis*), our model relied heavily on the use of expert-elicited quantities, including population sizes, threat-specific parameter reductions and spatial extents, as well as other demographic parameters (Table E1). Moreover, we conducted the elicitation remotely through a series of webinars and emails. The created several disadvantages (compared to an in-person workshop) as the extensive questionnaire (Appendix C) may have reduced the response rate, and the experts may not have had the opportunity to ask the SSA Core Team for clarification regarding the quantities they were asked to report. However, among the experts who provided responses, we had a 100% participation rate when the Core Team needed further clarification from experts on their answers. We also used a weighted approach to combine expert elicited responses for a given quantity, in which responses with a higher degree of confidence had a larger influence on the overall mean. Furthermore, estimates of variance for many elicited parameters were small (Tables E3–4), suggesting that the experts generally agreed with each other, even though they the values were elicited independently from each expert. Lastly, we chose to construct the Future Condition scenarios to address uncertainty in the expert elicitation responses, particularly regarding the threat-specific parameter reductions, which were translated into the three threat levels: decreased, expert-elicited, and increased.

Among the parameters, the legal collection totals represent the greatest uncertainty, which was noted by peer and partner reviewers. Louisiana does not collect data on the number of legally collected AST each year. Our legal collection function (Eq. 5) represents a conservative approximation, which is reflected in the separate analysis. Removing legal harvest had minimal influence on all population viability statistics with the exception of probability of extirpation (p_{EX}) in SOME (Table E11).

We did not use a spatially-explicit model due to a paucity of both knowledge of spatial variation in demographic parameters and abundance, as well as the distribution of threats within the analysis units with sufficient data. Despite the lack of spatial data, our model was able to produce heterogeneity in survival rates (within the same iteration and year) that would be expected in an area in which threats were overlapping and unevenly distributed on the landscape. With the exception of the HKI and BYR threats, the threat-weighted survival approach we used to produce heterogeneity in survival did not make any assumptions about potential spatial correlations among threats, as the probability of a threat to overlap with another threat was based on the proportion of the population each threat affected. In other words, two threats that affected the majority of the AST population would be expected to have extensive overlap. While this probabilistic approach may not fully capture spatial relationships among threats, it is objective, given the knowledge lacking in the distribution of threats.

Validation.— We also acknowledge an ongoing concern raised with regard to the model used herein, is that it does not match the published estimates of population growth for the Folt et al. (2016, entire) model and conflicts with the perceived stability of AST populations from some catch-per-unit-effort studies for this species. As for validating model inputs, for several parameters, especially population threats as noted above had to rely on expert elicitation rather than data analysis or published literature. Steen and Robinson (2017, p. 1336) conclude that an average of between 3% and 36%, (with wide credible intervals that exceeded out elicited values) of snapping turtles had ingested hook, and admit their sampling design likely underestimated hook injection rates. Furthermore, estimates of variance for many elicited parameters were small, suggesting that the experts generally agreed with each other, even though they the values were elicited independently from each expert.

For validating model predictions, the first thing to note is that the Folt et al. (2016, p. 23) paper primarily studied AST in an area with few or no illegal collection, bycatch, or hook ingestion threats. The original formulation of the Folt model had multiple errors in the timing of abundance accounting (pre- vs post- breeding census) and in the juvenile to adult transition parameters (Caswell 2001, Kendall et al., 2019), and mis-specified (under-estimated) the variance for multiple parameters. Correcting those errors changed the prediction form a population that was growing 3% annually to one that was declining 3% annually. The modeling effort used in the SSA further modified the (corrected) Folt baseline model to account for dispersal of juveniles which is not possible to estimate and measure in mark recapture studies. This modification (upward adjustment of the Juvenile survival parameter by 5%; Table E1) restored the threat-free (baseline) population trajectory predictions to apparent stability for all units except Northern Mississippi–East (Figure E12). Dispersal is likely among the juvenile age class, but mark recapture studies cannot account for permanent immigration so reincorporating these factors into the projection model seemed sensible.

As noted earlier, however, we identified a discrepancy in the baseline scenarios between the mean abundance trajectories (Figure E12) and the asymptotic growth rates (Table E6) for the non-NOME analysis units. It is important to consider that asymptotic growth rates are only relevant if the population is in a stable stage distribution. The initial stage distribution in both the baseline and future condition scenario simulations were parameterized based on the expert-elicited values, which did not necessarily reflect the stable stage distribution associated with the demographic parameters used in the projection matrices (Table E8). Given the AST's slow maturity, extensive time could be required for the population to transition to a stable stage distribution that is reflective of the asymptotic growth rate. This apparent disconnect between the realized- and stable-stage distributions can therefore account for the discrepancy between the two metrics. Additionally, the parametric uncertainty structure applied to all the simulations (both baseline and future conditions scenarios) has a tendency to inflate confidence intervals around mean abundance trajectories as an added measure of uncertainty (McGowan et al. 2011, p. 1404). Thus, the very large confidence intervals around the mean abundance estimates in the later time steps are to be expected based on the modeling structure that we applied. Lastly, while asymptotic lambdas are frequently uses as an assessment of population health, wild populations rarely conform to the assumption of a stable stage distribution (Koons et al. 2017, p. 2103), therefore, evidence from field studies are likely a more relevant option for validation of our model.

An additional component of Folt et al. (2016) evaluated population status and trajectories for a population in Arkansas and one on a wildlife refuge in Oklahoma, where several of these threats are present, and the authors predicted rapid declines for those populations based on estimated demographic rates at those sites. For example, they predicted that the population in Oklahoma would be extinct within 15 years (Folt et al. 2016, p. 30) based on the uncorrected model version that overestimates population growth rate (and therefore also overestimates time to extinction). These results in the published literature match fairly well with predicted trajectories for populations exposed to threats in the model. For example, in their simulation modeling, Steen and Robinson (2017, p. 1338) found that hook ingestion alone caused alligator snapping turtle populations that were increasing to reverse the predicted trend and decline by >50% in 30 years. Furthermore, since the completion of our work on the AST SSA report (RTM Version 1.0, October 2019), Ethan Kessler completed a PVA model for AST in southern Illinois (within the Northern Mississippi – East analysis unit) for his dissertation (Kessler, 2020). Radio telemetry was to directly estimate true survival and growth rates for AST populations (and the benefits of head starting and captive release programs). Kessler combined the parameters estimated from his study with productivity values from the peer reviewed literature into a PVA and reported a population growth rate (λ) of 0.95 (Kessler 2020, pg. 126) which is identical to the mean asymptotic population growth rates that we estimated for the Northern Mississippi – East unit across all scenarios (Table E6). Further, Kessler's analysis identified several of the same threats (especially recreational fishing bycatch), that were incorporated into the modeling used in the SSA, as key factors for future abundance and population growth rates. Of note, Kessler reported a catastrophic recreational bycatch incident in which a local resident illegally set a hoopnet and abandoned the device due to a sustained flooding event that limited trap accessibility. The abandoned hoopnet trapped and eventually drowned six adult and subadult alligator snapping turtles, including two individuals with radio transmitters (Kessler, personal communication. Kessler reports that the introduced population exhibits unstable demography and that reintroduction efforts are likely to fail unless bycatch can be reduced (Kessler 2020, pg. 116). It is not possible to fully validate model predictions from any single predictive model, but three independent models with similar results may bolster confidence in model predictions provided in the SSA.

LITERATURE CITED

Allen, E.R., and W.T. Neil. 1950. The alligator snapping turtle *Macrochelys temminckii* in Florida. Ross Allen's Reptile Institute Special Publication 4:1–15.

Bass, A.A. 2007. Habitat use and movements of Alligator Snapping Turtle (*Macrochelys temminckii*) hatchlings. Unpubl. M.Sc. Thesis. University of Louisiana at Monroe. Monroe, Louisiana, USA.

Caswell, H., 2001. Matrix Population Models, Construction, Analysis and Interpretation. Sinauer Associates, Inc. Publishers, Sunderland, MA, USA.

Congdon, J.D., A.E. Dunham, and R.C. van Loben Sels. 1994. Demographics for common snapping turtles (*Chelydra serpentina*): implications for conservation and management of long-lived organisms. American Zoologist 34:397–408.

Dobie, J.L. 1971. Reproduction and growth in the alligator snapping turtle, *Macroclemys temminckii* (Troost). Copeia 1971:645–658.

Dreslik, M.J., J.L. Carr, D.B. Ligon, and E.J. Kessler. 2017. Recovery of the alligator snapping turtle (*Macrochelys temminckii*) in the Mississippi River Valley drainages of southern Illinois, Oklahoma, and Louisiana. Illinois Department of Natural Resources.

Ewert, M.A. 1976. Nests, nesting and aerial basking of *Marcolemys* under natural conditions and comparisons with Chelydra (*Testudines: Chelydridate*). Herpetologica 32:150–156.

Ewert, M.A., and D.R. Jackson. 1994. Nesting ecology of the alligator snapping turtle *Marcoclemys temminckii*, along the Lower Apalachicola River, Florida. Florida Game and Fresh Water Fish Commission, Nongame Wildlife Program Final Report. 45 pp. Tallahassee, FL

Ewert, M.A., D.R. Jackson, and C.E. Nelson. 1994. Patterns of temperature-dependent sex determination in turtles. Journal of Experimental Zoology 270:3–15.

Ewert, M.A., D.R. Jackson, and P. Moler. 2006. *Macrochelys temminckii*– Alligator Snapping Turtle. Chelonian Research Monographs 3:58–71.

Folt, B., J.B. Jensen, A. Teare, and D. Rostal. 2016. Establishing reference demography for conservation: a case study of *Macrochelys temminckii* in Spring Creek, Georgia. Herpetological Monographs 30:21–33.

Heppel, S.S. 1998. Application of life-history theory and population model analysis to turtle conservation. Copeia 1998:367–375.

Iverson, J. B., C. P. Balgooyen, K. K. Byrd, and K. K. Lyddan. 1993. Latitudinal variation in egg and clutch size in turtles. Canadian Journal of Zoology 71:2448–2461.

Kendall, B.E. 2019. mpmtools: Tools for building and analyzing matrix population models. R package version 0.2.0, https://github.com/BruceKendall/mpmtools.

Kendall, B.E., M. Fujiwara, J. Diaz-Lopez, S. Schneider, J. Voigt, and S. Wiesner. 2019. Persistent problems in the construction of matrix population models. Ecological Modelling 406:33–43.

Kessler, E.J., 2020. Assessment of freshwater turtle reintroduction. University of Illinois at Urbana-Champaign.

Koons, D.N., T.W. Arnold, and M. Schaub. 2017. Understanding the demographic drivers of realized population growth rates. Ecological Applications 27:2102–2115.

Louisiana Department of Wildlife and Fisheries [LDWF]. 2019a. Reptile and amphibian fishing regulations. http://www.wlf.louisiana.gov/fishing/reptile_amphib-regulations Accessed 8 May 2019.

LDWF. 2019b. Statistics: Recreational fishing license sales by parish (2013–2017). http://www.wlf.louisiana.gov/licenses/statistics. Accessed 8 May 2019.

McGowan, C.P., M.C. Runge, and M.A. Larson. 2011. Incorporating parametric uncertainty into population viability analysis models. Biological Conservation 144:1400–1408.

McGowan, C.P., N. Allan, J. Servoss, S. Hedwall, and B. Woolridge. 2017. Incorporating population viability models into species status assessment and listing decisions under the U.S. Endangered Species Act. Global Ecology and Conservation 12:119–130.
Miller, J.L., and D.B. Ligon. 2014a. Sex ratios in naturally incubating *Marcochelys temminckii* nests, a species with type II temperature-dependent sex determination. Journal of Thermal Biology 39:6–11.
Miller, J.L., D.M. Thompson, D.B. Ligon. 2014b. Estimating Clutch Size Using Eggshell Fragments in Depredated *Macrochelys temminckii* Nests. Herpetological Review 45:401–404.
Mississippi Fish, Wildlife, and Parks [MFWP]. 2019. Information on harvest of turtles and other non-game wildlife. https://www.mdwfp.com/fishing-boating/freshwater-commercial/turtle-information/. Accessed 8 May 2019.
Morris, W.L., and D. Doak. 2002. Quantitative Conservation Biology. Sinauer Associates, Inc., Sunderland, Massachusetts, USA.
Riedle, J.D., P.A. Shipman, S.F. Fox, D.M. Leslie Jr. 2006. Microhabitat use, and movements of the alligator snapping turtle *Macrochelyns temminckii*, in Oklahoma. The Southwestern Naturalist 51:35–40.
Speirs-Bridge, A., F. Fidler, M. McBride, L. Flander, G. Cumming, and M. Burgman. 2010. Reducing overconfidence in the interval judgments of experts. Risk Analysis 30:512–523.
Stearns, S.C. 1992. The evolution of life histories. Oxford University Press, New York, NY, USA.
Stubben, C., B. Milligan, and P. N. Maintainer. 2016. Package "popbio" : Construction and Analysis of Matrix Population Models.
Thompson, D.M., D.B. Ligon, J.C. Patton, and M. Papes. 2016. Effects of life-history requirements on the distribution of a threatened reptile. Conservation Biology 32:427–436.
Tucker, A.D., and K.N. Sloan. 1997. Growth and reproductive estimates from Alligator Snapping Turtles, *Macroclemys temminckii*, taken by commercial harvest in Louisiana. Chelonian Conservation and Biology 2:587–592.
U.S. Fish and Wildlife Service. 2015. Species Status Assessment for the Sonoran Desert Tortoise. Version 1.0, September 2015. U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, NM.
R Core Team. 2019. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://R-project.org
Williams, B.K., J.D. Nichols, and M.J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, New York, UA.

Table E1. Summary of Alligator snapping turtle (*Macrochelys temminckii*) demographic parameter estimates used to populate a two-stage, female-only Lefkovitch matrix population model with a prebreeding census structure. The two stages included juvenile individuals (J) that were greater than one year of age, but reproductively immature, and adults (A) that had reached reproductive maturity. The matrix model contained four elements: (1) juvenile retention, the probability of surviving and remaining in the juvenile stage class (PJ = φJ×(1− γ J)); (2) juvenile growth, the probability of surviving as a juvenile and transitioning to the adult stage (GJ = φJ×γJ); (3) adult retention, the probability of surviving and remaining in the adult (terminal) stage (P_A: φ_A); and (4) adult fecundity, the number of female offspring produced per breeding adult female each year ($F_A = BP \times CS \times \varphi_N \times NSC \times FP \times \varphi_H$). The Sampling Variance (σ_S^2) column reflects the amount of variation in the parameter's mean value attributed to sampling error, whereas the Process Variance (σ_P^2) column reflects the temporal fluctuation in a parameter due to demographic or environmental stochasticity.

Matrix Element(s)	Demographic Parameter[a,b]	Mean (μ)	Sampling Var. (σ_S^2)	Process Var. (σ_P^2)	Source	Source Location
P_J, G_J	$\varphi_{J,1–7}$	0.860	0.0277^2	0.01053^2	Folt et al. 2016	Spring Creek, Georgia
P_J, G_J	$\varphi_{J,8}$	0.730	0.0354^2	0.01082^2	Dreslik et al. 2017	Illinois
P_J, G_J	γ_J	0.020	0.0111^2	0.00889^2	Tucker and Sloan 1997	Louisiana
P_A	φ_A	0.950	0.0174^2	0.00969^2	Folt et al. 2016	Spring Creek, Georgia
F_A	*BP*	0.980	0.0112^2	0.00894^2	Dobie 1971	Southern Louisiana
F_A	*CS*	33.200	10.0000^2	5.00000^2	Weighted average[b]; Folt et al. 2016 (SD)	Multiple
F_A	φ_N	0.130	0.0269^2	0.01037^2	Ewert et al. 2006	Lower Apalachicola River, Florida
F_A	*NSC*	0.723	0.0358^2	0.01097^2	Ewert et al. 2006	Lower Apalachicola River, Florida
F_A	*FP*	0.500	0.0400^2	0.01090^2	Expert opinion	–
F_A	φ_H	0.150	0.0285^2	0.01060^2	Expert opinion	–

[a]Demographic parameter mean, sampling variance, and process variance values apply to all analysis units (1–8), with the exception juvenile survival (φ_J), which used different values for analysis units 1–7 (row 1) and 8 (row 2).

[b]The φ symbols refer to the annual survival of adults (A), juveniles (J), and hatchlings (H) from nest emergence to one year of age, whereas φ_N is the proportion of AST nests in which at least one egg successfully hatched (i.e., nest survival). *BP* is the proportion of adult females that breed annually, *CS* is clutch size, *NSC* is the proportion of eggs from which a hatchling successfully emerged among surviving nests, *FP* is the proportion of female hatchlings, whereas γ_J is the proportion of juveniles that transition to the adult stage each year.

[c]Mean clutch size (*CS*) was derived using a weighted mean across multiple studies, using the sample size (number of nests) from each study as weights. Full details are given in Table E2

Table E2. Clutch sizes of alligator snapping turtles (*Macrochelys temminckii*) used to compute a weighted mean in a stochastic population simulation. The mean clutch sizes were weighted by the sample size (number of nests) from each study to derive the overall weighted mean (33.2).

Mean	Error (SD)[a]	Sample Size	Description	Location	Source
37.3	–	31	Mean number of eggs within an active wild nest.	Lower Apalachicola	Ewert et al. 2006
35.1	6.6	130[b]	Mean number of eggs within an active wild nest.	Lower Apalachicola	Ewert and Jackson 1994
32	12.17	3	Mean number of eggs within an active wild nest.	Northwest Florida river drainages (non-Apalachicola)	Ewert 1976
24.5	7.3	13	Dissected adult female AST taken as bycatch prior to nesting season; clutch size indicates the number of shelled eggs.	Louisiana	Dobie 1971
22.4	–	6	Mean number of eggs within an active wild nest.	Tishomingo NWR, Oklahoma	Miller and Ligon 2014a
18.6	5.68	16	Examination of depredated wild nests; clutch size estimated from shell membranes; method verified against nests with known clutch sizes (R^2: 0.97).	Tishomingo NWR, Oklahoma	Miller et al. 2014b

[a]Dashes (–) indicate that standard deviation or other measure of error were not reported.

[b]The sample size of the Ewert and Jackson (1994) study is mistakenly reported as 160 nests in Ewert et al. (2006).

Table E3. Threat-specific percent reductions (mean ± standard deviation) to alligator snapping turtle (*Macrochelys temminickii*; hereafter AST) survival parameters, derived from remote expert elicitation among a team of taxon experts. These quantities were assumed to remain constant across the AST's range, meaning that the percent reduction attributed to a specific threat was not assumed to vary among analysis units, though the proportion of the population exposed to a particular threat within an analysis unit may vary. The dashes (–) indicate that the survival parameter was not exposed to the specific threat within the model. For example, hatchlings are likely too small to ingest hooks, so their survival rate was not reduced by HKI. The mean values contained within each cell represent the percent reductions under the "expert-elicited" scenarios, with conservation action absent or present (EETH, EETH+), whereas they were reduced or increased by 25% for the "decreased threat" and "increased threat" threat scenarios, respectively.

	Commercial Bycatch (BYC)	**Recreational Bycatch (BYR)**	**Hook Ingestion (HKI)**	**Legal Collection (CLL)[a]**	**Illegal Collection (CLI)**	**Subsidized Nest Predators (SNP)**
Hatchling Survival (φ_H)	0.0001 ± 0.0007	–	–	0.0045 ± 0.0027	0.0047 ± 0.0028	–
Juvenile Survival (φ_J)	0.0403 ± 0.0258	0.0579 ± 0.0205	0.0615 ± 0.0195	0.0412 ± 0.0167	0.0565 ± 0.0191	–
Adult Survival (φ_A)	0.0630 ± 0.0361	0.0741 ± 0.0351	0.0824 ± 0.0322	0.1998 ± 0.0563	0.1947 ± 0.0625	–
Nest Survival (φ_N)	–	–	–	–	0.0110 ± 0.01167	0.6075 ± 0.1154

[a]We did not use the CLL values in the model because differences in legal collection policies among states violated the assumption of a constant percent-reduction across analysis units. Instead, we simulated CLL as a reduction in abundance, rather than survival rates.

Table E4. Expert elicited mean (± standard deviation) spatial extent of threats to alligator snapping turtle (*Macrochelys temminckii*) population viability within each analysis unit (columns). Dashes indicate that the threat does not occur in the specific analysis unit.

	Southern Miss. – East (SOME)	**Alabama (ALAB)**	**Apalachicola (APAL)**	**Northern Miss. – East (NOME)**
Commercial Bycatch (BYC)[a]	0.500 ± 0.081	0.500 ± 0.050	0.500 ± 0.050	–
Recreational Bycatch (BYR)[b]	0.443 ± 0.089	0.611 ± 0.104	0.443 ± 0.153	0.01 ± 0.005
Legal Collection (CLL)[c]	0.52 ± 0.063	0.400 ± 0.043	–	–
Illegal Collection (CLI)	0.647 ± 0.119	0.758 ± 0.074	0.389 ± 0.084	0.001 ± 0.006
Subsidized Nest Predators (SNP)	0.943 ± 0.109	0.902 ± 0.128	0.659 ± 0.041	0.923 ± 0.019
Habitat Loss Rate (HLR)	2.75 ± 1.25	2.80 ± 0.83	2.0 ± 1.0	2.0 ± 1.41

[a]We did not receive any responses for the BYC spatial extent in the ALAB or APAL units, so we assigned a mean value of 0.50 with a 0.20 coefficient of variation on standard deviation, to reflect the uncertainty regarding this parameter.

[b]In the expert elicitation questionnaire the spatial extents for BYR and hook ingestion (HKI) were considered the same, which was reflected in the model as well.

[c]We did not use the CLL values in the model because differences in legal collection policies among states violated the assumption of a constant percent-reduction across analysis units. Instead, we simulated CLL as a reduction in abundance, and used the proportion of the analysis unit that overlapped with Louisiana as a spatial extent.

Table E5. Summary of alligator snapping turtle (*Macrochelys temminckii*; AST) projection matrix elements from a stochastic population simulation. The framework simulated AST population dynamics within each of the four analysis units with sufficient data, under six different scenarios. For each analysis unit and scenario combination, we ran 500 replicates of AST population dynamics simulated for 50 years. Analysis unit names are given in italics above their respective sections. The six scenarios included decreased (DE-), expert-elicited (EE-), or increased (IN-) threat levels (rows within each analysis unit section), with conservation action absent (-TH) or present (TH+). The projection matrix elements (columns) describe stage class-specific demographic processes and include: juvenile retention (P_J), juvenile growth (G_J), adult retention (P_A), and adult fecundity (F_A). The mean ± standard deviations for each element, averaged across all iterations and years, are given below, with their overall range in parentheses. We also provide baseline element values, prior to incorporating stochasticity and threat effects, for the deterministic transition matrices all analysis units except Northern Mississippi – East ($\mathbf{D}_{SOME,\ ALAB,\ APAL}$) and Northern Mississippi – East ($\mathbf{D}_{NOME}$).

Southern Mississippi – East

Scenario	P_J	G_J	P_A	F_A
DETH	0.811 ± 0.040 (0.630, 0.943)	0.017 ± 0.016 (0, 0.155)	0.788 ± 0.056 (0.461, 0.951)	0.097 ± 0.064 (0.003, 0.687)
EETH	0.787 ± 0.041 (0.562, 0.918)	0.015 ± 0.016 (0, 0.19)	0.734 ± 0.058 (0.489, 0.910)	0.100 ± 0.065 (0.005, 0.788)
INTH	0.764 ± 0.041 (0.556, 0.896)	0.015 ± 0.015 (0, 0.203)	0.681 ± 0.06 (0.410, 0.867)	0.096 ± 0.060 (0.003, 0.791)
DETH+	0.829 ± 0.004 (0.621, 0.961)	0.016 ± 0.017 (0, 0.141)	0.829 ± 0.048 (0.588, 0.961)	0.098 ± 0.064 (0.004, 0.699)
EETH+	0.810 ± 0.04 (0.630, 0.936)	0.016 ± 0.017 (0, 0.172)	0.789 ± 0.05 (0.533, 0.936)	0.099 ± 0.063 (0.005, 0.788)
INTH+	0.793 ± 0.04 (0.612, 0.928)	0.017 ± 0.016 (0, 0.192)	0.749 ± 0.053 (0.478, 0.900)	0.102 ± 0.069 (0.003, 0.773)

Alabama

Scenario	P_J	G_J	P_A	F_A
DETH	0.792 ± 0.042 (0.542, 0.932)	0.015 ± 0.016 (0, 0.274)	0.754 ± 0.062 (0.429, 0.926)	0.100 ± 0.064 (0.003, 0.671)
EETH	0.760 ± 0.043 (0.580, 0.897)	0.016 ± 0.016 (0, 0.170)	0.688 ± 0.063 (0.420, 0.876)	0.104 ± 0.069 (0.005, 0.936)
INTH	0.734 ± 0.041 (0.491, 0.875)	0.015 ± 0.015 (0, 0.157)	0.623 ± 0.064 (0.282, 0.841)	0.104 ± 0.069 (0.002, 0.859)
DETH+	0.813 ± 0.040 (0.615, 0.947)	0.016 ± 0.016 (0, 0.217)	0.803 ± 0.051 (0.574, 0.951)	0.105 ± 0.068 (0.003, 0.789)
EETH+	0.792 ± 0.04 (0.558, 0.921)	0.016 ± 0.016 (0, 0.159)	0.755 ± 0.052 (0.519, 0.911)	0.104 ± 0.066 (0.002, 0.555)
INTH+	0.770 ± 0.040 (0.532, 0.904)	0.016 ± 0.016 (0, 0.272)	0.705 ± 0.054 (0.471, 0.873)	0.103 ± 0.066 (0.005, 0.808)

Apalachicola

Scenario	P_J	G_J	P_A	F_A
DETH	0.820 ± 0.042 (0.529, 0.953)	0.016 ± 0.017 (0, 0.255)	0.824 ± 0.048 (0.597, 0.96)	0.139 ± 0.079 (0.007, 0.883)
EETH	0.801 ± 0.042 (0.601, 0.944)	0.015 ± 0.016 (0, 0.187)	0.783 ± 0.051 (0.507, 0.94)	0.137 ± 0.081 (0.008, 0.853)
INTH	0.778 ± 0.043 (0.583, 0.92)	0.016 ± 0.016 (0, 0.214)	0.741 ± 0.054 (0.432, 0.922)	0.142 ± 0.084 (0.007, 0.921)
DETH+	0.836 ± 0.04 (0.545, 0.959)	0.017 ± 0.017 (0, 0.222)	0.855 ± 0.043 (0.64, 0.976)	0.137 ± 0.079 (0.007, 0.777)
EETH+	0.819 ± 0.041 (0.634, 0.954)	0.016 ± 0.016 (0, 0.175)	0.825 ± 0.045 (0.586, 0.967)	0.14 ± 0.081 (0.011, 0.838)
INTH+	0.803 ± 0.042 (0.597, 0.95)	0.017 ± 0.017 (0, 0.209)	0.793 ± 0.05 (0.54, 0.935)	0.138 ± 0.077 (0.005, 0.712)

Northern Mississippi – East

Scenario	P_J	G_J	P_A	F_A
DETH	0.714 ± 0.048 (0.482, 0.875)	0.014 ± 0.014 (0, 0.146)	0.947 ± 0.027 (0.812, 0.999)	0.1 ± 0.059 (0.002, 0.728)
EETH	0.714 ± 0.048 (0.522, 0.876)	0.014 ± 0.015 (0, 0.225)	0.946 ± 0.027 (0.806, 0.999)	0.099 ± 0.061 (0.001, 0.738)
INTH	0.712 ± 0.047 (0.539, 0.898)	0.014 ± 0.015 (0, 0.151)	0.946 ± 0.027 (0.767, 0.999)	0.099 ± 0.064 (0.003, 0.684)
DETH+	0.717 ± 0.047 (0.518, 0.896)	0.014 ± 0.014 (0, 0.19)	0.948 ± 0.028 (0.808, 1)	0.098 ± 0.061 (0.004, 0.781)
EETH+	0.713 ± 0.047 (0.526, 0.898)	0.014 ± 0.015 (0, 0.133)	0.946 ± 0.028 (0.818, 1)	0.099 ± 0.062 (0.001, 0.664)
INTH+	0.717 ± 0.048 (0.474, 0.895)	0.014 ± 0.015 (0, 0.292)	0.948 ± 0.028 (0.783, 1)	0.102 ± 0.065 (0.001, 0.826)

Deterministic

	P_J	G_J	P_A	F_A
$D_{SOME,ALAB,APAL}$	0.843	0.017	0.950	0.229
D_{NOME}	0.715	0.014	0.950	0.229

Table E6. Alligator snapping turtle (*Macrochelys temminckii*; AST hereafter) mean asymptotic population growth rates (lambdas; λ) derived from projection matrices for each analysis unit and scenario combination. Analysis unit abbreviations (for those with sufficient data) are bolded in each section: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). The six scenarios included Decreased, Expert-Elicited, or Increased threat levels (rows within each analysis unit section), with conservation action absent (*TH*) or present (*TH+*) columns). For each analysis unit and scenario combination, we ran 500 replicates of AST population dynamics simulated for 50 years. Our simulation generated a maximum of 25K λ values, though if the population declined to zero during an iteration, the projection stopped and began the next iteration. Mean λ quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all replicates) listed in parentheses below, in which λ<1 denotes a decreasing population, whereas λ≥1 indicates a stable or increasing population. For comparative purposes, we also calculated λ for the baseline scenario simulations. Though we ran two baseline scenarios, each consisting of 1,000 replicates per analysis unit per scenario, we pooled the output across scenarios to obtain the means here because asymptotic lambda would not have been influenced by the differences in scenario structure, which reflected the presence or absence of released turtles. The baseline projection matrices were parameterized with the baseline demographic parameter values (i.e., the raw values before adjustment for threat exposure) listed in Table E1. We further pooled across non-NOME units (SOME, ALAB, APAL; bottom left) as the baseline demographic parameters were the same, and the only difference among analysis units was the initial population size; whereas the NOME unit (right) differed in juvenile survival and was kept separate.

Threat Level	**SOME**		**ALAB**	
	TH	*TH+*	*TH*	*TH+*
Decreased	0.848 ± 0.036	0.873 ± 0.035	0.824 ± 0.037	0.854 ± 0.035
	(0.657, 1.015)	(0.741, 1.027)	(0.663, 0.980)	(0.706, 1.007)
Expert-Elicited	0.812 ± 0.036	0.845 ± 0.035	0.783 ± 0.038	0.822 ± 0.035
	(0.657, 0.958)	(0.703, 0.995)	(0.622, 0.931)	(0.661, 1.002)
Increased	0.782 ± 0.037	0.821 ± 0.036	0.749 ± 0.038	0.793 ± 0.036
	(0.620, 0.931)	(0.668, 0.984)	(0.579, 0.936)	(0.628, 0.941)

Threat Level	**APAL**		**NOME**	
	TH	*TH+*	*TH*	*TH+*
Decreased	0.871 ± 0.038	0.895 ± 0.036	0.953 ± 0.028	0.954 ± 0.028
	(0.714, 1.03)	(0.74, 1.043)	(0.816, 1.062)	(0.818, 1.077)
Expert-Elicited	0.841 ± 0.039	0.87 ± 0.038	0.952 ± 0.028	0.952 ± 0.028
	(0.665, 1.003)	(0.71, 1.027)	(0.824, 1.059)	(0.821, 1.056)
Increased	0.812 ± 0.041	0.847 ± 0.04	0.952 ± 0.028	0.954 ± 0.028
	(0.647, 0.985)	(0.687, 1.012)	(0.775, 1.046)	(0.784, 1.063)

Baseline			
Non-NOME Units:	0.988 ± 0.038	NOME:	0.963 ± 0.030

Table E7. Projection matrix element elasticities from simulated alligator snapping turtle (*Macrochelys temminckii*) populations. The projection matrix elements are listed in the four columns to the right and include: juvenile retention (P_J), juvenile growth (G_J), adult retention (P_A), and adult fecundity (F_A). Analysis unit names are given in italics above their respective sections, only units with sufficient data are included here. The six scenarios included three different threat levels— decreased (DE-), expert-elicited (EE-), or increased (IN-), with conservation action absent (-TH) or present (-TH+). For each analysis unit (for which sufficient data were available) and scenario combination, we calculated mean elasticities (± standard deviation) for the projection matrix elements across all timesteps (*n*=50) and iterations (*n*=500), with the range (i.e., minimum and maximum values observed) values given in parentheses. For comparison, we also provide elasticities from the matrix elements of the deterministic projection matrices that contain baseline demographic parameters (Table E1), prior to incorporating stochasticity and threat effects. The elasticities are separated by analysis units: all analysis units except Northern Mississippi – East ($\mathbf{D}_{\text{SOME,ALAB, APAL}}$) and Northern Mississippi – East ($\mathbf{D}_{\text{NOME}}$).

Southern Mississippi – East

Scenario	P_J	G_J	P_A	F_A
DETH	0.578 ± 0.321 (0, 1)	0.025 ± 0.047 (0, 0.25)	0.372 ± 0.308 (0, 1)	0.025 ± 0.047 (0, 0.25)
EETH	0.636 ± 0.313 (0, 1)	0.054 ± 0.088 (0, 0.25)	0.257 ± 0.251 (0, 1)	0.054 ± 0.088 (0, 0.25)
INTH	0.653 ± 0.302 (0, 1)	0.079 ± 0.106 (0, 0.25)	0.188 ± 0.183 (0, 1)	0.079 ± 0.106 (0, 0.25)
DETH+	0.496 ± 0.327 (0, 1)	0.016 ± 0.018 (0, 0.25)	0.471 ± 0.323 (0, 1)	0.016 ± 0.018 (0, 0.25)
EETH+	0.582 ± 0.317 (0, 1)	0.026 ± 0.049 (0, 0.25)	0.366 ± 0.303 (0, 1)	0.026 ± 0.049 (0, 0.25)
INTH+	0.634 ± 0.298 (0, 1)	0.045 ± 0.077 (0, 0.25)	0.276 ± 0.251 (0, 1)	0.045 ± 0.077 (0, 0.25)

Alabama

Scenario	P_J	G_J	P_A	F_A
DETH	0.643 ± 0.322 (0, 1)	0.017 ± 0.025 (0, 0.25)	0.325 ± 0.312 (0, 1)	0.017 ± 0.025 (0, 0.25)
EETH	0.703 ± 0.292 (0, 1)	0.041 ± 0.074 (0, 0.25)	0.216 ± 0.233 (0, 1)	0.041 ± 0.074 (0, 0.25)
INTH	0.712 ± 0.306 (0, 1)	0.069 ± 0.101 (0, 0.25)	0.149 ± 0.161 (0, 1)	0.069 ± 0.101 (0, 0.25)
DETH+	0.541 ± 0.323 (0, 1)	0.016 ± 0.014 (0, 0.25)	0.427 ± 0.318 (0, 1)	0.016 ± 0.014 (0, 0.25)
EETH+	0.659 ± 0.302 (0, 1)	0.017 ± 0.024 (0, 0.25)	0.307 ± 0.290 (0, 1)	0.017 ± 0.024 (0, 0.25)
INTH+	0.727 ± 0.269 (0, 1)	0.031 ± 0.059 (0, 0.25)	0.212 ± 0.225 (0, 1)	0.031 ± 0.059 (0, 0.25)

Apalachicola

Scenario	P_J	G_J	P_A	F_A
DETH	0.478 ± 0.31 (0, 1)	0.019 ± 0.016 (0, 0.25)	0.484 ± 0.306 (0, 1)	0.019 ± 0.016 (0, 0.25)
EETH	0.577 ± 0.308 (0, 1)	0.02 ± 0.025 (0, 0.25)	0.382 ± 0.298 (0, 1)	0.02 ± 0.025 (0, 0.25)
INTH	0.628 ± 0.288 (0, 1)	0.036 ± 0.06 (0, 0.25)	0.301 ± 0.256 (0, 1)	0.036 ± 0.06 (0, 0.25)
DETH+	0.409 ± 0.293 (0, 1)	0.019 ± 0.015 (0, 0.093)	0.553 ± 0.292 (0, 1)	0.019 ± 0.015 (0, 0.093)
EETH+	0.469 ± 0.301 (0, 1)	0.019 ± 0.016 (0, 0.25)	0.492 ± 0.298 (0, 1)	0.019 ± 0.016 (0, 0.25)
INTH+	0.541 ± 0.293 (0, 1)	0.021 ± 0.02 (0, 0.25)	0.417 ± 0.285 (0, 1)	0.021 ± 0.02 (0, 0.25)

Northern Mississippi – East

Scenario	P_J	G_J	P_A	F_A
DETH	0.017 ± 0.021 (0, 0.273)	0.006 ± 0.007 (0, 0.079)	0.972 ± 0.034 (0.668, 1)	0.006 ± 0.007 (0, 0.079)
EETH	0.018 ± 0.023 (0, 0.308)	0.006 ± 0.007 (0, 0.066)	0.971 ± 0.035 (0.604, 1)	0.006 ± 0.007 (0, 0.066)
INTH	0.017 ± 0.021 (0, 0.236)	0.006 ± 0.007 (0, 0.083)	0.972 ± 0.034 (0.661, 1)	0.006 ± 0.007 (0, 0.083)
DETH+	0.017 ± 0.022 (0, 0.39)	0.005 ± 0.006 (0, 0.078)	0.972 ± 0.033 (0.597, 1)	0.005 ± 0.006 (0, 0.078)
EETH+	0.017 ± 0.022 (0, 0.622)	0.006 ± 0.007 (0, 0.25)	0.971 ± 0.034 (0.25, 1)	0.006 ± 0.007 (0, 0.25)
INTH+	0.019 ± 0.024 (0, 0.849)	0.006 ± 0.007 (0, 0.08)	0.97 ± 0.037 (0.144, 1)	0.006 ± 0.007 (0, 0.08)

Deterministic

	P_J	G_J	P_A	F_A
$\mathbf{D}_{\text{SOME, ALAB, APAL}}$	0.1510	0.0244	0.8002	0.0244
$\mathbf{D}_{\text{NOME}}$	0.0383	0.0133	0.9351	0.0132

Table E8. Projection matrix stable stage distributions from simulated alligator snapping turtle (*Macrochelys temminckii*; hereafter AST) populations under. Analysis unit abbreviations (for which sufficient data were available) are given in italics above their respective sections, and include: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). We simulated AST populations for each analysis unit under six different future condition scenarios, listed in the far left column. The six scenarios included three different threat levels—decreased (DE), expert-elicited (EE), or increased (IN)—with conservation action absent (TH) or present (TH+). For each analysis unit and scenario combination, we computed the mean proportion (± standard deviation) of each stage class across all iterations (*n*=500) and timesteps (max=50), with the range (i.e., minimum and maximum values observed) values given in parentheses. For comparison, we provide the stable stage distributions of the deterministic projection matrices that contain baseline demographic parameters (Table E1), prior to incorporating stochasticity and threat effects. The stable stage distributions are separated by analysis units: all analysis units except Northern Mississippi – East and Northern Mississippi – East.

Scenario	**SOME**		**Scenario**	**ALAB**	
	Juveniles	Adults		Juveniles	Adults
DETH	0.738 ± 0.196 (0.067, 1)	0.262 ± 0.196 (0, 0.933)	DETH	0.770 ± 0.191 (0.065, 1)	0.222 ± 0.191 (0, 0.935)
EETH	0.762 ± 0.193 (0.111, 1)	0.238 ± 0.193 (0, 0.889)	EETH	0.795 ± 0.180 (0.094, 1)	0.205 ± 0.180 (0, 0.906)
INTH	0.757 ± 0.200 (0.117, 1)	0.243 ± 0.200 (0, 0.883)	INTH	0.791 ± 0.194 (0.162, 1)	0.209 ± 0.194 (0, 0.838)
DETH+	0.709 ± 0.204 (0.069, 1)	0.291 ± 0.204 (0, 0.931)	DETH+	0.733 ± 0.198 (0.046, 1)	0.267 ± 0.198 (0, 0.954)
EETH+	0.746 ± 0.196 (0.074, 1)	0.254 ± 0.196 (0, 0.926)	EETH+	0.791 ± 0.18 (0.069, 1)	0.209 ± 0.180 (0, 0.931)
INTH+	0.760 ± 0.186 (0.037, 1)	0.240 ± 0.186 (0, 0.963)	INTH+	0.814 ± 0.171 (0.082, 1)	0.191 ± 0.169 (0, 0.897)
Scenario	**APAL**		**Scenario**	**NOME**	
	Juveniles	Adults		Juveniles	Adults
DETH	0.741 ± 0.178 (0.065, 1)	0.259 ± 0.178 (0, 0.935)	DETH	0.283 ± 0.118 (0.012, 0.824)	0.717 ± 0.118 (0.176, 0.988)
EETH	0.781 ± 0.172 (0.106, 1)	0.219 ± 0.172 (0, 0.894)	EETH	0.282 ± 0.121 (0, 0.811)	0.718 ± 0.121 (0.189, 1)
INTH	0.792 ± 0.167 (0.17, 1)	0.208 ± 0.167 (0, 0.83)	INTH	0.278 ± 0.122 (0.012, 0.897)	0.722 ± 0.122 (0.103, 0.988)
DETH+	0.707 ± 0.181 (0.106, 1)	0.293 ± 0.181 (0, 0.894)	DETH+	0.281 ± 0.121 (0.013, 0.926)	0.719 ± 0.121 (0.074, 0.987)
EETH+	0.739 ± 0.175 (0.118, 1)	0.261 ± 0.175 (0, 0.882)	EETH+	0.28 ± 0.12 (0, 0.844)	0.72 ± 0.12 (0.156, 1)
INTH+	0.763 ± 0.166 (0.112, 1)	0.237 ± 0.166 (0, 0.888)	INTH+	0.287 ± 0.124 (0, 0.933)	0.713 ± 0.124 (0.067, 1)

Analysis Unit(s)	**Deterministic Matrices**	
	Juveniles	Adults
SOME, ALAB, APAL	0.6275	0.3725
NOME	0.4804	.0.5196

Table E9. Summary of alligator snapping turtle (*Marcochelys temminckii*; hereafter AST) population outcomes from six different scenarios, separated by analysis unit. For each analysis unit (italics above each section) and scenario combination, we ran 500 iterations of AST population dynamics simulated for 50 years. The six scenarios included three threat levels, Decreased, Expert-Elicited, or Increased (rows within each analysis unit section), with conservation action absent (TH) or present (TH+) (columns) for each level. For each scenario, we calculated the proportion of iterations in which the total population (both stage classes, females only) declined to zero (extirpation probability; p_{EX}) or less than 5% of the starting population size (quasi-extirpation probability; p_{QX}). For the iterations in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, t_{EX} and t_{QX}, respectively. Mean quantities and their standard deviations are listed with the range (minimum and maximum quantity observed across all iterations) listed in parentheses below. Dashes (–) indicate that no simulation reached the extirpation or quasi-extirpation threshold, meaning that t_{EX} or t_{QX} were not calculated, whereas an asterisk (*) indicates only a single simulation crossed the threshold, precluding a standard deviation calculation.

Southern Mississippi – East

	p_{EX}		t_{EX}		p_{QX}		t_{QX}	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.434	0.058	47.46 ± 3.05 (41,53)	49.45 ± 1.92 (43, 51)	1.0	1.0	17.69 ± 2.40 (11, 29)	20.9 ± 3.34 (14, 35)
Expert-Elicited	0.950	0.476	43.33 ± 3.97 (32, 51)	47.49 ± 2.84 (39, 51)	1.0	1.0	14.89 ± 1.75 (10, 22)	17.74 ± 2.34 (12, 26)
Increased	0.998	0.856	38.07 ± 3.37 (30, 49)	44.92 ± 3.87 (33, 51)	1.0	1.0	12.97 ± 1.39 (9, 18)	15.74 ± 1.98 (11, 25)

Alabama

	p_{EX}		t_{EX}		p_{QX}		t_{QX}	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.130	0.002	48.91 ± 2.09 (43, 51)	51 ± * (51, 51)	1.0	1.0	17.68 ± 2.27 (12, 29)	22.84 ± 3.20 (14, 33)
Expert-Elicited	0.846	0.114	45.64 ± 3.36 (36, 51)	49.14 ± 2.23 (40, 51)	1.0	1.0	14.20 ± 1.6 (10, 20)	17.91 ± 2.27 (13, 26)
Increased	1.0	0.658	40.19 ± 3.47 (30, 51)	47.21 ± 2.76 (40, 51)	1.0	1.0	12.11 ± 1.35 (8, 16)	15.11 ± 1.72 (12, 23)

Apalachicola

	p_{EX}		t_{EX}		p_{QX}		t_{QX}	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.004	0.0	49.5 ± 0.71 (49, 50)	–	0.990	0.980	33.11 ± 6.09 (19, 51)	32.44 ± 6.1 (20, 51)
Expert-Elicited	0.124	0.006	49.02 ± 2.05 (44, 51)	50.67 ± 0.58 (50, 51)	1.0	1.0	26.28 ± 4.65 (16, 47)	32.04 ± 5.79 (18, 51)
Increased	0.660	0.052	46.82 ± 3.15 (35, 51)	48.92 ± 1.94 (48 51)	1.0	1.0	21.21 ± 3.25 (15, 36)	26.22 ± 4.75 (16, 51)

Northern Mississippi – East								
	p_{EX}		t_{EX}		p_{QX}		t_{QX}	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.0	0.0	– –	–	0.020	0.038	45.90 ± 4.01 (38, 51)	48.21 ± 2.90 (42, 51)
Expert-Elicited	0.0	0.002	– –	51.00 ± * (51, 51)	0.016	0.036	48 ± 4.11 (39, 51)	46.72 ± 3.39 (39, 51)
Increased	0.0	0.0	– –	–	0.024	0.020	45.42 ± 3.42 (41, 51)	46.60 ± 2.50 (42, 50)

Table E10. Regression-based sensitivity analysis to identify factors to which simulated alligator snapping turtle (*Marcochelys temminckii*; hereafter AST) realized growth rates were most sensitive. In each model, realized lambda (λ = total abundance at time t+1 divided by total abundance at time t) was modeled as a response to a single predictor variable (univariate models). The suite of predictor variables included the draws for each demographic parameter, threat-specific reduction to stage class survivals and the threat-specific spatial extents within each analysis unit. Each model included a maximum of 600,000 data points, based on 50 timesteps × 500 simulations × 6 scenarios × 4 analysis units. The demographic parameter predictor variables included adult survival (φ_A), juvenile survival (φ_J), hatchling survival (φ_H), nest survival (φ_N), breeding probability (BP), nest success (NSC), hatchling sex ratio (SR), and juvenile growth probability (γ_J). The threats (subscripts in Model column) included illegal collection (CLI), hook ingestion (HKI), recreational bycatch (BYR), commercial bycatch (BYC), and subsidized nest predators (SNP). Each threat had an analysis unit spatial extent ($\omega_{a,u,s,i,t}$) for threat a in unit u scenario s in simulation i at time t (Table E3), as well as a stage specific percent reduction to survival p ($\theta_{p,a,s,i,t}$; Table E4). The Model column lists the effect contained in the model; if a demographic parameter was included (either alone or through its connection to a threat effect) it is listed first, followed by the threat effect (ω or θ). For example, the first model represents the percent reduction to adult survival (φ_A) attributed to illegal collection (θ_{CLI}).

Model	**ΔAICc**	**w$_i$**	**Deviance**
φ_A / θ_{CLI}	0	1	9112.32
ω_{CLI}	7568.71	0	5008.31
φ_A / θ_{HKI}	12308.49	0	9269.43
φ_A / θ_{BYR}	13239.71	0	9281.43
φ_J / θ_{HKI}	15935.79	0	9316.24
φ_J / θ_{CLI}	16220.58	0	9319.93
φ_J / θ_{BYR}	16599.41	0	9324.83
φ_A / θ_{BYC}	17022.93	0	9330.32
φ_J / θ_{BYC}	20083.1	0	9370.06
φ_H / θ_{CLI}	21713.07	0	9391.29
φ_N / θ_{SNP}	21894.61	0	9393.66
φ_H / θ_{BYC}	23797.60	0	9418.52
ω_{BYR}	39472.36	0	5294.08
φ_J	141043.70	0	6317.07
φ_A	284981.00	0	8114.16
CS	289941.90	0	8184.47
φ_J	290605.80	0	8193.93
φ_H	291137.10	0	8201.50
γ_J	292294.10	0	8218.03
NSC	292958.00	0	8227.52
BP	293018.60	0	8228.39
SR	293022.20	0	8228.44
ω_{SNP}	293026.70	0	8228.50
ω_{BYC}	339076.70	0	4122.09

Table E11. Summary of alligator snapping turtle (AST hereafter) population outcomes averaged across the six future condition scenarios (Table 7), with and without legal collection (harvest). Note that all other output in the SSA contains the effect of legal collection. Louisiana is the only state within the AST's range that permits legal collection AST females. Here we show outcomes for the two analysis units that overlap with Louisiana: (a) Southern Mississippi East and (b) Alabama. We calculated the proportion of iterations in which the total population (both stage classes, females only) declined to zero (extirpation probability; p_{EX}) or less than 5% of the starting population size (quasi-extirpation probability; p_{QX}). For the iterations in which the population reached extirpation or quasi-extirpation, we then calculated the mean number of years until those thresholds were reached, t_{EX} and t_{QX}, respectively.

a. Southern Mississippi – East

	p_{EX}	t_{EX}	p_{QX}	t_{QX}
No legal collection	0.416 ± 0.40	47.35 ± 3.31	1.0 ± 0.0	19.33 ± 3.82
Legal collection	0.62 ± 0.37	45.11 ± 4.07	1.0 ± 0.0	16.66 ± 2.75

b. Alabama

	p_{EX}	t_{EX}	p_{QX}	t_{QX}
No legal collection	0.46 ± 0.42	45.56 ± 3.63	1.0 ± 0.00	18.11 ± 3.14
Legal collection	0.46 ± 0.42	47.01 ± 3.81	1.0 ± 0.0	16.48 ± 3.42

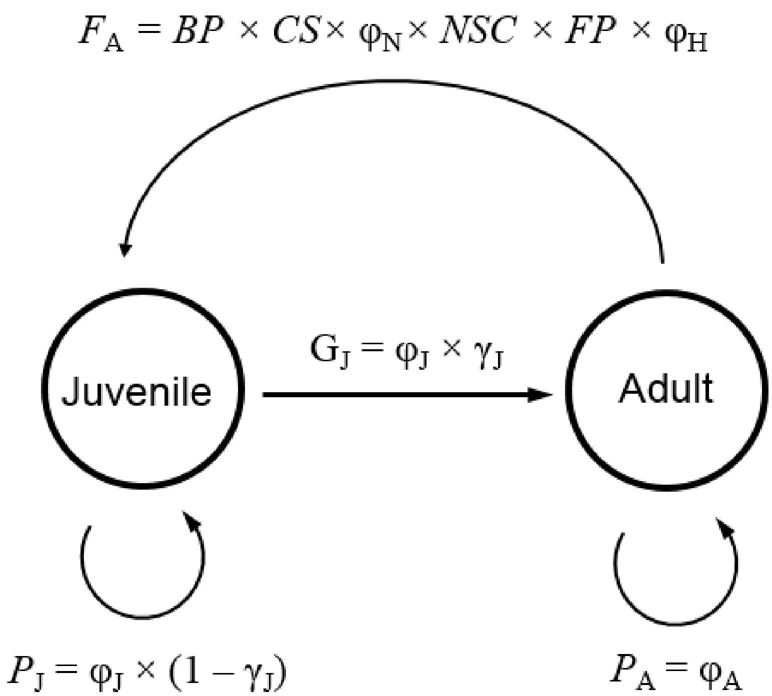


Figure E1. Alligator snapping turtle (*Macrochelys temminckii*) life cycle diagram for a female only two-stage prebreeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals), denoted by the J and A subscripts, respectively. At each timestep, Juveniles can either remain in their current stage with probability P_J, which is the product of juvenile survival (φ_J) and one minus the annual proportion of juveniles that recruit to the adult stage class (γ_J). Alternative, juveniles may transition to the adult stage (grow) with probability G_J, the product of φ_J and γ_J. Adults represent the terminal stage, therefore the probability that an individual remains in this stage (P_A) is simply their annual survival probability (φ_A). The arc shows the adult fecundity contribution (F_A), the number of juvenile females produced by each adult AST annually. Adult fecundity is the combined product of the annual probability that an adult females breeds (*BP*), clutch size (*CS*), the proportion of nests in which one egg hatches (i.e., nest survival; φ_N), the proportion of eggs that hatch in surviving nests (i.e., nest success; *NSC*), the proportion of female hatchlings (*FP*), and hatchling survival from nest emergence to one year of age (φ_H). The quantities used for each of the demographic parameters (e.g., φ_A) and their sources are given in Table E1.

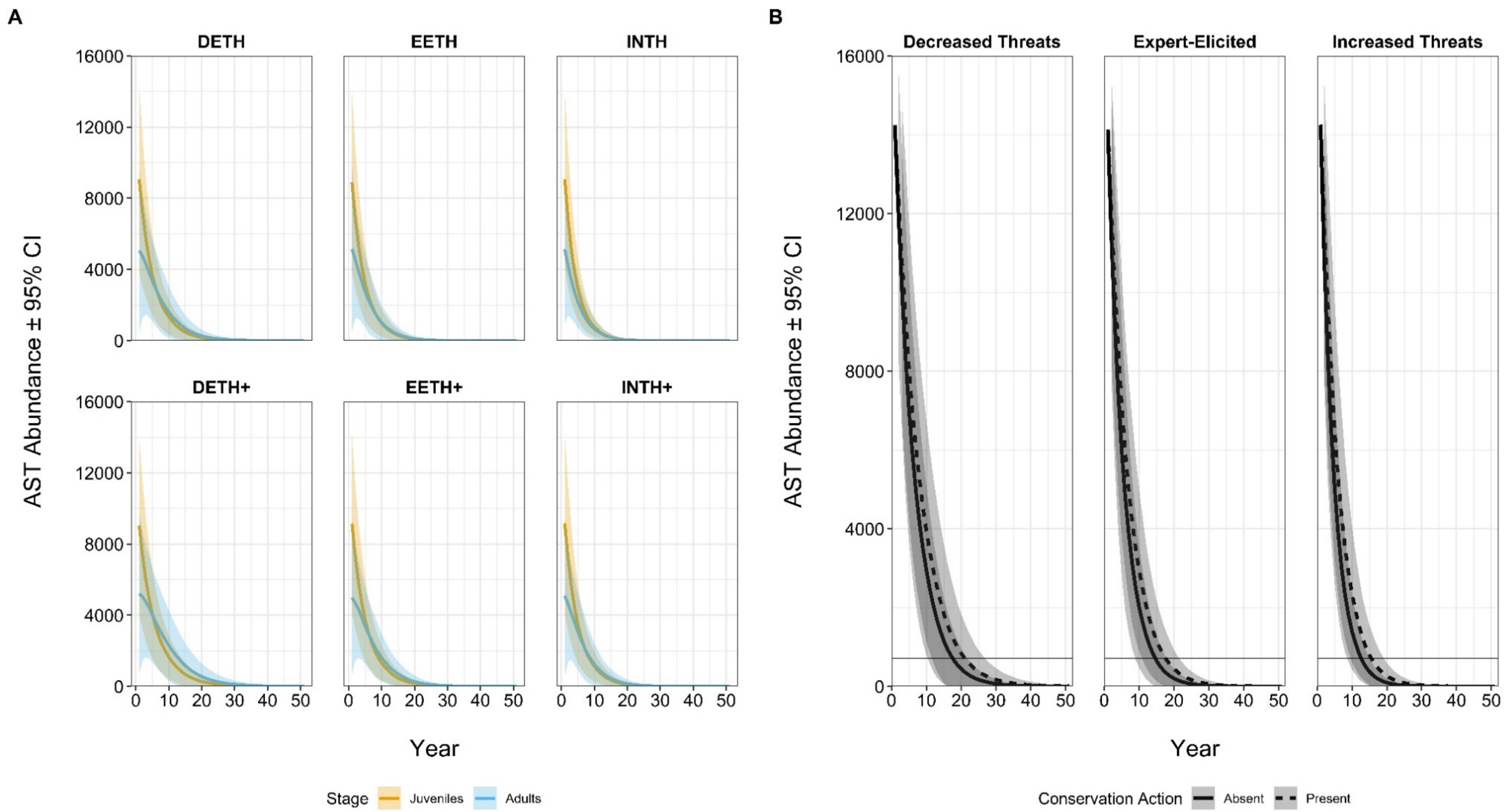


Figure E2. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Southern Mississippi – East analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the shaded areas reflect the 95% confidence intervals (CI). In (A) each panel represents one of six scenarios, varying by three threat levels (Decreased [DE], Expert-Elicited [EE], or Increased [IN]) across columns, and conservation actions absent (TH; top row) or present (TH+; bottom row). The orange line shows stage-specific abundance for juveniles and adults in blue. The columns in (B.) indicate the scenario's threat level (increasing from left to right). The solid and dashed lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the analysis unit-specific quasi-extirpation threshold (<5% of total abundance in Year 1) is given by the thin flat line.

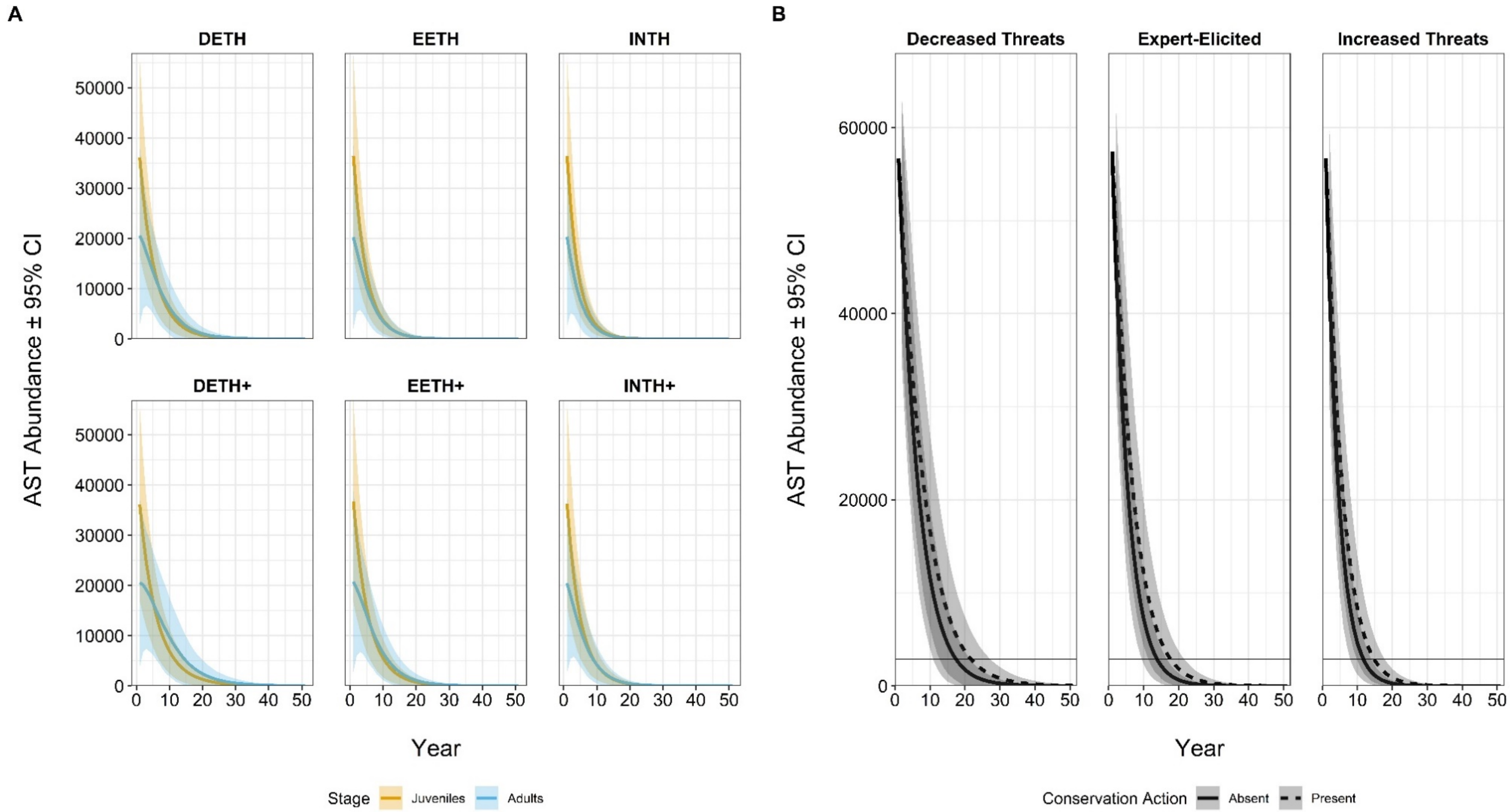


Figure E3. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Alabama analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the shaded areas reflect the 95% confidence intervals (CI). In (A) each panel represents one of six scenarios, varying by three threat levels (Decreased [DE], Expert-Elicited [EE], or Increased [IN]) across columns, and conservation actions absent (TH; top row) or present (TH+; bottom row). The orange line shows stage-specific abundance for juveniles and adults in blue. The columns in (B.) indicate the scenario's threat level (increasing from left to right). The solid and dashed lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the analysis unit-specific quasi-extirpation threshold (<5% of total abundance in Year 1) is given by the thin flat line.

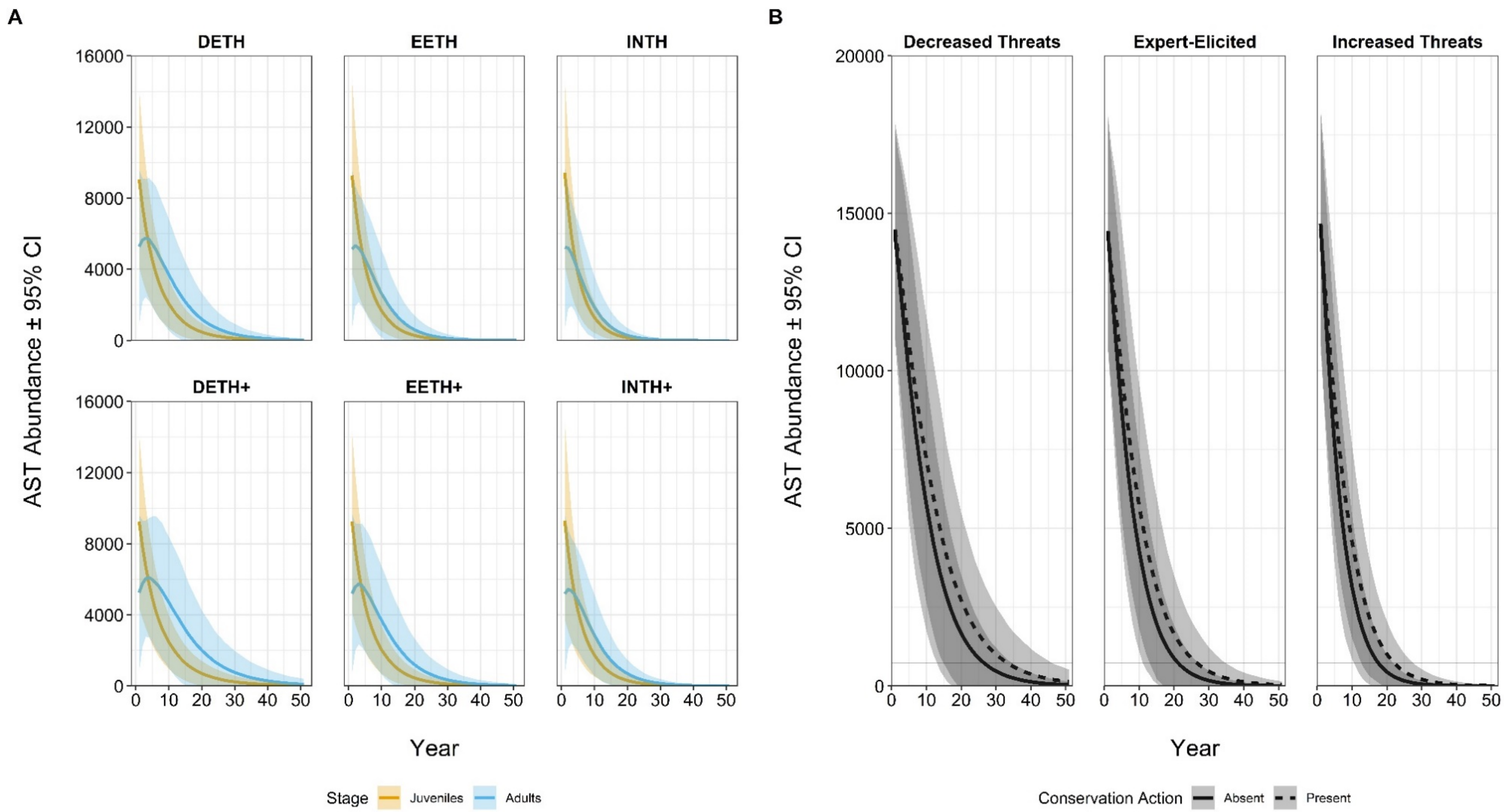


Figure E4. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Apalachicola analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the shaded areas reflect the 95% confidence intervals (CI). In (A) each panel represents one of six scenarios, varying by three threat levels (Decreased [DE], Expert-Elicited [EE], or Increased [IN]) across columns, and conservation actions absent (TH; top row) or present (TH+; bottom row). The orange line shows stage-specific abundance for juveniles and adults in blue. The columns in (B.) indicate the scenario's threat level (increasing from left to right). The solid and dashed lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the analysis unit-specific quasi-extirpation threshold (<5% of total abundance in Year 1) is given by the thin flat line.

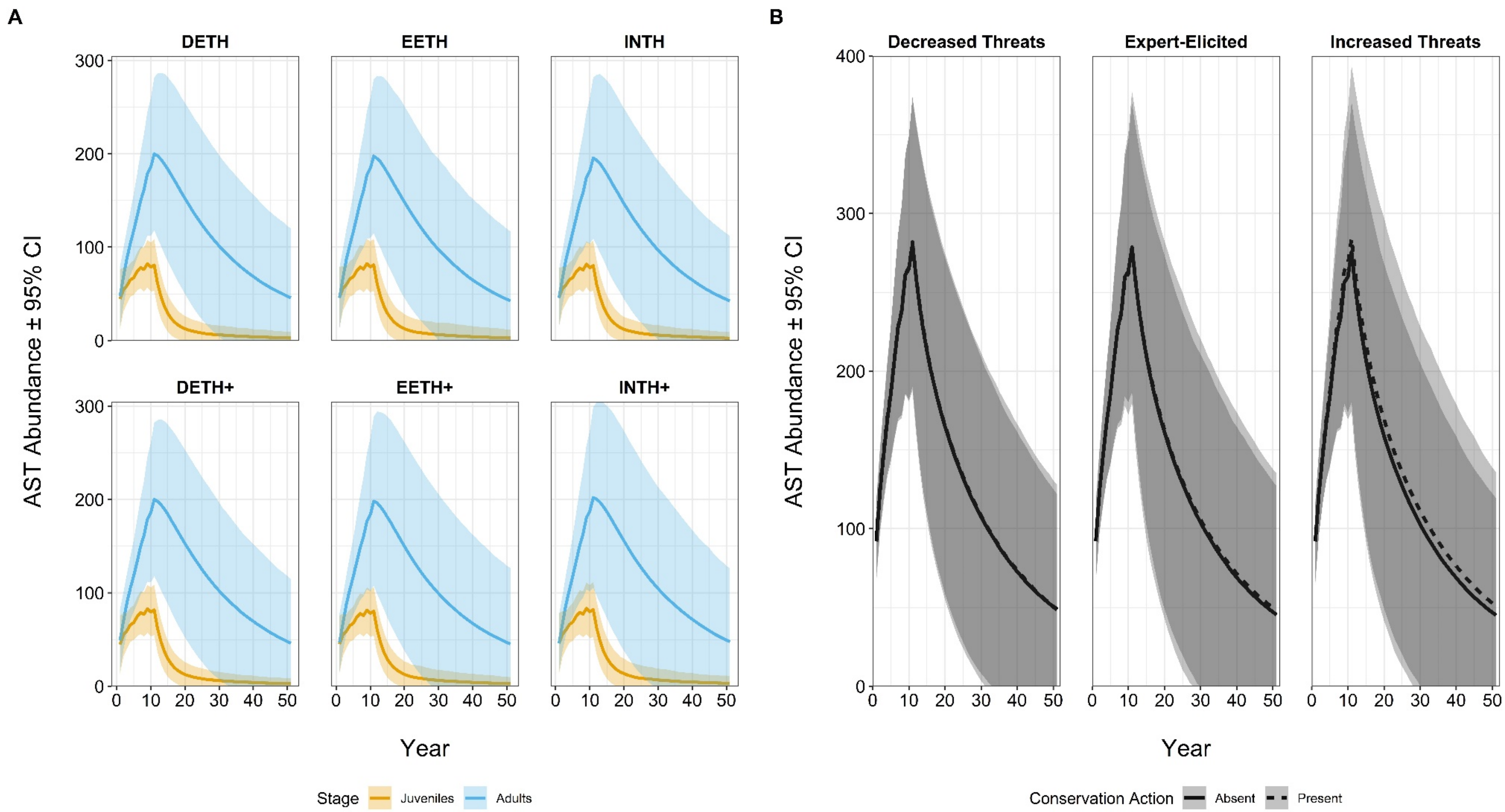


Figure E5. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Northern Mississippi – East analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the shaded areas reflect the 95% confidence intervals (CI). In (A) each panel represents one of six scenarios, varying by three threat levels (Decreased [DE], Expert-Elicited [EE], or Increased [IN]) across columns, and conservation actions absent (TH; top row) or present (TH+; bottom row). The orange line shows stage-specific abundance for juveniles and adults in blue. The columns in (B.) indicate the scenario's threat level (increasing from left to right). The solid and dashed lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the analysis unit-specific quasi-extirpation threshold (<5% of total abundance in Year 1) is given by the thin flat line.

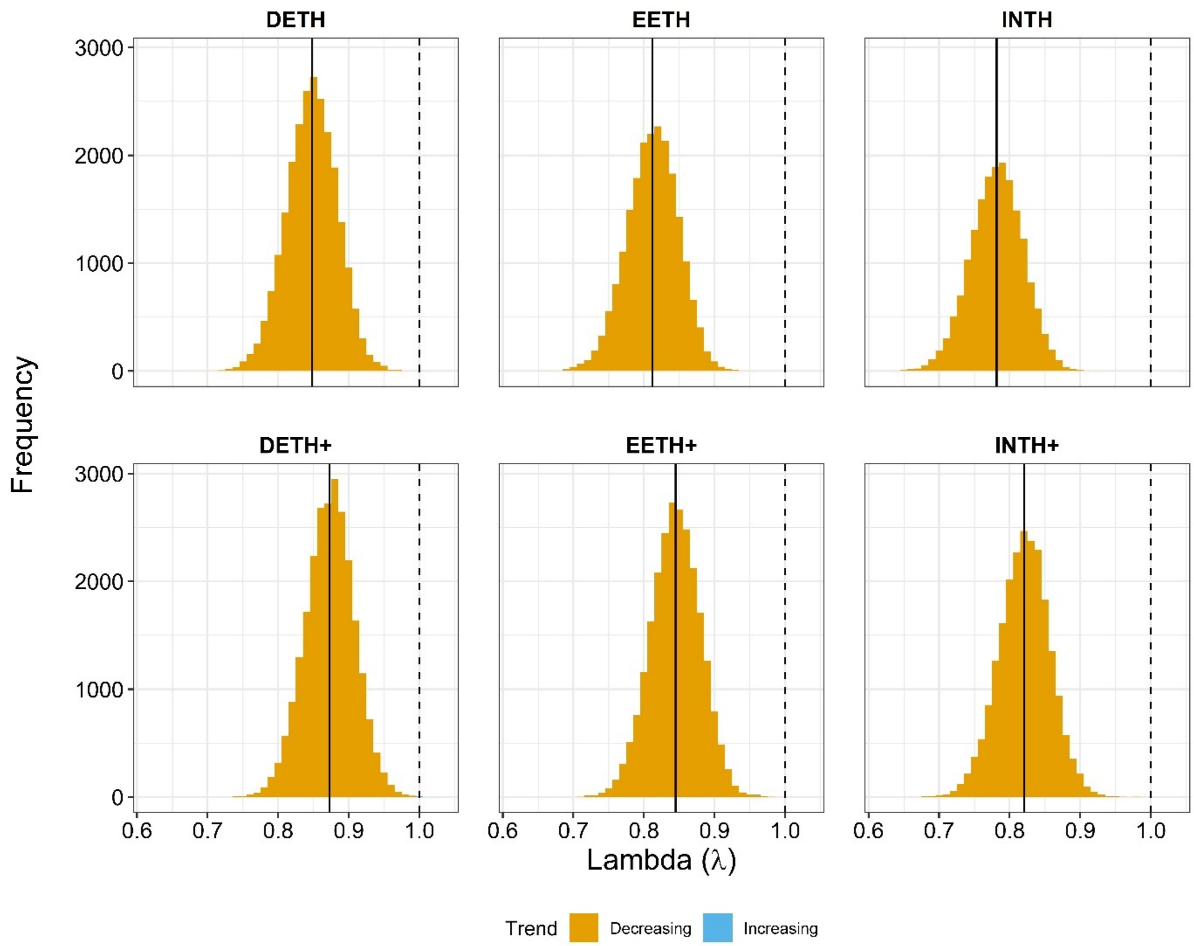


Figure E6. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Southern Mississippi – East analysis unit. Each panel represents a different scenario in which the threat level increases from left to right (decreased [DE], expert-elicited [EE], increased [IN]) across columns, whereas conservation action absent scenarios are in the top row (TH) whereas present (TH+) scenarios on the bottom. Each scenario generated a maximum of 25K projection matrices (50-year projection repeated for 500 iterations), though if the population declined to zero during an iteration the projection stopped and began the next iteration. The stochastic simulation framework randomly drew baseline demographic parameters (Table E1), threat specific parameter reductions (Table E3), and analysis unit-specific spatial extents (Table E4) of threats at each iteration and timestep that created variation among the projection matrices and their associated λs. The solid vertical line represents the λ distribution mean, whereas the dashed vertical reference line is at λ=1 to separate values of λ that indicate a decreasing population (λ<1; orange) from those that indicate stable or increasing population (; λ>1; blue).

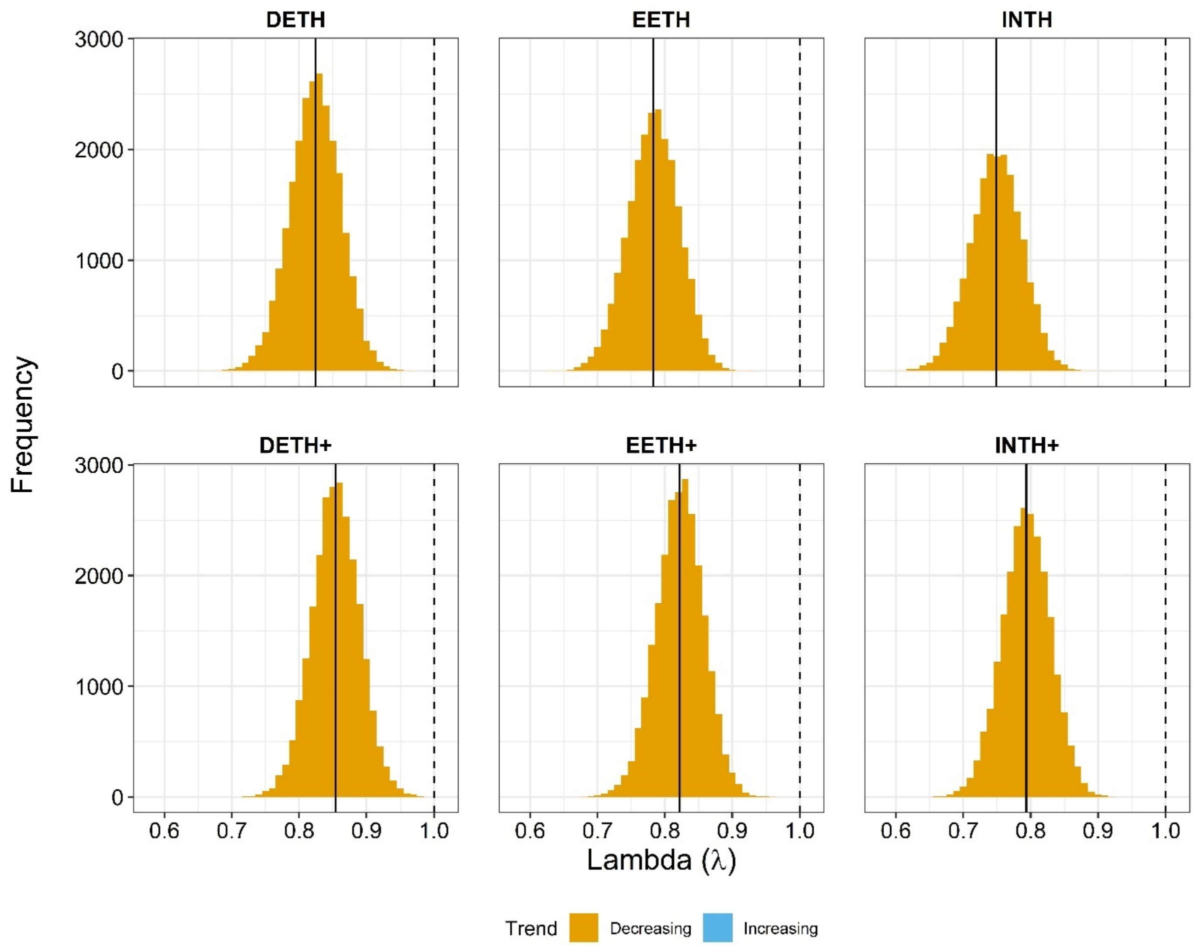


Figure E7. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Alabama analysis unit. Each panel represents a different scenario in which the threat level increases from left to right (decreased [DE], expert-elicited [EE], increased [IN]) across columns, whereas conservation action absent scenarios are in the top row (TH) whereas present (TH+) scenarios on the bottom. Each scenario generated a maximum of 25K projection matrices (50-year projection repeated for 500 iterations), though if the population declined to zero during an iteration the projection stopped and began the next iteration. The stochastic simulation framework randomly drew baseline demographic parameters (Table E1), threat specific parameter reductions (Table E3), and analysis unit-specific spatial extents (Table E4) of threats at each iteration and timestep that created variation among the projection matrices and their associated λs. The solid vertical line represents the λ distribution mean, whereas the dashed vertical reference line is at λ=1 to separate values of λ that indicate a decreasing population (λ<1; orange) from hose that indicate stable or increasing population (; λ>1; blue).

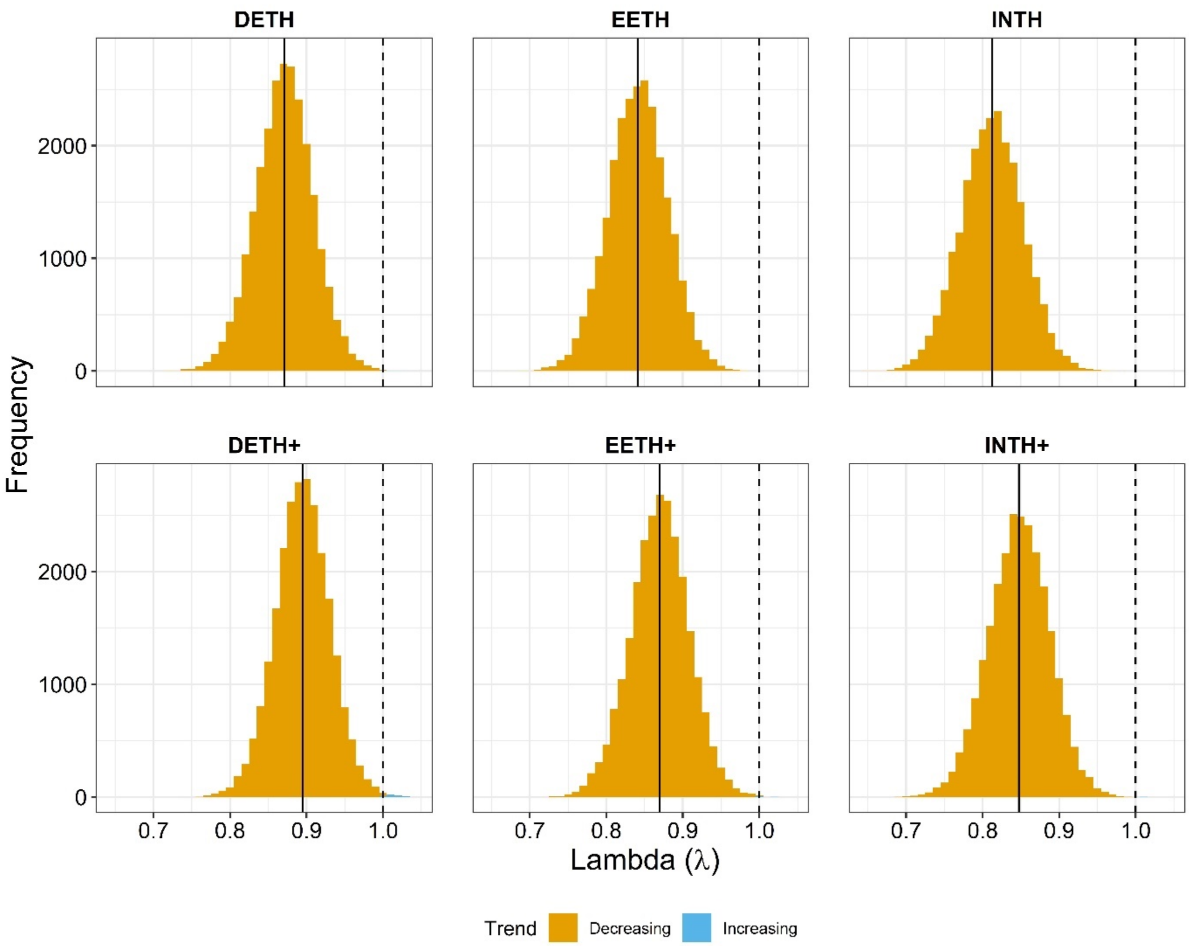


Figure E8. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Apalachicola analysis unit. Each panel represents a different scenario in which the threat level increases from left to right (decreased [DE], expert-elicited [EE], increased [IN]) across columns, whereas conservation action absent scenarios are in the top row (TH) whereas present (TH+) scenarios on the bottom. Each scenario generated a maximum of 25K projection matrices (50-year projection repeated for 500 iterations), though if the population declined to zero during an iteration the projection stopped and began the next iteration. The stochastic simulation framework randomly drew baseline demographic parameters (Table E1), threat specific parameter reductions (Table E3), and analysis unit-specific spatial extents (Table E4) of threats at each iteration and timestep that created variation among the projection matrices and their associated λs. The solid vertical line represents the λ distribution mean, whereas the dashed vertical reference line is at λ=1 to separate values of λ that indicate a decreasing population (λ<1; orange) from those that indicate stable or increasing population (; λ>1; blue).

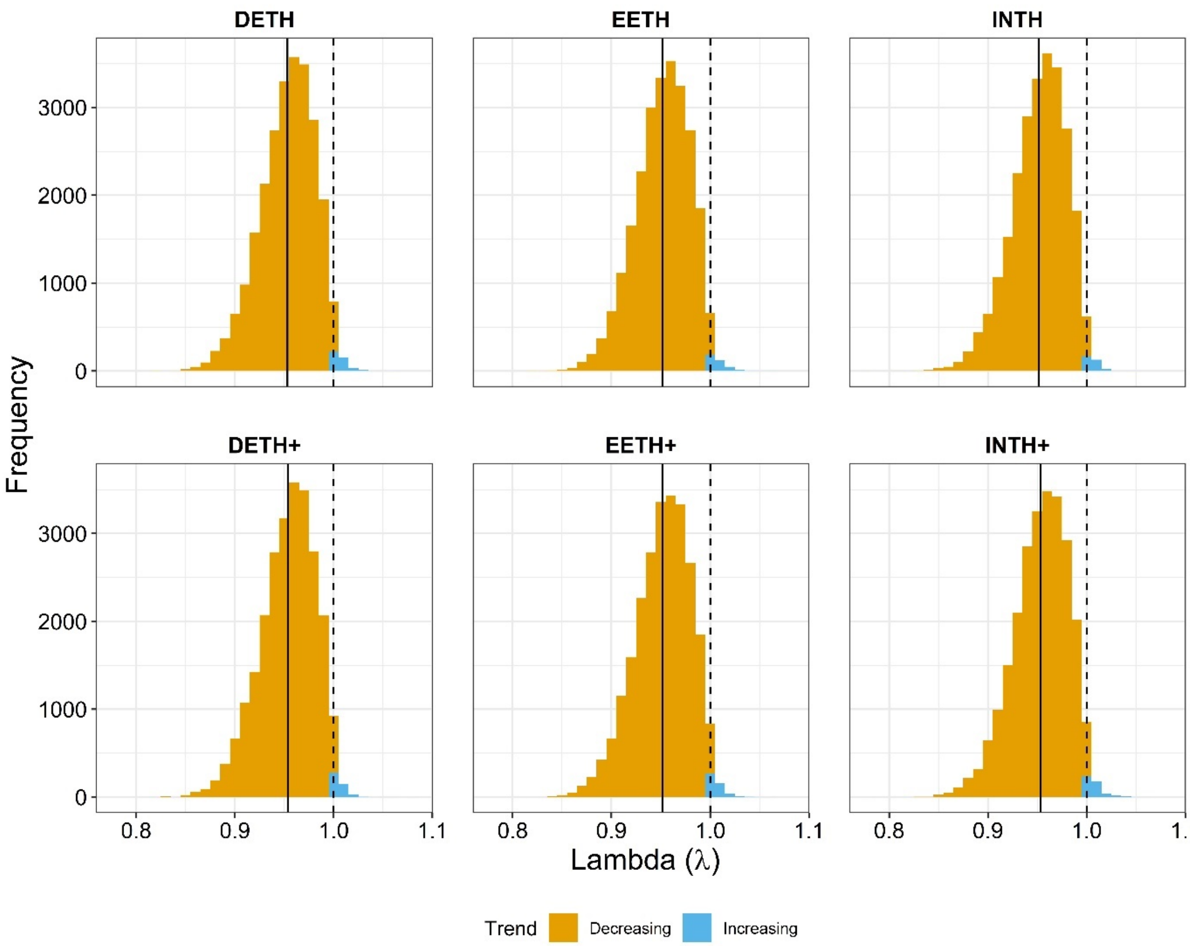


Figure E9. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Northern Mississippi – East analysis unit. Each panel represents a different scenario in which the threat level increases from left to right (decreased [DE], expert-elicited [EE], increased [IN]) across columns, whereas conservation action absent scenarios are in the top row (TH) whereas present (TH+) scenarios on the bottom. Each scenario generated a maximum of 25K projection matrices (50-year projection repeated for 500 iterations), though if the population declined to zero during an iteration the projection stopped and began the next iteration. The stochastic simulation framework randomly drew baseline demographic parameters (Table E1), threat specific parameter reductions (Table E3), and analysis unit-specific spatial extents (Table E4) of threats at each iteration and timestep that created variation among the projection matrices and their associated λs. The solid vertical line represents the λ distribution mean, whereas the dashed vertical reference line is at λ=1 to separate values of λ that indicate a decreasing population (λ<1; orange) from those that indicate stable or increasing population (; λ>1; blue).

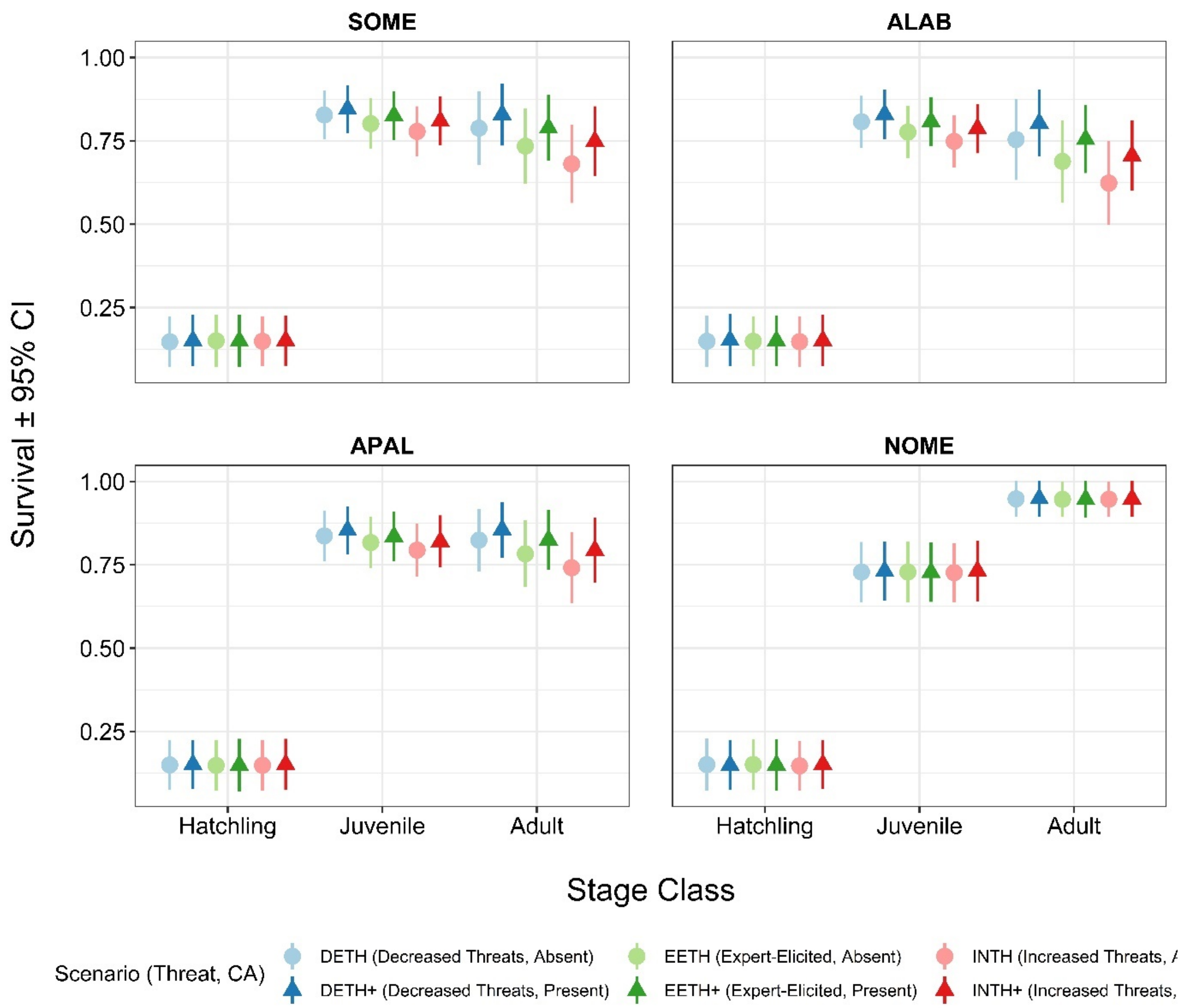


Figure E10. Mean stage class-specific alligator snapping turtle (*Macrochelys temminckii*; AST) survival parameters and their associated 95% confidence intervals (CI) for each analysis unit: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). The matrix model used to project AST population dynamics was comprised of two stages (juveniles and adults), though the hatchling (neonate) survival parameter was contained within the adult fecundity element (F_A, Eq. 1, Table E1) and was exposed to threats in the model (Tables E3, E4). Within each panel and stage class, the individual points reflect different scenarios that differ by decreased (blue), expert-elicited (green), or increased (red) threat levels, as well as the absence (circles, light colors) or presence (triangles, bold colors) of conservation action (TH or TH+, respectively in the legend).

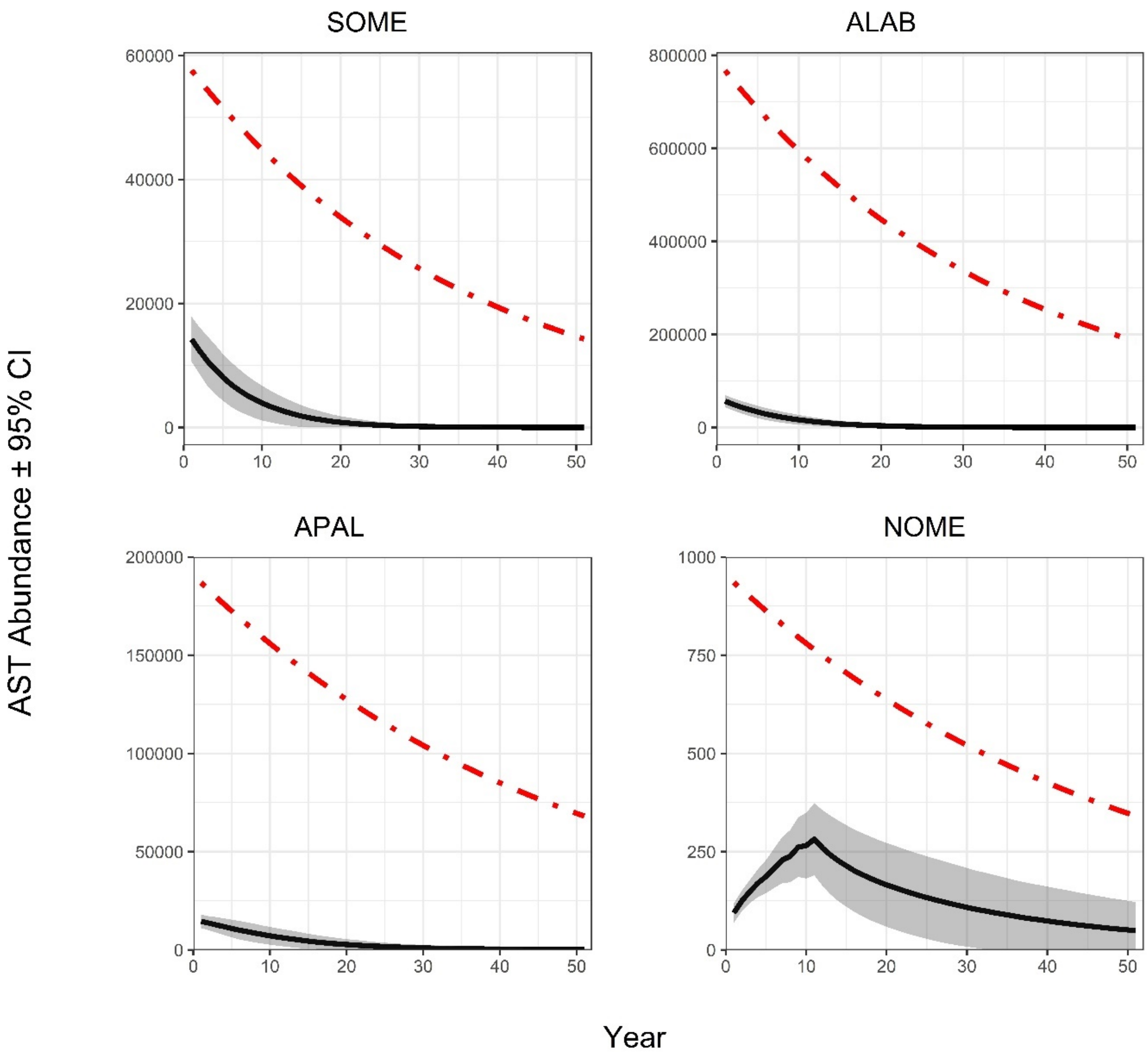


Figure E11. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) female total abundance over a 50-year period under the decreased threat with conservation action (DETH+) scenario for each analysis unit. Analysis unit abbreviations are listed above each panel and include: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). The solid black lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI), whereas the dot-dashed red line is the unit's population ceiling. The initial population ceiling was set at the expert-elicited current maximum AST abundance +25%, adjusted to include non-hatchling females only. The population ceiling was annually reduced by the unit's habitat loss rate (HLR in Table E3) using Equation 6.

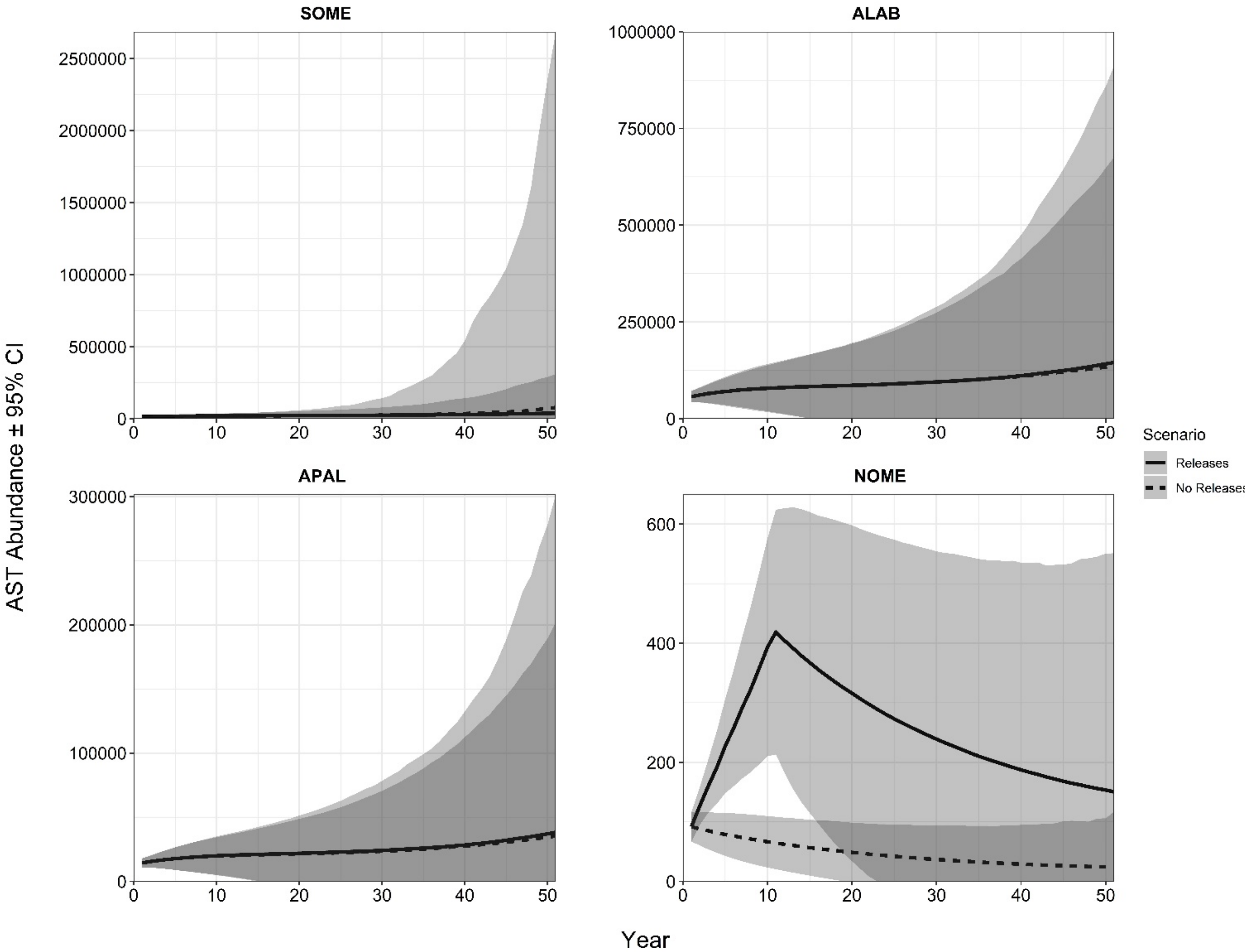


Figure E12. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) female total abundance over a 50-year period under the baseline scenario, with (solid line) and without (dashed line) head start and adult releases. The baseline scenarios used demographic parameters listed in Table E1, sampled from a distribution in each iteration. Analysis unit abbreviations are listed above each panel and include: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). The lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI).

Mean Predicted Future Abundances

Table E12. Model-predicted mean abundances and standard deviations at 5 year intervals for alligator snapping turtles in five analysis units (ALAB = Alabama, APAL = Apalachicola, NOME = Northern Mississippi – East, SOME = Southern Mississippi – East) under six scenarios (DETH = decreased threats, EETH = expert-elicited threats, INTH = increased threats, + = conservation actions present). Results are from a female-only, stage-structured stochastic matrix model run for 50 years with 500 iterations for each analysis unit/scenario combination.

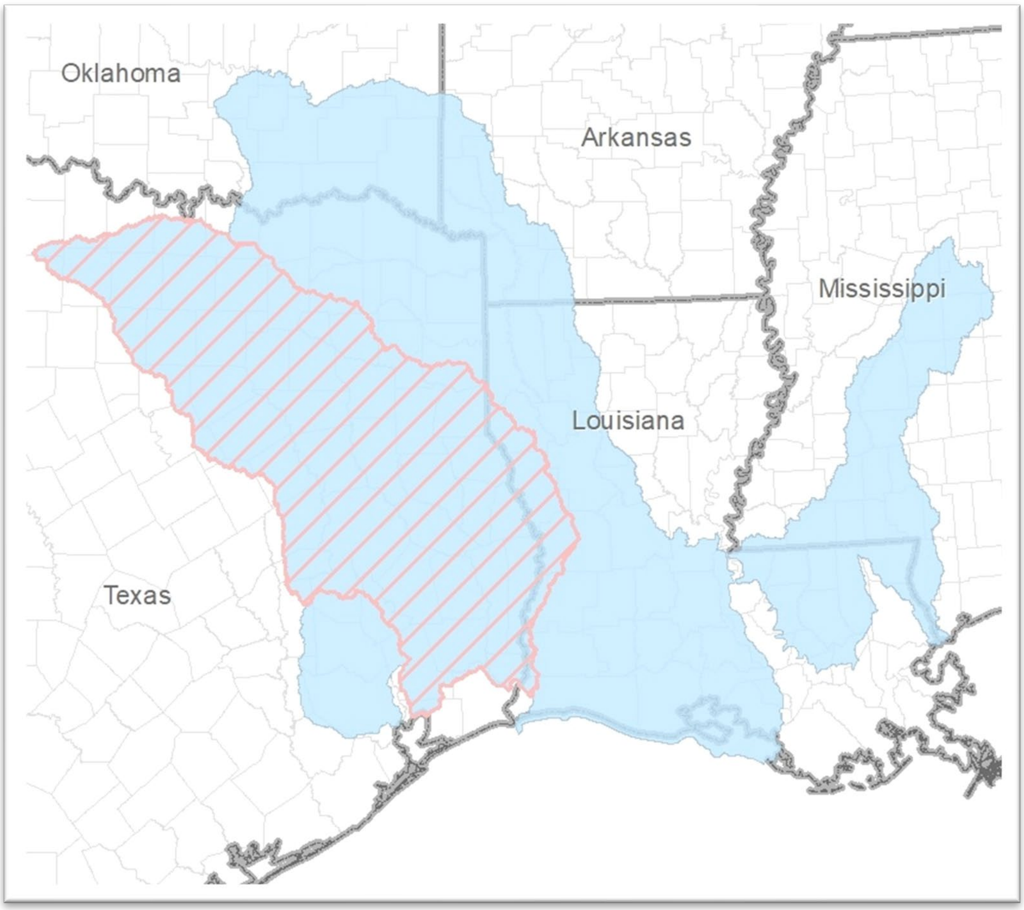
Scenario	Analysis Unit	Year	Abundance	SD	Scenario	Analysis Unit	Year	Abundance	SD
DETH	ALAB	0	56627.9	7146.7	DETH+	ALAB	0	56668.1	6717.7
DETH	ALAB	5	23787.8	5696.7	DETH+	ALAB	5	29015.8	6562.9
DETH	ALAB	10	9634.7	3530.0	DETH+	ALAB	10	14373.4	4911.8
DETH	ALAB	15	3737.1	1766.0	DETH+	ALAB	15	6782.9	2951.8
DETH	ALAB	20	1444.7	843.0	DETH+	ALAB	20	3180.6	1679.4
DETH	ALAB	25	562.7	406.5	DETH+	ALAB	25	1499.7	957.0
DETH	ALAB	30	222.8	190.7	DETH+	ALAB	30	713.9	543.8
DETH	ALAB	35	85.2	82.5	DETH+	ALAB	35	336.2	290.3
DETH	ALAB	40	33.3	35.9	DETH+	ALAB	40	161.0	163.9
DETH	ALAB	45	13.1	15.2	DETH+	ALAB	45	77.9	90.0
DETH	ALAB	50	5.2	7.1	DETH+	ALAB	50	37.6	49.8
EETH	ALAB	0	56695.8	6726.7	EETH+	ALAB	0	57455.2	7342.0
EETH	ALAB	5	18377.0	4681.1	EETH+	ALAB	5	24714.8	5951.8
EETH	ALAB	10	5673.7	2077.0	EETH+	ALAB	10	10032.4	3498.9
EETH	ALAB	15	1699.3	807.5	EETH+	ALAB	15	3927.9	1750.6
EETH	ALAB	20	509.3	299.3	EETH+	ALAB	20	1520.2	837.5
EETH	ALAB	25	154.1	107.1	EETH+	ALAB	25	594.5	388.4
EETH	ALAB	30	46.8	38.5	EETH+	ALAB	30	231.5	176.0
EETH	ALAB	35	14.5	13.6	EETH+	ALAB	35	90.0	79.7
EETH	ALAB	40	4.5	5.0	EETH+	ALAB	40	35.6	36.7
EETH	ALAB	45	1.3	1.9	EETH+	ALAB	45	14.3	16.4
EETH	ALAB	50	0.2	0.7	EETH+	ALAB	50	5.7	7.6
INTH	ALAB	0	56707.4	7237.7	INTH+	ALAB	0	56699.6	7088.0
INTH	ALAB	5	14204.1	4019.1	INTH+	ALAB	5	19918.7	5068.1
INTH	ALAB	10	3537.0	1393.7	INTH+	ALAB	10	6753.8	2459.5
INTH	ALAB	15	843.7	420.1	INTH+	ALAB	15	2175.2	1007.4
INTH	ALAB	20	204.9	126.6	INTH+	ALAB	20	700.1	408.0
INTH	ALAB	25	50.4	38.4	INTH+	ALAB	25	229.0	163.4
INTH	ALAB	30	12.4	11.0	INTH+	ALAB	30	74.9	62.1
INTH	ALAB	35	3.0	3.1	INTH+	ALAB	35	24.7	23.6
INTH	ALAB	40	0.6	1.0	INTH+	ALAB	40	8.2	9.3
INTH	ALAB	45	0.1	0.3	INTH+	ALAB	45	2.7	3.8
INTH	ALAB	50	0.0	0.0	INTH+	ALAB	50	0.8	1.6
DETH	APAL	0	14340.9	1733.1	DETH+	APAL	0	14496.0	1731.3
DETH	APAL	5	8959.6	2381.7	DETH+	APAL	5	10053.9	2395.1
DETH	APAL	10	5146.2	1936.2	DETH+	APAL	10	6572.4	2260.4
DETH	APAL	15	2775.0	1348.3	DETH+	APAL	15	4043.9	1795.5
DETH	APAL	20	1471.1	872.8	DETH+	APAL	20	2470.5	1339.5
DETH	APAL	25	783.7	546.6	DETH+	APAL	25	1492.0	979.9

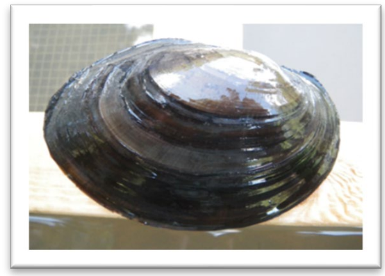
Scenario	Analysis Unit	Year	Abundance	SD	Scenario	Analysis Unit	Year	Abundance	SD
DETH	APAL	30	418.2	343.4	DETH+	APAL	30	911.8	729.3
DETH	APAL	35	222.8	213.4	DETH+	APAL	35	557.3	530.9
DETH	APAL	40	119.5	134.8	DETH+	APAL	40	343.6	382.3
DETH	APAL	45	64.9	85.3	DETH+	APAL	45	211.1	265.9
DETH	APAL	50	36.0	57.0	DETH+	APAL	50	132.5	197.3
EETH	APAL	0	14416.4	1861.8	EETH+	APAL	0	14441.7	1896.4
EETH	APAL	5	7609.9	2085.5	EETH+	APAL	5	8935.6	2338.4
EETH	APAL	10	3718.0	1518.4	EETH+	APAL	10	5106.5	1892.2
EETH	APAL	15	1680.7	859.8	EETH+	APAL	15	2760.4	1351.1
EETH	APAL	20	751.2	474.8	EETH+	APAL	20	1465.5	875.8
EETH	APAL	25	332.9	254.5	EETH+	APAL	25	778.3	559.4
EETH	APAL	30	147.9	138.1	EETH+	APAL	30	412.0	354.9
EETH	APAL	35	66.8	73.9	EETH+	APAL	35	218.6	220.8
EETH	APAL	40	30.4	42.0	EETH+	APAL	40	118.7	139.3
EETH	APAL	45	14.0	23.7	EETH+	APAL	45	65.4	90.3
EETH	APAL	50	6.5	13.3	EETH+	APAL	50	36.2	57.9
INTH	APAL	0	14671.3	1861.2	INTH+	APAL	0	14482.4	1937.7
INTH	APAL	5	6452.7	1824.4	INTH+	APAL	5	7923.0	2028.3
INTH	APAL	10	2642.3	1023.1	INTH+	APAL	10	3998.1	1417.5
INTH	APAL	15	991.3	506.7	INTH+	APAL	15	1869.2	835.5
INTH	APAL	20	370.5	239.8	INTH+	APAL	20	865.0	472.8
INTH	APAL	25	138.0	110.6	INTH+	APAL	25	392.4	258.9
INTH	APAL	30	51.5	50.0	INTH+	APAL	30	181.7	147.7
INTH	APAL	35	19.6	23.7	INTH+	APAL	35	84.7	81.9
INTH	APAL	40	7.6	11.2	INTH+	APAL	40	39.7	45.0
INTH	APAL	45	2.9	5.9	INTH+	APAL	45	18.9	24.9
INTH	APAL	50	1.0	2.7	INTH+	APAL	50	9.0	14.2
DETH	NOME	0	91.9	13.0	DETH+	NOME	0	93.9	12.7
DETH	NOME	5	207.2	25.7	DETH+	NOME	5	208.0	25.7
DETH	NOME	10	280.8	47.5	DETH+	NOME	10	281.8	46.8
DETH	NOME	15	200.8	55.4	DETH+	NOME	15	201.5	53.5
DETH	NOME	20	157.6	55.9	DETH+	NOME	20	158.3	54.5
DETH	NOME	25	126.6	54.8	DETH+	NOME	25	127.6	53.4
DETH	NOME	30	103.3	52.5	DETH+	NOME	30	103.9	50.2
DETH	NOME	35	84.7	50.2	DETH+	NOME	35	85.5	46.8
DETH	NOME	40	70.0	47.2	DETH+	NOME	40	70.8	44.0
DETH	NOME	45	58.1	43.4	DETH+	NOME	45	58.7	40.5
DETH	NOME	50	48.6	40.5	DETH+	NOME	50	49.0	37.2
EETH	NOME	0	92.3	13.3	EETH+	NOME	0	92.0	12.1
EETH	NOME	5	206.3	25.9	EETH+	NOME	5	206.4	27.1
EETH	NOME	10	278.6	46.9	EETH+	NOME	10	278.6	50.4
EETH	NOME	15	197.4	55.4	EETH+	NOME	15	199.0	60.5
EETH	NOME	20	153.7	57.0	EETH+	NOME	20	155.1	61.3
EETH	NOME	25	123.1	55.3	EETH+	NOME	25	124.6	60.3
EETH	NOME	30	99.6	54.3	EETH+	NOME	30	101.5	57.9
EETH	NOME	35	81.3	51.5	EETH+	NOME	35	83.3	54.6
EETH	NOME	40	66.3	48.2	EETH+	NOME	40	68.7	51.6

Scenario	Analysis Unit	Year	Abundance	SD
EETH	NOME	45	54.8	44.9
EETH	NOME	50	45.4	41.6
INTH	NOME	0	92.0	12.3
INTH	NOME	5	205.0	26.9
INTH	NOME	10	275.8	48.6
INTH	NOME	15	195.0	56.6
INTH	NOME	20	151.7	55.7
INTH	NOME	25	121.7	54.3
INTH	NOME	30	98.6	51.7
INTH	NOME	35	80.3	48.4
INTH	NOME	40	66.0	44.6
INTH	NOME	45	54.4	40.9
INTH	NOME	50	45.0	37.9
DETH	SOME	0	14127.8	1882.0
DETH	SOME	5	5918.6	1589.4
DETH	SOME	10	2463.8	964.0
DETH	SOME	15	952.9	484.5
DETH	SOME	20	365.8	225.9
DETH	SOME	25	142.7	109.2
DETH	SOME	30	55.0	51.1
DETH	SOME	35	21.2	21.8
DETH	SOME	40	8.8	10.9
DETH	SOME	45	3.7	5.3
DETH	SOME	50	1.5	3.0
EETH	SOME	0	14043.9	1825.9
EETH	SOME	5	4790.5	1467.4
EETH	SOME	10	1622.3	664.4
EETH	SOME	15	513.0	267.9
EETH	SOME	20	160.1	106.3
EETH	SOME	25	50.4	40.3
EETH	SOME	30	16.2	14.6
EETH	SOME	35	5.4	5.5
EETH	SOME	40	1.8	2.2
EETH	SOME	45	0.5	0.9
EETH	SOME	50	0.1	0.3
INTH	SOME	0	14210.1	1715.8
INTH	SOME	5	3910.3	1053.6
INTH	SOME	10	1104.4	405.5
INTH	SOME	15	282.4	139.9
INTH	SOME	20	72.3	43.9
INTH	SOME	25	18.8	14.0
INTH	SOME	30	5.2	4.5
INTH	SOME	35	1.4	1.6
INTH	SOME	40	0.2	0.6
INTH	SOME	45	0.0	0.2
INTH	SOME	50	0.0	0.0

Scenario	Analysis Unit	Year	Abundance	SD
EETH+	NOME	45	57.3	47.7
EETH+	NOME	50	48.2	44.4
INTH+	NOME	0	91.9	13.3
INTH+	NOME	5	208.7	29.3
INTH+	NOME	10	284.2	56.0
INTH+	NOME	15	205.4	64.6
INTH+	NOME	20	161.5	64.5
INTH+	NOME	25	130.5	62.2
INTH+	NOME	30	107.1	59.4
INTH+	NOME	35	88.5	56.0
INTH+	NOME	40	73.4	51.3
INTH+	NOME	45	60.8	47.3
INTH+	NOME	50	50.9	43.4
DETH+	SOME	0	14248.0	1859.1
DETH+	SOME	5	6975.4	1841.1
DETH+	SOME	10	3413.0	1306.2
DETH+	SOME	15	1569.6	801.1
DETH+	SOME	20	716.1	458.7
DETH+	SOME	25	321.0	246.3
DETH+	SOME	30	146.0	132.4
DETH+	SOME	35	66.9	69.8
DETH+	SOME	40	31.0	37.8
DETH+	SOME	45	14.9	21.4
DETH+	SOME	50	7.4	11.8
EETH+	SOME	0	14130.0	1689.8
EETH+	SOME	5	5940.4	1554.2
EETH+	SOME	10	2476.1	918.8
EETH+	SOME	15	959.8	454.1
EETH+	SOME	20	365.1	210.6
EETH+	SOME	25	137.3	93.0
EETH+	SOME	30	51.6	39.2
EETH+	SOME	35	20.0	17.8
EETH+	SOME	40	8.1	7.9
EETH+	SOME	45	3.3	3.5
EETH+	SOME	50	1.3	1.7
INTH+	SOME	0	14254.9	1739.3
INTH+	SOME	5	5315.1	1511.6
INTH+	SOME	10	1923.4	751.1
INTH+	SOME	15	642.5	336.4
INTH+	SOME	20	212.4	142.1
INTH+	SOME	25	70.1	56.6
INTH+	SOME	30	23.8	24.6
INTH+	SOME	35	8.3	10.6
INTH+	SOME	40	3.0	4.6
INTH+	SOME	45	1.0	2.4
INTH+	SOME	50	0.2	1.1

Species Status Assessment Report for Two Freshwater Mussels: Louisiana Pigtoe (*Pleurobema riddellii*) and Texas Heelsplitter (*Potamilus amphichaenus*)





TEXAS HEELSPLITTER (*POTAMILUS AMPHICHAENUS*) (HISTORICAL RANGE SHOWN IN RED CROSS-HATCH)

LOUISIANA PIGTOE (*PLEUROBEMA RIDDELLII)* (HISTORICAL RANGE SHOWN IN BLUE)

Version 1.2

February 2022
U.S. Fish and Wildlife Service
Region 2
Albuquerque, NM

This document was prepared by the U.S. Fish and Wildlife Service's (USFWS) Louisiana Pigtoe and Texas Heelsplitter Mussel SSA Team and supporting staff, comprised of the core USFWS team (Jacob Lewis, Robert Allen, Susan Oetker, Erik Orsak, and Gary Pandolfi), and individuals from a variety of state and federal agencies, including Charrish Stevens, David Martinez, Chris Davidson, Dave Oster, Amy Trahan, Matthew Johnson, Mike Dick, and Matthew Wagner from the USFWS, Clint Robertson from the Texas Parks and Wildlife Department, Bill Posey from the Arkansas Game and Fish Commission, and Beau Gregory, Jared Streeter, and Kerui Lejeune from the Louisiana Department of Wildlife and Fisheries.

The following conservation partners, researchers, and independent peer reviewers provided valuable input on the content and accuracy of this SSA report: Dr. Charles Randklev, Dr. John Harris, Dr. Kentaro Inoue, Dr. Neil Ford, Lance Williams, Debra Bills, Chis Harper, Glenn Clingenpeel, Webster Mangham, Terry Corbett, Mark Fisher, Colin McDonald, Jennifer M. Khan, and Stephanie Ni.

The core USFWS team wishes to thank the above individuals, as well as the researchers listed in the Literature Cited section, for their time, commitment, and support, without which this report would not be possible.

While this Species Status Assessment Report contains content and contributions from various experts who provided assistance in the development of this report, this is a Service document and the analysis, assumptions, opinions, and conclusions reported herein represent those of the Service. This evaluation is based on, and limited by, the information that was available at the time of the writing of this report.

Suggested reference:

U.S. Fish and Wildlife Service. 2022. Species status assessment report for two freshwater mussels: Louisiana Pigtoe (*Pleurobema riddellii*) and Texas Heelsplitter (*Potamilus amphichaenus*). Version 1.2. February 2022. Arlington, Texas.

TABLE OF CONTENTS

CHAPTER 1. INTRODUCTION

The U.S. Fish and Wildlife Service (Service) is responsible for identifying species in need of protection under the Endangered Species Act of 1973, as amended (Act), due to ongoing threats such as habitat loss, and increasing concerns that a species may become extinct. The framework used by the Service to review the status of a species, known as a Species Status Assessment (SSA, Smith et al. 2018, entire), is intended to be an in-depth review of the species' biology, an evaluation of its biological status and threats to survival, and an assessment of the resources and conditions needed to maintain long-term viability. Information contained in the SSA Report is used to support a decision by the Service as to whether a species should be listed as threatened or endangered, and thereby afforded protection under the Act. If listing is warranted, the SSA Report can be updated as new information on a species becomes available and continues to support a myriad of other regulatory actions under the Act, such as recovery, Section 7 consultation, Section 10 permits, and reclassification decisions.

This document contains information collected as part of a status review of two freshwater mussels occurring in east Texas, the Louisiana Pigtoe (*Pleurobema riddellii*) and Texas Heelsplitter (*Potamilus amphichaenus*). Both species were petitioned for federal listing under the Act in 2007 by Forest Guardians, which resulted in substantial 90-day findings published in 2009. The Louisiana Pigtoe and Texas Heelsplitter are freshwater mussels in the Family *Unionidae*. Like most mussels, they occur in gravel and coarse sandy substrates of rivers, streams, and in the case of the Texas Heelsplitter, reservoirs. Mussels are filter feeders that rely on natural, high quality (pollutant free) flowing water of sufficient volume to support their life cycle, and that of their host fishes, which are essential for reproduction. Previous status reviews indicated these two freshwater mussel species face threats including habitat loss, changes to water quality, changes to hydrology, and riverbank destabilization (USFWS 2009a, p. 66889 and USFWS 2009b, p. 66265). Although both species are found in east Texas rivers, the range of the Louisiana Pigtoe is more expansive, extending into portions of east Oklahoma, southeast Arkansas, south Louisiana, and west Mississippi. The Texas Heelsplitter is currently known to occur in portions of three major river basins in Texas (Trinity, Neches, and Sabine), and the Louisiana Pigtoe currently occupies areas within five states across seven major river basins (San Jacinto, Neches, Sabine, Big Cypress-Sulphur, Red, Calcasieu-Mermentau, and Pearl). This SSA Report will refer to the species individually by common name and by scientific name (i.e., genus and specific epithet), where appropriate.

The Service will use this SSA Report to form the biological basis for whether these two freshwater mussel species warrant protection under the Act. Importantly, the SSA Report is not a decisional document, rather it provides a review of available information strictly related to the biological status of the species. A listing decision is made by the Service after reviewing this document and all relevant laws, regulations, and policies, and after the results of a proposed decision are announced in the Federal Register, with appropriate opportunity for public input. If listing is not warranted, the Service will continue to support conservation efforts, where appropriate. If listing is warranted, there are two possible outcomes based on both the level and timing of threats, including 1) Endangered – defined as a species that is in danger of extinction throughout all or a significant portion of its range (i.e., risk of extinction is high and imminent), or 2) Threatened – defined as a species that is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range (i.e., risk of extinction is high but not imminent).

For the purpose of this assessment, we generally define viability as the ability of Louisiana Pigtoe and Texas Heelsplitter to sustain populations in natural river systems over time. Using the SSA framework (Figure 1.1), we consider what these species need to maintain viability by characterizing the status of each species in terms of its resiliency, redundancy, and representation (i.e., the 3Rs; Smith et al. 2018, entire). The 3Rs are defined as:

Resiliency reflects a species' ability to withstand stochastic events (e.g., droughts, floods). Demographic measures that reflect the health of each population, such as fecundity (e.g., birth rate), survival, and population size, are some of the metrics used to evaluate resiliency. A resilient population is better able to withstand and recover from disturbances such as random fluctuations in birth rates (demographic stochasticity), variations in rainfall (environmental stochasticity), and the effects of anthropogenic activities.

Redundancy reflects a species' ability to withstand catastrophic events (such as a rare destructive natural event or episode involving many populations). Redundancy is about spreading the risk of such an event across multiple, resilient populations. As such, redundancy can be measured by the number and distribution of resilient populations across the range of the species.

Representation describes the ability of a species to adapt to changing nvironmental conditions over time. Representation is measured by the breadth of genetic or environmental diversity within and among populations across the range of the species by gauging the probability that a species is capable of adapting to environmental changes. The more representation, or diversity, a species has, the more it is capable of adapting to changes (natural or human-caused) in its environment. In the absence of species-specific genetic and ecological diversity information, we evaluate representation based on the extent and variability of habitat characteristics across the geographical range.

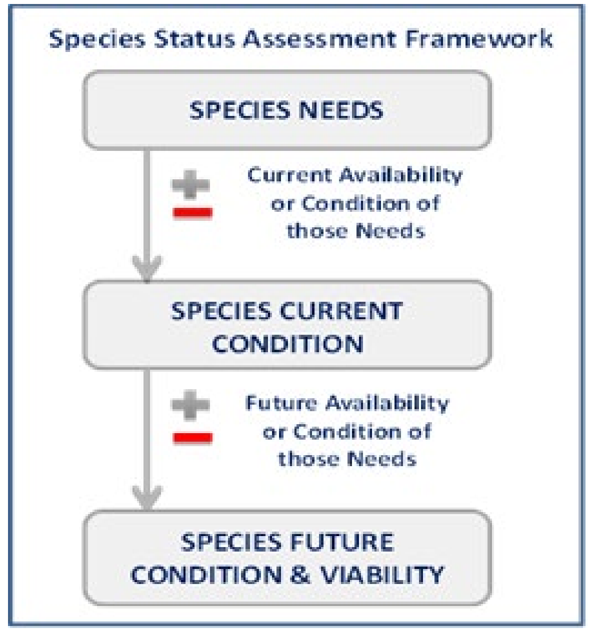


Figure 1.1. *Species Status Assessment Framework (USFWS, 2016).*

To evaluate the biological status of Louisiana Pigtoe and Texas Heelsplitter, both currently and into the future, we assessed a range of conditions to allow us to consider the species' resiliency, redundancy, and representation. This SSA Report provides a thorough assessment of existing information on these species, including the biology and natural history, demographic risks, stressors, and limiting factors in the context of determining their viability and risk of extinction, as well as estimates of how these variables will change in the future.

The format for this SSA Report includes: a description of the resource needs of individuals (Chapter 2); current and historical species distribution, and factors affecting population resiliency, redundancy, and representation (Chapter 3); estimates of current condition (Chapter 4); risk factors affecting species viability (Chapter 5); and estimates of future condition and population viability (Chapter 6). This document is a compilation of the best scientific and commercial information available, and a description of past, present, and likely future risk factors (i.e., threats) to Louisiana Pigtoe and Texas Heelsplitter.

Appendix A includes all references cited, which are available upon request, in portable document format (pdf), from the Arlington Texas Ecological Services Field Office[1]. Appendix B contains Cause and Effects Tables, which evaluate the stressors to the species historically and into the future. Appendix C contains detailed narratives, tables, and maps for each population based on our analysis and model output for future condition. Appendix D contains descriptions for select Indices of Hydrologic Alteration (IHA) used as part of our analysis.

[1] 2005 NE Green Oaks Blvd, Suite 140, Arlington, Texas, 76006 or call 817-277-1100

CHAPTER 2 - INDIVIDUAL NEEDS

This chapter reviews the basic biological and ecological information currently available for Louisiana Pigtoe and Texas Heelsplitter. This information includes taxonomy, phylogenetic relationships, morphology, and a description of known life history traits, with an emphasis on life history traits that are important to the viability of the species now and in the future. We then outline the resource needs at the level of the individual. Basic information is included about freshwater mussels in general, to Louisiana Pigtoe and Texas Heelsplitter in particular, and characteristics that are unique to individual species where appropriate. We caution that some aspects of the biology and life history of these species are not fully understood by the scientific community, and research is ongoing, therefore the following information is intended as an introduction to the basic needs of the species and is subject to change as new scientific information becomes available.

2.A. LOUISIANA PIGTOE AND TEXAS HEELSPLITTER – GENERAL INDIVIDUAL NEEDS

2.A.1. TAXONOMY OF LOUISIANA PIGTOE AND TEXAS HEELSPLITTER

Both Louisiana Pigtoe and Texas Heelsplitter belong to the Family Unionidae, also known as the naiads and pearly mussels, a group of bivalve mollusks that have been in existence for over 400 million years (Howells et al. 1996, p.1) and now represent over 600 species worldwide, of which over 250 species occur in North America (Strayer et al. 2004, p. 429; Lopes-Lima et al. 2018, pp. 2-3).

This report follows the most recently published and accepted taxonomic treatment of North American freshwater mussels as provided by Williams et al. (2017a, entire) which applies to the species assessed in this report.

PHYLUM	**Mollusca Linnaeus, 1758**
CLASS	**Bivalvia Linnaeus, 1758**
ORDER	**Unionida Gray, 1854**
FAMILY	**Unionidae Rafinesque, 1820**
SUBFAMILY	**Ambleminae Rafinesque, 1820**

Louisiana Pigtoe and Texas Heelsplitter, along with approximately 85% of North American mussel species, belong to the subfamily Ambleminae. Generally speaking, members of this group share the following common characteristics: 1) are typically slow-growing and commonly live for more than twenty years, with growth rates typically between 1–5mm/year, depending on conditions (Howells et al. 1996, p.17), 2) are frequently summer breeders (Howells et al. 1996, p. 9) although the Lampsilini (e.g., Texas Heelsplitter) typically spawn in fall and brood through the winter, 3) possess either unhooked or axe-head-type glochidia; may brood larvae in either all four or the outer two (lateral) demibranchs (McMahon and Bogan 2001, p. 342), 4) glochidia attach primarily to gills of the host fish (Barnhart et al. 2008, p. 375), 5) produce and store conglutinates in their mantle to facilitate rapid discharge of glochidia when fish attempt to feed (Barnhart et al. 2008, p. 375) and 6) free glochidia (not attached) may be released to water for hours or weeks prior to host infestation (Barnhart et al. 2008, p. 375).

2.A.2. Life History of Louisiana Pigtoe and Texas Heelsplitter

Freshwater mussels, including Louisiana Pigtoe and Texas Heelsplitter, have a complex life history (Figure 2.1) involving an obligate parasitic larval life stage, called glochidia, which are wholly dependent on host fish. As freshwater mussels are generally sedentary, dispersal is accomplished primarily through the behavior of host fish and their tendencies to travel upstream and against the current (positive rheotaxis) in rivers and streams. Mussels are broadcast spawners; males release sperm into the water column, which is taken in by the female through the incurrent aperture (the tubular structure used to draw water into the body of the mussel). The sperm fertilizes the eggs, which are held during maturation in an area of the gills called the marsupial chamber. The developing larvae remain in the marsupial chamber until they mature and are ready for release as glochidia, to attach on the gills, head, or fins of fishes (Vaughn and Taylor 1999, p. 913; Barnhart et al. 2008, pp. 371-373). Glochidia die if they fail to find a host fish, attach to the wrong species of host fish, attach to a fish that has developed immunity from prior infestations, or attach to the wrong location on a host fish (Neves 1991, p. 254; Bogan 1993, p. 599). Glochidia encyst (enclose in a cyst-like structure) on the host's tissue, draw nutrients from the fish, and develop into juvenile mussels weeks or months after attachment (Arey 1932, pp. 214-215). The glochidia will remain encysted for about a month through a transformation to the juvenile stage. Once transformed, the juveniles will excyst (release) from the fish and drop to the substrate. Freshwater mussel species vary in both onset and duration of spawning, how long developing larvae are held in the marsupial gill chambers, and which fish species serve as hosts. The mechanisms employed by mussel species to increase the likelihood of interaction between host fish and glochidia also vary by species.

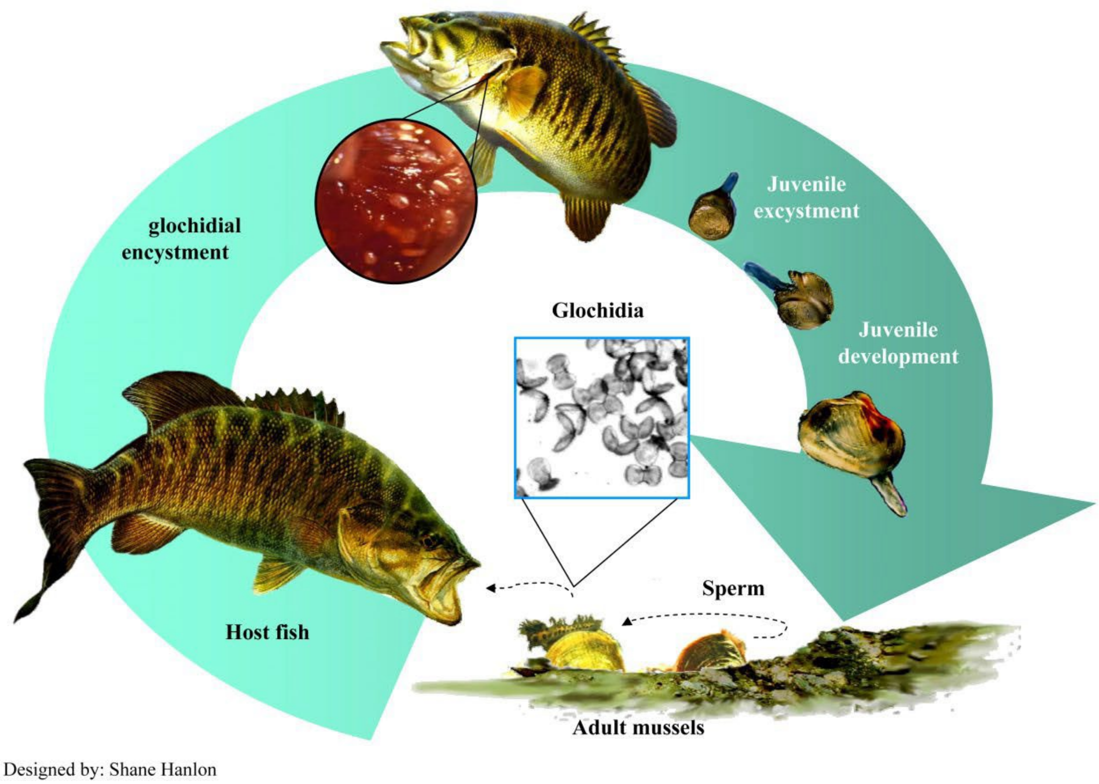


Figure 2.1. *Generalized freshwater mussel life cycle. Freshwater mussels, including the Louisiana Pigtoe and Texas Heelsplitter, have a complex life history involving an obligate parasitic larval life stage, called glochidia, which are wholly dependent on host fish. (Image courtesy of Shane Hanlon, USFWS).*

Although mature mussels are capable of moving short distances using a muscular foot appendage, they are generally sedentary and therefore experience their primary opportunity for dispersal and movement within a stream as glochidia attached to a mobile host fish (Smith 1985, p. 105). Upon release from the host, newly transformed juveniles drop to the substrate on the bottom of the stream. Those juveniles that drop in unsuitable substrates die because their immobility prevents them from relocating to more favorable habitat. Juvenile freshwater mussels burrow into interstitial substrates and grow to a larger size that is less susceptible to predation and displacement from high flow events (Yeager et al. 1994, p. 220). Adult mussels typically remain within the same general location where they drop off (excyst) their host fish as juveniles.

Host specificity can vary across mussel species, which may have specialized or generalized relationships with one or more taxa of fish. Mussels have evolved a wide variety of adaptations to facilitate transmission of glochidia to host fish including: 1) display of mantle lures that mimic fish or invertebrates, 2) packages of glochidia (conglutinates) that mimic worms, insect larvae, larval fish, or fish eggs, and 3) release of glochidia in mucous webs that entangle fish (Strayer et al. 2004, p. 431). Polymorphism of mantle lures and conglutinates frequently exists within mussel populations (Barnhart et al. 2008, p. 383), representing important adaptive capacity in terms of genetic diversity and ecological representation.

Freshwater mussels are generally considered to be long-lived and slow-growing (also see Haag and Rypel 2010, p. 2), with some individuals estimated to be decades or even centuries old based on measured growth rates (Strayer et al. 2004, p. 433). Due in part to their long life spans, recruitment is episodic and populations may be slow to recover from disturbance. Thin-shelled mussels (like Texas Heelsplitter) often live 4–10 years while thick-shelled mussels (like Louisiana Pigtoe) can live for 20–40 years, or longer (Howells et al. 1996, p.17).

Fast-growing species (like Texas Heelsplitter) may mature as early as their first year, while slow-growing species (like Louisiana Pigtoe) may take as long as 5–20 years to mature (Haag and Rypel 2010, p. 19). Fast-growing, short-lived species may be better adapted to more variable environments and therefore better suited to recover from high-mortality events than slower-growing long-lived species that are better adapted to more stable environments (Haag and Rypel 2010, p. 20). Nevertheless, growth rates and longevity often vary somewhat within and among populations of the same species.

2.A.3. Resource (Habitat) Needs of Individuals

Here we describe general habitat needs common to both Louisiana Pigtoe and Texas Heelsplitter. We describe the specific needs of each species in section 2.B (Species-Specific Needs of Louisiana Pigtoe and Texas Heelsplitter).

Louisiana Pigtoe and Texas Heelsplitter generally occur in medium to large streams and rivers, requiring 1) flowing water of sufficient quantity and quality (i.e., low or no contaminants) to meet their life history requirements and that of their host fishes, 2) adequate food supply, 3) habitat that provides refugia from both high- and low-flow events, 4) appropriate substrate that is generally characterized as stable and free of excessive fine sediment, 5) access to appropriate fish hosts, and 6) habitat connectivity (i.e., lack of impoundments and other barriers to fish passage) (Figure 2.2, Table 2.1).

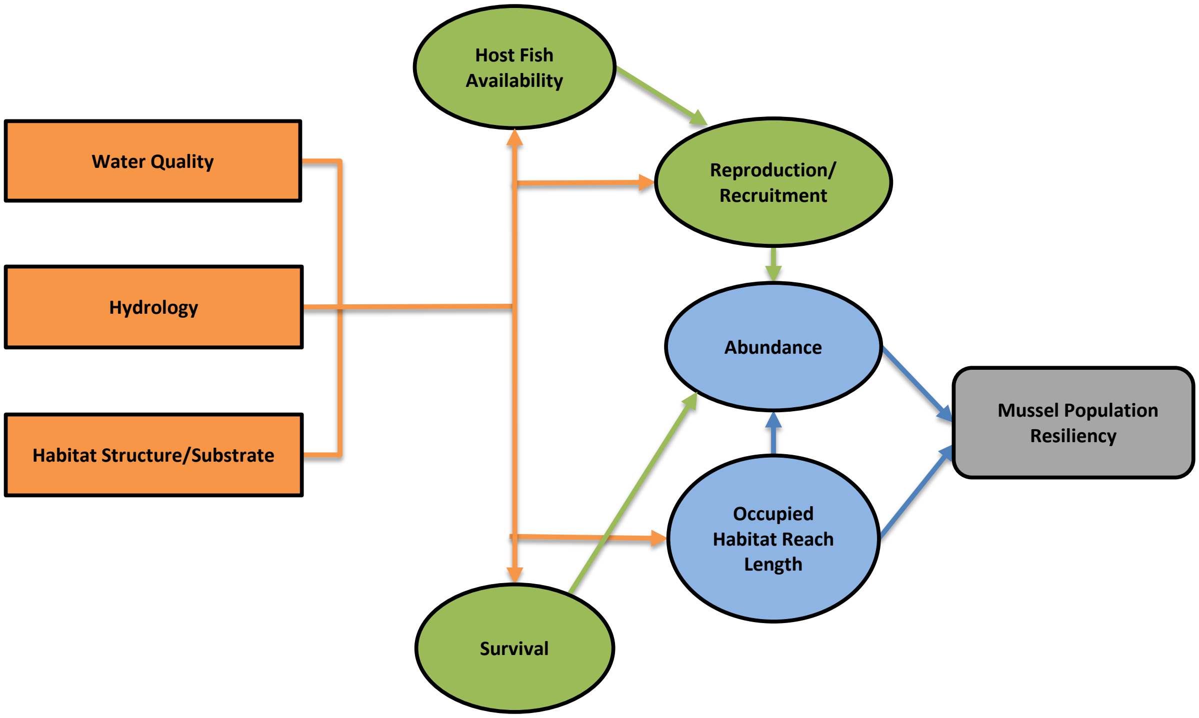


Figure 2.2. *Influence diagram representing the general population needs of Louisiana Pigtoe and Texas Heelsplitter. Habitat factors (orange boxes) influence demographic factors (green ovals) that affect population attributes (blue ovals) which influence overall resiliency of Louisiana Pigtoe and Texas Heelsplitter mussel populations.*

<u>Flowing water and protection from low-flow (dry or dewatering) events</u>. Louisiana Pigtoe are not adapted to lentic or non-flowing environments (e.g., reservoirs and impoundments) and do not persist or thrive in habitats unless they are free-flowing (lotic), such as unimpeded stream and river reaches. *Potamilus* species, including Texas Heelsplitter, are considered riverine lentic microhabitat specialists (Haag 2012, p. 135). Microhabitat refers to smaller habitat types within a larger habitat, such as localized areas that offer flow refugia (e.g., pooled areas behind structures) and interstitial spaces in substrate. Texas Heelsplitter are able to persist in lentic conditions, inhabiting several impoundments in north and east Texas. Both species of freshwater mussels in this report are considered to be lotic-habitat specialists, with Texas Heelsplitter tolerant of lentic-habitats.

Since both species evolved in, and are adapted to, free-flowing environments, they require (i.e., necessary to meet life-history requirements) unaltered rivers and streams that are free from major impoundments and other structures that impede flow. Free-flowing water provides appropriate oxygenation, nutrition, thermal buffering, and access to fish hosts for reproduction and dispersal. Louisiana Pigtoe and Texas Heelsplitter require adequate, but not excessively high flows, which may lead to scouring of suitable substrates.

Louisiana Pigtoe and Texas Heelsplitter generally do not tolerate exposure to a non-watered environments. Dewatering of occupied habitat can lead to reduced reproduction, health, body condition, or fitness, and can result in eventual death or stranding of mussels, along with exposure to predation. Dewatering can also affect, limit, or prevent mussel-host fish interaction. As such, these freshwater mussels require habitats and meso-habitats (*e.g.,* medium-sized habitats such as riffle, pool, and backwater areas within streams) that consistently provide minimum flows necessary to meet life history requirements. Preferred (i.e., selected for by the species) habitat for Louisiana Pigtoe and Texas Heelsplitter will maintain the necessary minimum flows and are protected from dewatering throughout the year. While some mussel species in other regions of Texas are more tolerant of dewatering, or have adaptations to avoid stranding (Bonner et al 2018, p. 196), Louisiana

Pigtoe and Texas Heelsplitter are not well adapted to persist in habitats subject to rapid and frequent dewatering (Mitchell et al. 2018, p. 16).

Protection from high-flow (scour) events. Louisiana Pigtoe and Texas Heelsplitter live in the substrate of rivers and streams, also known as the benthic environment (stream bed and bank habitats). Benthic habitats are typically comprised of a mix of sediments and cobble that are subject to periodic disturbance from high storm flows. The increased velocity of these storm flows can scour sediments and dislodge mussels, transporting them downstream to locations that may or may not be suitable habitat. Although mussels have adapted to increased flows associated with natural storm events, changing land uses such as increases in impervious cover and storm run-off from urban areas, may exceed their capacity to remain entrenched in substrates and result in mortality. Therefore, Louisiana Pigtoe and Texas Heelsplitter require microhabitats (flow refugia) that are naturally protected from scouring high-flow events that may occur during flood conditions. Some examples of flow refugia include boulders, crevices, bedrock shelves, bends, meanders, undercut banks, eddies, riffles, and living or dead vegetation (i.e., tree roots and coarse woody debris). In summary, Louisiana Pigtoe and Texas Heelsplitter require a balance between periods of low flow where the volume must be sufficient to meet their basic life history needs (discussed in the previous paragraph) and high flows that must not reach levels capable of scouring substrate, or otherwise degrading or destroying their habitat.

Water quality. Louisiana Pigtoe and Texas Heelsplitter require natural, high quality (pollutant free) water and are sensitive to both point and non-point source contaminants that deteriorate water quality and degrade their habitat. Contaminants are capable of altering the chemical, physical, and biological characteristics of a stream to the point where mussels or their host fish can no longer survive. A variety of pollutants can cause lethal and sub-lethal effects in aquatic biota, but mussel-specific data are generally lacking regarding their sensitivity to the more than 80,000 chemical compounds and their metabolites that are currently in commerce and are routinely released into the environment. Contaminants that are sometimes elevated in rivers and are a concern to mussel health include excess nutrients such as ammonia (NH_3), which is highly toxic to aquatic organisms, chemicals related to wastewater disinfection such as chlorine (Cl), trace metals like copper or cadmium (March et al. 2007, p. 270, Wang et al. 2010, p. 2057), dissolved solids (e.g., salinity), pharmaceuticals and personal care products (e.g., musks, fragrances, growth hormones, estradiol), and a variety of pesticides commonly used for residential and commercial applications; these pollutants, individually or collectively (i.e., synergism) can interfere with the ability of mussels or their host fishes to feed, breed, or otherwise meet their life history needs (Cope et al. 2008, p. 452). Augspurger et al. (2003) estimated a safe range of ammonia concentrations for all mussel life stages of 0.3-0.7 mg/L total ammonia nitrogen (TAN) at pH 8 (p. 2574) and noted that "sediment pore-water concentrations of ammonia typically exceed those of overlying surface water" (p. 2574). Healthy mussel populations need natural, high quality (pollutant free) water that is free of pollutants, has appropriate water chemistry including desirable oxygenation (generally expressed as mg/L dissolved oxygen), and is within appropriate upper and lower thermal limits (Khan et al. 2019, entire). It is worth noting that water quality and water quantity are interrelated and interdependent, so as water quantity decreases, the concentration of pollutants introduced to streams generally increases as does the likelihood that pollutants may reach levels harmful to aquatic biota.

Firm and stable substrate. Since freshwater mussels live in the substrate of benthic environments, the composition of the substrate material is vital to their ability to properly anchor and remain firmly in place. A firm and stable substrate comprised of the appropriate mix of materials is necessary for mussels to withstand changes in stream flow such as perturbations associated with storm events and prevent transport downstream. Sediments such as shifting sands and unconsolidated silts generally do not provide appropriate anchoring substrate, and thus appropriate habitat, for Louisiana Pigtoe and Texas Heelsplitter.

Nutrition and food supply. Adult freshwater mussels, including Louisiana Pigtoe and Texas Heelsplitter, are filter-feeders, siphoning suspended phytoplankton, zooplankton, rotifers, protozoans, detritus and dissolved organic matter from the water column (Strayer et al. 2004, p. 430) and from sediment; juvenile mussels are capable of using their foot to collect food items from sediments (pedal feeding; Vaughn et al. 2008, pp. 409-411). Glochidia derive what little nutrition they need from their obligate fish hosts (Barnhart et al. 2008, p. 372). Stable isotope studies suggest some mussel species feed on coarse particulate organic matter (CPOM) or bacteria and fungi associated with and decomposing the CPOM (Bonner et al. 2018, pp. 7, 215). Freshwater mussels must keep their shells open (gaped) to obtain food and facilitate gas exchange. They are sometimes able to sense perturbations to water quality and may respond by temporarily closing their shells (Bonner et al. 2018, p. 141). Food supply is not generally considered limiting in the environments inhabited by Louisiana Pigtoe and Texas Heelsplitter.

Fish hosts. Louisiana Pigtoe and Texas Heelsplitter have an obligate parasitic relationship with their respective host fishes. Nearly all freshwater mussels are unable to successfully reproduce or disperse in the absence of appropriate host fish. Host fish are necessary to facilitate dispersal and represent the only mechanism to do so in a free-flowing environment, although downstream movement of individuals may occur during high flow events if they become dislodged from the substrate. Both large and small run of river impoundments act as barriers to fish passage, and therefore inhibit mussel dispersal and recolonization. In some cases, freshwater mussels may be more tolerant of water quality degradation than their host fish. For example, mussels generally prefer dissolved oxygen concentrations greater than 3 mg/L and will begin to experience respiratory distress below approximately 2 mg/L (Bonner et al. 2018, p. 131), but dissolved oxygen below 5 mg/L is generally considered to be harmful to many fish species, and fish mortality is almost certain below 2 mg/L (Francis-Floyd 2011, p. 1).

Table 2.1. General life history and resource needs of Louisiana Pigtoe and Texas Heelsplitter.		
Life Stage	**Resource Need(s) - Habitat Requirements**	**Reference(s)**
All life Stages	**Water Quality**: Natural, high quality water with no or very low levels of harmful pollutants (i.e., potentially harmful constituents and toxicants are ideally absent, or at a minimum are below the tolerance limits of mussels, their host fishes, and prey items consumed by mussels or their hosts). Desirable conditions include, but are not limited to the following: - Natural, unaltered ambient water temperature; generally below 27°C (80.6°F) is considered protective, but sensitivity can vary by species and many species have not been tested for thermal tolerance - Dissolved oxygen generally > 3 mg/L or parts per million (ppm) -Low salinity/total dissolved solids (TDS) (e.g., trends for TDS and conductivity within watershed are stable (not increasing due to anthropogenic activity)) -No excess nutrients (e.g., nitrogen and ammonia levels are low (NH_3 below 0.3–0.7 mg/L NH_3-N at pH 8 and 25°C (77°F) cited by Augspurger as generally protective of unionids)) -No or low levels of copper, nickel, and other potentially harmful trace metals -No or low levels of pesticides, sulfate, chloride, potassium, and other potentially harmful constituents -No or low pollutants related to municipal and industrial wastewater or urban run-off, including pharmaceuticals, hormones, coliform bacteria, antibiotics, disinfection by-products (e.g., chlorine), petroleum hydrocarbons, and other environmental contaminants common to wastewater	Khan et al. 2019, entire. Gascho-Landis and Stoeckel 2016, p. 8; Gascho-Landis et al. 2013, pp. 76, 79; Augspurger et al. 2003, pp. 2569, 2571, 2574; Augspurger et al. 2007, p. 2,025; Cope et al. 2008, p. 454, 456.
	Water Quantity: Flowing water in sufficient quantity to support the life history requirements of mussels and their host fishes	Galbraith and Vaughn 2009, p. 46; Allen and Vaughn 2010, p. 390; Randklev et al. 2013b, p. 269. Randklev et al. 2017a, pp. 1, 5.
Gamete (broadcast sperm, egg development, to fertilization)	Sexually mature male and female mussels with appropriate water temperatures for spawning, fertilization, and brooding. Temperature is a primary cue for spawning. Low temperatures can suspend reproduction and high temperatures can lead to premature expulsion of glochidia	Haag 2012, pp. 38–39; Galbraith and Vaughn 2009, p. 45-46; Randklev et al. 2013a, pp. 3, 19.
Glochidium (from attachment through excystment)	Presence of host fish with sufficient flows to allow attachment, encystment, relocation, excystment, and dispersal of glochidia. Note that glochidia can be up to four times more sensitive to pollutants in water than juveniles	Barnhart et al. 2008, p. 372; Randklev et al. 2013b, p. 269.
	Stable substrates appropriate for burrowing	
Juvenile, sub-adult, and Adult (from excystment through maturity)	Stable substrates comprised of suitable sediment types and appropriate for burrowing	Allen and Vaughn 2010, pp. 384-385.
	Appropriate food source in adequate supply	

2.B. SPECIES-SPECIFIC NEEDS OF LOUISIANA PIGTOE AND TEXAS HEELSPLITTER

2.B.1. LOUISIANA PIGTOE, PLEUROBEMA RIDDELLII (LEA, 1862)

Figure 2.3. *Louisiana Pigtoe observed from Neches River, Angelina/Trinity Counties, Texas (USFWS photo).*

2.B.1.A. TAXONOMIC AND MORPHOLOGICAL DESCRIPTIONS

The Louisiana pigtoe (Figure 2.3) was originally described as the species *Unio riddellii* by Isaac Lea (1862, p. 228) from the Trinity River near the City of Dallas, Dallas County, Texas. The holotype (Smithsonian National Museum of Natural History, USNM 84635) was recently confirmed as Louisiana Pigtoe through genetic analysis (Randklev 2019b, p. 3). Simpson (1914), Vidrine (1993), and Howells et al. (1996) recognized the following synonyms:

Unio friersoni Wright (1896);
Quadrula friersoni (Wright) of Simpson (1914) and Frierson (1927);
Fusconaia friersoni (Wright) of Stern (1976) and Vidrine (1985);
Quadrula ridelli (Lea) of Strecker (1931);
Pleurobema riddelli (Lea) of Vidrine (1993), Howells et al. (1996), Turgeon et al. (1998), and others.

The current recognized scientific name for Louisiana Pigtoe is *Pleurobema riddellii*, and this report refers to it as such. The following taxonomic treatment follows Williams et al. (2017a, pp. 35, 42).

CLASS	**Bivalvia Linnaeus, 1758**
ORDER	**Unionida Gray, 1854**
FAMILY	**Unionidae Rafinesque, 1820**
SUBFAMILY	**Ambleminae Rafinesque, 1820**
TRIBE	**Pleurobemini Hannibal, 1912**
GENUS	***Pleurobema* Rafinesque, 1819**
SPECIES	***Pleurobema riddellii* (Lea, 1861)**

The Louisiana Pigtoe is a medium-sized freshwater mussel (shell lengths to greater than 62 mm) with a brown to black, triangular to subquadrate shell without external sculpturing, sometimes with greenish rays. Burlakova et al. (2011a, p. 158) considered the species rare throughout its range. For a detailed description see Howells et al. 1996 (pp. 91-92) and Howells 2014 (p. 65). Other native mussel species (e.g. Pimpleback, *Cyclonaias pustulosa*; Texas Pigtoe, *Fusconaia askewi*; Trinity Pigtoe, *F. chunii*; and Wabash Pigtoe, *F. flava*) can easily be mistaken for Louisiana Pigtoe when identified by shell morphology alone. A recent survey suggested experienced malacologists had a 76% success rate accurately identifying the species in the Little River, Oklahoma, when field identifications were compared with genetic analysis results (Inoue 2018, p. 1).

2.B.1.B. GENETIC DIVERSITY

Williams et al. (2017a, p. 51) recognized 23 species from the genus *Pleurobema*. Recent genetic work supports the monophyly of genus *Pleurobema* and subgenus *Pleurobema* (*Sintoxia*), with *P. cordatum*, *P. plenum*, *P. riddellii* (Louisiana Pigtoe), *P. rubrum*, and *P. sintoxia* forming a single clade, and all other *Pleurobema* species in a second clade (Inoue et al. 2018, pp. 694, 698; Williams 2017a, p. 51). Inoue et al. (2018, p. 669) also suggested divergence within the *P. riddellii* complex due to phylogenetic distinction between *Pleurobema cf. riddellii* from the Ouachita River drainage and *Pleurobema riddellii* from the Red River and west Gulf Coast drainages, although additional samples would be required to assess *P. cf. riddellii* as a possible new species. The type locality specimen (holotype, Smithsonian National Museum of Natural History, USNM 84635) was described by Lea in 1861 from the Trinity River near Dallas (Lea 1862, p. 392) and recently confirmed as Louisiana Pigtoe through genetic analysis (Randklev 2019b, p. 3).

2.B.1.C. REPRODUCTION AND FISH HOST INTERACTIONS

The reproductive cycle strategy of Louisana Pigtoe is currently unconfirmed. Marshall (2014, pp. 46-47) considered Louisiana Pigtoe to be bradytictic (i.e., longterm brooders; spawning occurs during the summer, glochidia are held by the female over winter and released the following spring); however, gravid females have been observed in July. A closely related congener, *Pleurobema plenum*, is known to utilize the tachytictic reproductive cycle (i.e., short term brooders; fertilization occurs in the spring and glochidia are expelled during the summer or early fall)(EPA 2007, p. 37). Freshwater mussel recruitment does not occur every year (Ford et al. 2016, p. 28).

The primary host fish for Louisiana Pigtoe has not been confirmed. Marshall (2014, pp. 59-60) suggested Bullhead Minnow (*Pimephales vigilax*), Red Shiner (*Cyprinella lutrensis*), and Blacktail Shiner (*Cypinella venusta*) as potential fish hosts based on a fish host distribution modeling effort. When modeled individually, Bullhead Minnow, Red Shiner, Dusky Darter (*Percina sciera*), and Blacktail Shiner accounted for 47%, 59%, 75%, and 77% of the gain of the full mussel model, respectively (Marshall 2014, pp. 57, 59-60). In this same study, and as part a model validation effort, encysted Louisiana Pigtoe glochidia were collected from wild Bullhead Minnow and Red Shiner from the Neches River; however, none were found encysted on Blacktail Shiner or Dusky Darter. Marshall (2014, p. 60) proposed that since Blacktail Shiner and Red Shiner are closely related and are known to hybridize, they likely serve as hosts to the same freshwater mussel species. Hinkle (2018) collected glochidia infected wild fish from the upper Neches River and kept them under laboratory conditions through glochidia metamorphosis. Results indicated six genetically confirmed Louisiana Pigtoe juveniles excysted from Blacktail Shiners (Hinkle 2018, p. 9, 11).

Hinkle (2018) reported male gametogenesis occurred from mid-July through mid-August with peak production occurring at 30°C (p. 19). Male gametes were flagellated and had an average length of 4.2 micrometers (µm), average width of 1.96 µm, and were found in concentrations ranging from 500,000 to approximately 20,000,000 gametes per milliliter. Female gametogenesis occurred from March through September with peak production at 25°C in early September through early October (Hinkle 2018, p. 19, 21). In females, concentrations of gametes ranged from 0 (but with clusters of oogonia and oocytes) up to 219,400 nonviable ova and 173,200 viable ova and averaged 12,500 nonviable and viable ova among sampled sexually mature females (Hinkle 2018, p. 19).

2.B.1.D. Age and Growth

A single Louisiana Pigtoe juvenile from the Neches River, Texas, was reported to grow 15 mm during its first year from an initial shell length of 2 mm (Ford et al. 2016, p. 30). Sexual maturity is achieved at shell lengths around 40 mm and mature adults grow approximately 2.5 mm in shell length per year (Ford et al. 2016, pp. 28, 30). At these growth rates, juvenile Louisiana Pigtoe could reach maturity in 3-4 years. Sexually mature males were estimated to be between 9 and 12 years old based on external valve annuli and were between 37-50 mm in shell length (Hinkle 2018, p. 19). Based on ova production, sexually mature females were estimated by external annuli to be between 4 and 12 years of age with shell lengths ranging from 29-59 mm (Hinkle 2018, p. 19).

2.B.1.E. Habitat

Louisiana Pigtoe occur in medium to large-sized streams and rivers in flowing waters (0.3-1.4 m/s) over substrates of cobble and rock or sand, gravel, cobble, and woody debris; they are often associated with riffle, run, and sometimes larger backwater tributary habitats (Ford et al. 2016, pp. 42, 52; Howells 2010a, p. 3-4; Williams et al. 2017b, p. 21). Specimens are typically found in shallower waters (0.1-1.2 m in depth; Howells 2010a, p. 3); however, recent surveys found Louisiana Pigtoe as deep as 3.33 m in the lower Neches River (downstream of B.A. Steinhagen Lake)(Corbett 2020, pp. 2, 4). Other specimens collected from the Neches River occupied substrates of gravel mixtures at depths between 0.57-1.12 m in run habitat with flow velocities of 0.44-0.66 m/s (Glen 2017, p. 17).

Table 2.2. Louisiana Pigtoe Life History Characteristics and Resource Needs

Life Stage	Resource Needs	Reference
Glochidia: through host fish attachment	**Potential Hosts:** Red Shiner (*Cyprinella (Notropis) lutrensis*), Blacktail Shiner (*Cyprinella venusta*), Bullhead Minnow (*Pimephales vigilax*)	Ford and Oliver 2015, p. 6; Bertram 2015, p. 32; Marshall 2014, p. 37 Hinkle 2018, p. 9, 11)
Juveniles: excystment through sexual maturity	Habitat requirements assumed to be similar to adults	
	Growth rate: One 2 mm individual grew 15 mm in the first year	Ford et al. 2016, p. 30
	Growth rate: May grow to 35 mm during first 3 years	Ford et al. 2016, p. 30
	Size at maturity: Approximately 40 mm	Ford et al. 2016, p. 28
Adults	**Stream flow:** Intermediate flow volume; 0.3-1.4 m/s in Neches River, TX; larger backwater tributaries of Neches River upper reaches	Ford et al. 2016, p. 42; Howells 2010a, pp. 3-4; Williams et al. 2017b, p. 21; Vaughan 2017, p. 9
	Depth: Typically 0.1 – 1.2 m	Howells 2010a, p. 3
	Substrate: Riffles of cobble and rock; sand, gravel, cobble, woody debris; runs with subdominant gravel mixtures	Ford et al. 2016, p. 52; Howells 2010a, p. 3; Burlakova et al. 2012, p. 5; Glen 2017, p. 17.
	Growth rate: Approximately 2.5 mm shell length per year	Ford et al. 2016, p. 30
	Abundance: Considered rare	Ford et al. 2016, p.4

Figure 2.4. Texas Heelsplitter observed from Neches River, Angelina/Trinity Counties, Texas (USFW photo).

2.B.2. TEXAS HEELSPLITTER, POTAMILUS AMPHICHAENUS (FRIERSON, 1898)

2.B.2.A. TAXONOMIC AND MORPHOLOGICAL DESCRIPTION

The Texas Heelsplitter (Figure 2.4) was first described as the species *Unio (Lampsilis) amphichaenus* by Frierson (1898, p. 109) from the Sabine River near Logansport, Louisiana. Vidrine (1993), Neck and Howells (1995, p. 4), and Howells (1996, p. 95) recognized the following synonyms (Howells 2010b, p. 4):

Unio (Lampsilis) amphichaenus of Frierson (1898);
Lampsilis (Proptera) amphichaenus (Frierson 1898) of Simpson (1900);
Lampsilis (Proptera) amphichaena (Frierson 1898) of Simpson (1914);
Proptera amphichaena (Frierson 1898) of Frierson (1927) and Haas (1969);
Leptodea amphichaena (Frierson 1898) of Burch (1975);
Lastena amphichaena (Frierson 1898) of Hoggarth (1988);
Potamilus amphichaenus (Frierson 1898) of Turgeon et al. (1988), Williams et al. (2017a), and others.

The recognized scientific name for Texas Heelsplitter is *Potamilus amphichaenus*, and this report refers to it as such. The following taxonomic treatment follows Williams et al. (2017a, pp. 35, 42).

CLASS Bivalvia Linnaeus, 1758

CLASS	**Bivalvia Linnaeus, 1758**
ORDER	**Unionida Gray, 1854**
FAMILY	**Unionidae Rafinesque, 1820**
SUBFAMILY	**Ambleminae Rafinesque, 1820**
TRIBE	**Lampsilini Ihering, 1901**
GENUS	***Potamilus* Rafinesque, 1818**
SPECIES	***Potamilus amphichaenus* (Frierson, 1898)**

The Texas Heelsplitter is a medium to large-sized freshwater mussel (up to 177 mm shell length) that has a tan to brown or black elliptical shell, with lighter coloration on the beaks. The hinge line is relatively straight. Texas Heelsplitter exhibit slight sexual dimorphism; females have a broadly rounded posterior margin and males are more pointed (Howells 2010b, p. 2). The base of the anterior margin exhibits a long, narrow gape, while a shorter, much wider gape is located along the posterior margin, presumably to accommodate the incurrent and excurrent apertures (Neck and Howells 1995, p. 4). Burlakova et al. (2011, p. 158) considered the species rare throughout its range. For a detailed morphological description see Neck and Howells (1995, p. 5-6), Howells et al. (1996, p. 95) and Howells (2014, p. 69).

2.B.2.B. GENETIC DIVERSITY

N. Ford et al. (2016, p. 48) sequenced the mitochondrial gene known as ND1 from six Texas Heelsplitter, six Pink Papershell (*Potamilus ohiensis*), and one suspected Texas Heelsplitter/Pink Papershell hybrid. Results showed that the suspected hybrid had a mix of both species genetic characteristics preventing positive species level identification. The hybrid morphology also exhibited a blending of the two species. Texas Heelsplitter and Pink Papershell are known to co-occur in the Trinity River drainage but the extent to which Texas Heelsplitter populations have been compromised by Pink Papershell genetics is currently unknown (Ford et al. 2016, p. 49).

2.B.2.C. REPRODUCTION AND FISH HOST INTERACTIONS

Although information specific to Texas heelsplitter reproduction is unavailable, other species from the tribe Lampsilini release glochidia in packets, called conglutinates, and are known to use mantle lures to attract sight feeding fishes that attack and rupture the marsupium, thereby becoming infested by glochidia (Barnhart et al. 2008, p. 377, 380). Most species of Lampsilini are long-term brooders (bradytictic) (p. 384). Howells (2010b) observed eggs and glochidia from two females during January from the Neches River; however, 13 others collected in January, July, and August were not gravid (p. 3). A single female, 90 mm in shell length, was estimated to have 6,665 eggs and 871,665 glochidia while another female with a 104 mm in shell length had 599,375 eggs and 646,250 glochidia (Howells 2010b, p. 3). Freshwater Drum (*Aplodinotus grunniens*) were confirmed as host fish for Texas Heelsplitter (Bosman et al. 2015, p. 15). Freshwater mussel recruitment does not occur every year (Ford et al. 2016, p. 28).

2.B.2.D. AGE AND GROWTH

A congener (*Potamilus purpuatus* (common name Bluefer)) from the southeast United States was reported by Haag and Rypel (2011) to reach a maximum age of 9–26 years (Table 1, p. 229) and members of tribe Lampsilini ranged from 4–50 years (p. 234) with a higher growth rate compared to other tribes (p. 239). Texas Heelsplitter has been reported mature at approximately 60 mm and juvenile presence has been confirmed in the Sabine River (Ford et al. 2016, p. 31).

2.B.2.E. HABITAT

Texas Heelsplitter occur in streams and rivers of the Trinity, Neches, and Sabine River drainages on substrates consisting of "firm mud, sand, or finer gravels bottoms, in still to moderate flows" and sometimes associated with fallen timber (Howells 2014, p. 69; Howells 2010b, p. 3, and Table 2.3). Vaughan (2017, p.15) collected specimens in substrates with high organic matter content. Dickson (2018, p. 23) reported Texas Heelsplitter were found in areas of large channel widths, with at least one low bank, in sandy substrates, at depths of 10 cm and deeper within the substrate, and in areas prone to bankfall. Texas Heelsplitter can tolerate man-made impoundments and have been found in several east Texas reservoirs (Howells 2010b, p. 3).

Table 2.3. Texas Heelsplitter Life History Characteristics and Resource Needs		
Life Stage	**Resource Needs**	**Reference**
Glochidia: through host fish attachment	**Host:** Freshwater Drum (*Aplodinotus grunniens*)	Bosman et al. 2015, p. 15
Juveniles: excystment through sexual maturity	Habitat requirements assumed to be similar to adults	
	Size at maturity: Approximately 60mm	Ford et al. 2016, p. 31
Adults	**Stream flow:** Slow to moderately flowing streams; tolerates impoundments	Howells 2010b, p. 3
	Depth: Deeper pools with sand	Ford et al. 2010, p. 13
	Substrate: Mud, sand, finer gravels, and mixtures of those with high organic matter content; sometimes associated with fallen timber	Howells 2010b, p. 3 Vaughan 2017, p. 15
	Brooding: Both eggs and glochidia found in two females in Neches River in January, glochidia found in one from Sabine River in July, others collected in January, July, and August not gravid	Howells 2010b, p. 3
	Fecundity: 90 mm sl (shell length) female (6,665 eggs, 871,665 glochidia), 104 mm sl female (599,375 eggs, 646,250 glochidia)	Howells 2010b, p. 3
	Habitat availability: Not declining in east Texas rivers	Williams et al. 2017b, p. 21
	Species abundance: Considered very rare	Howells 2010b, pg. 7; Williams et al. 2017b, p. 21
	Hybridization: May hybridize with Pink Papershell (*Potamilus ohiensis*) in Trinity basin; hybridized offspring morphology a mixture of both species characteristics	Ford et al. 2016, p.48

2.C. SUMMARY

This report considers two species of freshwater mussels, Louisiana Pigtoe and Texas Heelsplitter, both of which belong to the subfamily Ambleminae of the family Unionidae. The two species occur in three or more of the following seven basins in Arkansas, Louisiana, Oklahoma, and Texas: Little River, Calcasieu River, Big Cypress Bayou, Sabine River, Angelina River, Neches River, and Trinity River. The Louisiana Pigtoe and Texas Heelsplitter are among the 15 mussel species added to the list of Texas state threatened species by the Texas Parks and Wildlife Department in 2009 (TPWD 2009, pp. 1-2).

Species needs for Louisiana Pigtoe and Texas Heelsplitter generally include water environs with suitable substrate, adequate but not scouring flows, high-quality water (within optimal thermal and dissolved oxygen limits, and without harmful pollutants or contaminants), refuge from high and low flow events, stable substrates, access to appropriate host fishes, and appropriate nutrition (adequate but not excessive levels of CPOM and associated bacteria and fungi, or suspended phytoplankton).

CHAPTER 3 – SPECIES NEEDS AT THE INDIVIDUAL AND POPULATION LEVEL

This chapter considers the current and historical distribution of the Louisiana Pigtoe and Texas Heelsplitter, and evaluates factors important to assessing the viability of each species. Along with species distribution, we examine the needs of the species as they pertain to population resiliency, redundancy, and representation, which support species viability and reduce the likelihood of extinction.

For the purposes of this assessment, a mussel population is defined as a stream reach that is occupied by a collection of mussel beds through which host fish infested with glochidia may travel freely, allowing for dispersal of juveniles among and within mussel beds. Viability is defined as the ability of the species to sustain populations in the wild over time, in this case, 50 years. Fifty years represents at least five mussel generations and reflects the approximate forecasting horizon for climate projections and estimates of future development. This assessment considers the viability of each species following the SSA framework based on "the conservation biology principles of representation, resiliency, and redundancy (the 3Rs) to evaluate the current and future conditions of a species" as described by Smith et al. (2018, p. 7).

3.A. HISTORICAL RANGE AND CURRENT DISTRIBUTION

3.A.1. LOUISIANA PIGTOE

The range of the Louisiana Pigtoe is comprised of multiple river drainages throughout portions of east Texas, Louisiana, west Mississippi, southeast Oklahoma, and southwest Arkansas (Vidrine 1993, p.66; Howells et al. 1997, p.22; Randklev et al. 2013b, p. 269; Randklev 2018, entire). In Texas, the Louisiana Pigtoe has been recorded from several east Texas rivers, including the Big Cypress-Sulphur, Neches-Angelina, Sabine, San Jacinto, and Trinity River basins (Strecker 1931, p.29; Howells et al. 1996, p. 91; Howells 1997, p. 22; Howells 2006, p. 98; Burlakova et al. 2012, p. 12; D. Ford 2013, pp. 75 – 80; Ford et al. 2014, p. 10; Ford et al. 2016, p. 20; Randklev 2018, entire) (see Figure 3.1). In Louisiana, the species has been recorded within the Amite, Bayou Boeuf, Calcasieu, Red, Sabine, and Pearl River systems (Vidrine 1993, p.66; Randklev et al. 2013b, p. 269; LNHP 2018, entire; Randklev 2018, entire; Johnson et al. 2019, p. 11). In Mississippi, the species has been observed from the Pearl River (Johnson et al. 2019, p. 11). In Arkansas, the species has been recorded in the Cossatot, Saline, Rolling Fork, and Little Rivers (USFWS 2014, p. 29; USFWS 2015, p. 5; USFWS 2017, p. 8; Randklev 2018, entire). In Oklahoma, the species has been recorded in the mainstem of the Little River (Inoue 2018, p. 1). Reported populations from the Ouachita River system in Arkansas were determined to be phylogenetically distinct from Louisiana Pigtoe and are not considered in this report (Inoue et al. 2018, p. 699). We assume the historical distribution of the species would have included the entirety of the river basins described above where connectivity was not an issue and conditions were suitable (see stream segments highlighted black on Figure 3.1)(Note: our estimates of historical range include the mainstem and major tributaries within basins, but do not include an often vast network of minor tributaries even though these areas may have been occupied by mussels in the past).

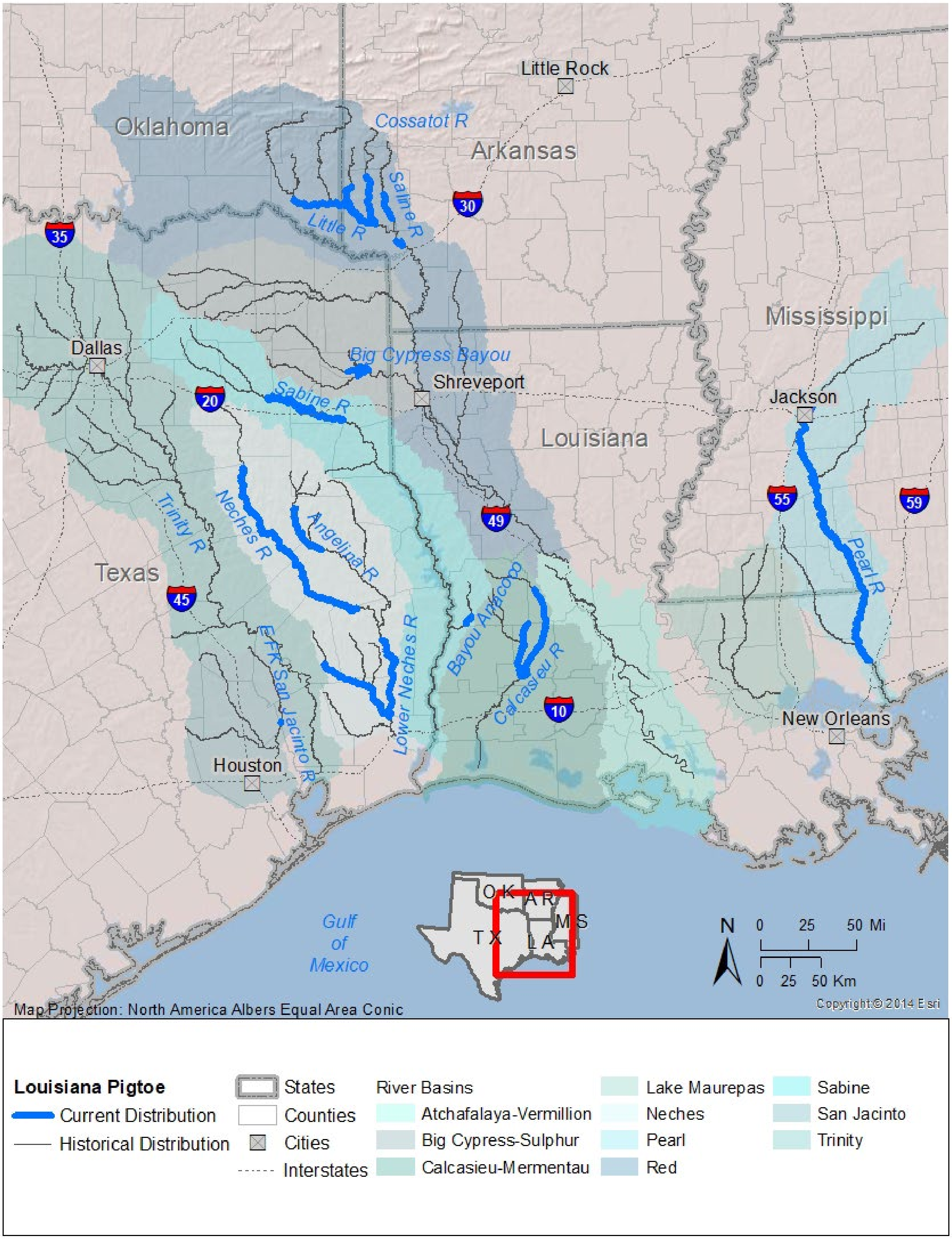


Figure 3.1. Estimated Louisiana Pigtoe current and historical distribution.

For this assessment a current Louisiana Pigtoe population, also referred to as a focal area and labeled "Current Distribution" on Figure 3.1 (see stream segments highlighted blue), is defined as a contiguous (hydrologically connected) reach of stream containing freshwater mussel beds with live or recent dead individuals (recent dead individuals likely indicate the presence of undetected live individuals; Randklev 2011, p. 17) observed in surveys performed from the year 2000 to present. Recent dead refers to dead individuals with valves still attached by the hinge, lustrous nacre, and intact periostracum; soft tissues may or may not be present (Howells 1996, pp. ii, 4). Since mussels are likely to occur beyond known sampled areas, estimates of the upper and lower extent of populations were determined by extending 0.5 miles beyond the most upstream or downstream location with live or recent dead observations since 2000. Populated tributaries (tributaries with live or recent dead observations since 2000) that were hydrologically connected (i.e., no impoundments or other barriers to host fish passage) to another population were considered a single population; if appropriate, the lower extent was then determined by extending the population line approximately 0.5 river miles downstream of the confluence of the populated streams. Specific survey location information was not available for the Pearl River population as of the writing of this report other than at the Hydrologic Unit Code 10 (HUC10) scale. This population was delineated from the upper boundary of the most upstream occupied HUC10 to the lower boundary of the most downstream occupied HUC10. Table 3.1 displays the estimated length of each population in river miles, extracted from the U.S. Geological Survey's National Hydrography Dataset (USGS 2014).

Table 3.1. Current known populations of Louisiana Pigtoe and estimated length of occupied reach.

Louisiana Pigtoe Focal Areas			
River Basin (Representaton Area)	**State**	**Population (Focal Area)**	**Length of Occupied Reach (miles)**
Red	AR/OK	**Little River /Rolling Fork**	103.6
	AR	**Cossatot River**	41.9
	AR	**Saline River**	27.9
	AR	**Lower Little River**	8.5
Big Cypress-Sulphur	TX	**Big Cypress Bayou**	32.3
Calcasieu-Mermentau	LA	**Upper Calcasieu River**	133.8
Pearl	LA/MS	**Pearl River**	280.8
Sabine	TX	**Sabine River**	86.8
	LA	**Bayou Anacoco**	9.1
Neches	TX	**Angelina River**	53.2
	TX	**Neches River**	203.0
	TX	**Lower Neches River**	160.4
San Jacinto	TX	**East Fork San Jacinto**	1.3

3.A.1.A. Little River/Rolling Fork Population (Red River Basin)

The Rolling Fork reach of this population extends from the confluence of the Rolling Fork with the Little River near Horatio, Arkansas, upstream to approximately 1.4 miles below DeQueen Lake. The Little River reach begins near Alleene, Arkansas and continues upstream to near Garvin, Oklahoma. Multiple survey efforts have observed a total of 280 Louisiana Pigtoe in the Little River/Rolling Fork population from 2013 to 2018 (Bouldin et al. 2013, entire; Davidson et al. 2014, entire; AGFC 2018, entire; Davidson 2017, entire; Inoue 2018, p. 1). The combined length of the Little River/Rolling Fork population is approximately 103.6 river miles within McCurtain County, Oklahoma, and Sevier and Little River counties, Arkansas.

3.A.1.B. Cossatot River Population (Red River Basin)

The Cossatot River population begins near its confluence with Little River at Millwood Lake and extends upstream to approximately five miles below Gillham Lake. In 2013, Louisiana Pigtoe were first recorded from the Cossatot River at 39 sites with 148 detections (AGFC 2018, entire). The length of the Cossatot River population is estimated at 41.9 river miles in Sevier County, Arkansas.

3.A.1.C. Saline River Population (Red River Basin)

The Saline River population extends approximately 28 miles upstream from its confluence with the Little River at Millwood Lake. In 2013, the Saline River was sampled for the first time at eight sites resulting in 18 Louisiana Pigtoe detections (Bouldin et al. 2013, entire; AGFC 2018, entire). The Saline River population occupies an estimated 27.9 river miles in Sevier and Howard counties, Arkansas.

3.A.1.D. Lower Little River Population (Red River Basin)

The Lower Little River population extends approximately 8.5 miles downstream of the Millwood Dam. The freshwater mussel community of the lower Little River was sampled only in 2012 resulting in two live and two recent dead Louisiana Pigtoe detections at three sites (AGFC 2018, entire). The Lower Little River population is approximately 8.5 river miles in length within Little River and Hempstead counties, Arkansas.

3.A.1.E. Big Cypress Bayou Population (Big Cypress-Sulphur Basin)

The Big Cypress Bayou portion of the Big Cypress Bayou population extends from approximately 0.5 miles downstream of the confluence with Little Cypress Bayou to approximately 4.5 miles downstream of Ferrell's Bridge Dam on Lake O' the Pines. The Little Cypress Bayou reach of the population extends approximately 10.6 miles upstream of its confluence with Big Cypress Bayou. From 2011 to 2016, 27 Louisiana Pigtoe were observed at 12 sites from this population (Randklev 2018, entire). The length of the entire Big Cypress Bayou Population is estimated at 32.3 river miles in Marion and Harrison counties, Texas.

3.A.1.F. Pearl River Population (Pearl Basin)

The Pearl River population extends from a point approximately 2.5 miles northwest of Nicholson, Mississippi, to the Ross Barnett Reservoir dam located approximately 8 mile northeast of Jackson, Mississippi. From 2005 to 2018, seven Louisiana Pigtoe were observed from three sites on the Pearl River (Johnson et al. 2019, p. 11). Additional surveys are needed, but based on the limited information available we estimate the length of the Pearl River population at 280.8 river miles encompassing portions of St. Tammany and Washington parishes, Louisiana; and Copiah, Hinds, Lawrence, Marion, Pearl River, Rankin, and Simpson counties, Mississippi.

3.A.1.G. Calcasieu River Population (Calcasieu-Mermentau Basin)

The Calcasieu River population extends upstream from approximately 0.5 mile downstream of the Whiskey Chitto Creek confluence to 0.5 miles upstream of the State Highway 121 bridge south of Hineston, Louisiana, including Whiskey Chitto Creek from its confluence with Calcasieu River to approximately 0.5 mile upstream of the Tenmile Creek confluence, and an approximately 25.8 mile portion of Tenmile Creek. During a survey effort conducted in 2000, "several" Louisiana Pigtoe were reported from two sites (LNHP 2018, entire). In a 2019 survey effort, eight Louisiana Pigtoe were recorded at two sites, including two individuals reported from a site on Tenmile Creek, a secondary tributary of the Calcasieu River (Kinney 2019, p. 1, 2). The Calcasieu River population extends for an estimated 133.8 river miles in Allen, Rapides, and Vernon parishes, Louisiana.

3.A.1.H. Sabine River Population (Sabine River Basin)

The Sabine River population begins approximately 3 miles upstream of the State Highway 43 bridge in Harrison and Rusk counties and continues 86.8 river miles upstream through Gregg, Upshur, Smith, and Wood counties, Texas, to approximately one mile downstream of the Farm-to-Market Road 14 bridge south of Hawkins, Texas. From 2010 to 2018, 39 live and one recently dead Louisiana Pigtoe were reported from 12 sites within this population (Ford et al. 2016, p. 27; Randklev 2018, entire).

3.A.1.I. Bayou Anacoco Population (Sabine River Basin)

The Bayou Anacoco population, located in Vernon Parish, is comprised of 9.1 river miles located west of Rosepine, Louisiana. In 2010, 14 Louisiana Pigtoe were collected from two sites within this population (Randklev 2013b, p. 269; LNHP 2018, entire).

3.A.1.J. Angelina River Population (Neches River Basin)

The Angelina River population, located in Angelina, Cherokee, and Nacogdoches counties, Texas, begins approximately 1.3 miles downstream of the U.S. Highway 59 bridge and extends upstream 53.2 river miles to approximately 0.8 miles upstream of Farm-to-Market Road 343. From 2006 to 2019, 18 sites were surveyed with 45 live and one recently dead Louisiana Pigtoe observations (Randklev 2018, entire).

3.A.1.K. Neches River Population (Neches River Basin)

The Neches River population runs 203.0 river miles through portions of Anderson, Angelina, Cherokee, Houston, Jasper, Polk, Trinity, and Tyler counties, Texas. The upper extent of this population is immediately downstream of Lake Palestine's Blackburn Crossing Dam and continues downstream to approximately 0.7 miles below the U.S. Highway 69 bridge south of Nancy, Texas. From 2006 to 2019, 1,030 live and three recently dead Louisiana Pigtoe were recorded at 147 sites within the delineated Neches River population (Randklev 2018, entire; Ford et al. 2016, p. 27; Ford et al. 2018, p. 11-12; Bio-West 2019, unpublished data), making it the largest known population in terms of number of individuals detected.

3.A.1.L. Lower Neches River Population (Neches River Basin)

The Lower Neches River population is comprised of portions of the Neches River below B.A. Steinhagen Lake's Town Bluff Dam, Big Sandy Creek, and Village Creek within Hardin, Jasper, Polk, and Tyler counties, Texas. The Big Sandy segment begins at its confluence with Kimball Creek, which then becomes Village Creek and continues upstream to approximately 4 miles west of Dallardsville, Texas. The population includes Village Creek in its entirety. The Neches River segment of this population starts approximately 0.5 mile downstream of the confluence with Village Creek and continues upstream for approximately 53 miles. The combined length of the Lower Neches River Population is 160.4 river miles. From 2000 to 2019, 169 live and eight recently dead Louisiana Pigtoe were collected from 37 sites within this population (Randklev 2018, entire; Bio-West 2019, unpublished data).

In 2019, an additional population (107 individuals) was discovered within the Lower Neches Valley Authority canal system near Beaumont, Texas (Bio-West 2021, p. 1; Bio-West 2019, p. 6). As of 2021, a total of 2,102 individuals, including juveniles and gravid females, were found in the same system (Bio-West 2021, p. 3, 4). The canal system provides an artificial waterway that receives pump-lifted water from the lower Neches River and delivers the water to customers located down gradient. Freshwater mussels likely became established in the canal system when one or more infected host fish in the Neches River became entrained in the pumps. Portions of the canal system are dewatered for maintenance on a five year schedule and age estimates indicate that all individuals were no older than four years (Bio-West 2019, p. 6). The canal population occupies artificially maintained habitat and may be transient, therefore it was not included as part of the analysis in this SSA. However, this population is an area of ongoing interest and research, and the population may prove useful for future conservation of the Louisiana Pigtoe by providing insight into habitat conditions that facilitate mussel growth and survival, or as a source of broodstock.

3.A.1.M. East Fork San Jacinto River Population (San Jacinto River Basin)

The East Fork San Jacinto River population's lower extent is approximately 0.9 mile downstream of Farm-to-Market Road 2090 near Plum Grove in Liberty County, Texas, and continues up the East Fork San Jacinto River to its upper extent, located approximately 0.4 mile upstream of the same bridge crossing. The length of this population is 1.3 river miles. In 2019, three live Louisiana Pigtoe were recorded at one site within the East Fork San Jacinto River population segment (Randklev 2019c, p. 1).

3.A.2. TEXAS HEELSPLITTER

The Texas Heelsplitter is endemic to the Neches, Sabine, and Trinity River drainages of east Texas (Howells et al. 1997, pg. 22). The type locality specimen was described by Frierson in 1898 from the Sabine River on the Texas – Louisiana border near Logansport, Louisiana (Frierson 1898, pg. 109). Within the Neches River drainage, the Texas Heelsplitter has been recorded at multiple locations throughout the system below Lake Palestine, including areas downstream of B.A. Steinhagen Reservoir (Vidrine 1993, pg.159; Howells et al. 1996, pg. 96; Howells et al. 1997, pg. 8, 22; Howells 2006, pp. 25-33; Ford et al. 2014, pg. 10; Ford et al. 2016, p. 22; Randklev 2018, entire; Bio-West 2019, unpublished data) (see Figure 3.2). Within the Sabine River drainage, the species has been recorded at several locations throughout the system from Lake Tawakoni to below Toledo Bend Reservoir (Vidrine 1993, pg. 159; Howells et al. 1996, pg. 96; Howells 2006 pp. 17-21, 83; Ford et al.2010, pg. 6; Hollis 2013, pg. 68; Ford et al. 2016, pg. 22; Randklev 2018, entire). Within the Trinity River drainage, the species has been recorded at several locations throughout the system, including reservoirs, from Lake Lewisville and Lake Grapevine to Lake Livingston (Howells 2006, pg. 42, 48; Bosman et al. 2015, pg. 15; Randklev 2018, entire). We assume the historical distribution of the species would have included the entirety of the river basins described above where connectivity was not an issue and conditions were suitable (see stream segments highlighted black on Figure 3.2)(Note: our estimates of historical range include the mainstem and major tributaries within basins, but do not include an often vast network of minor tributaries even though these areas may have been occupied by mussels in the past).

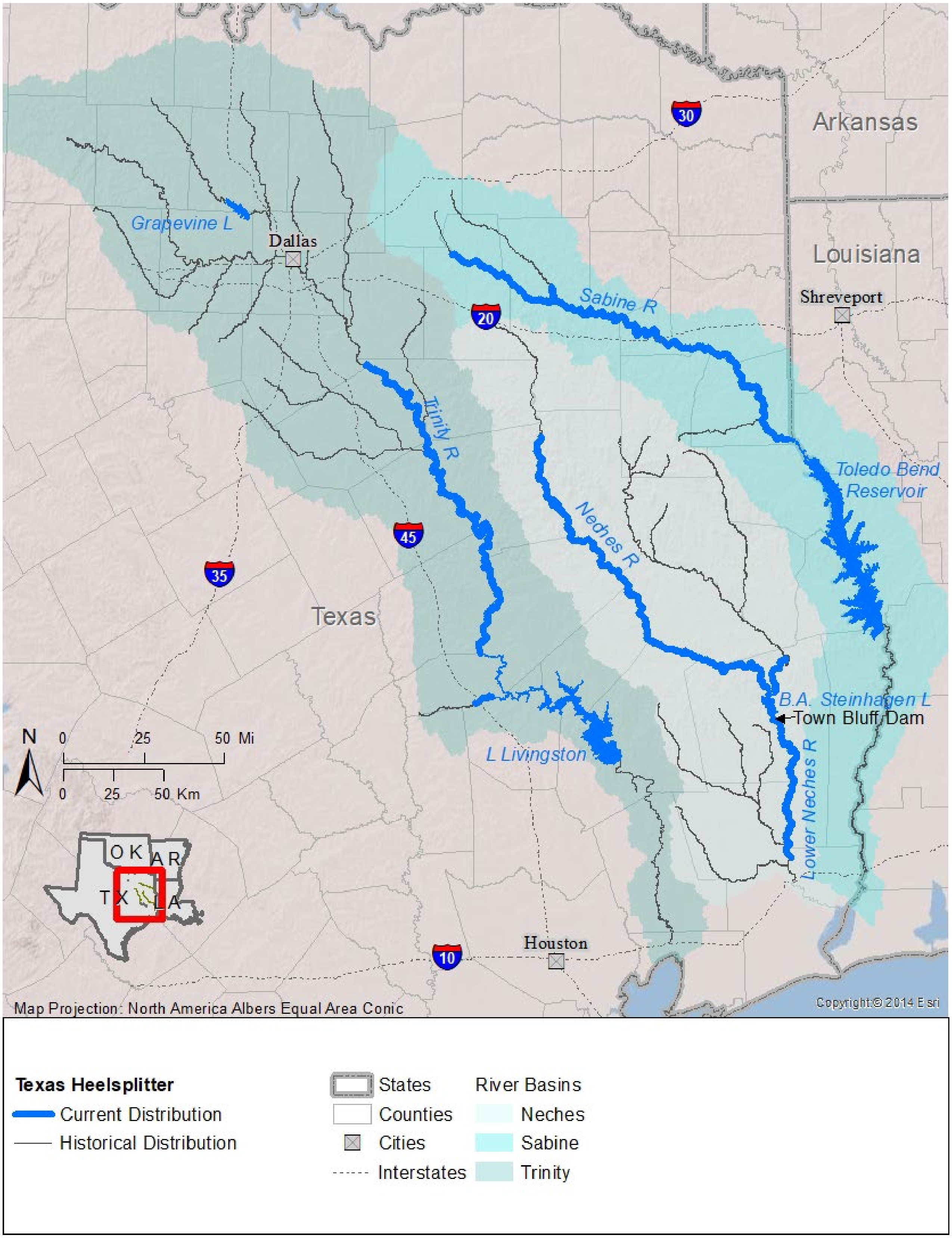


Figure 3.2. Estimated Texas Heelsplitter current and historical distribution.

Current Texas Heelsplitter populations (also referred to as focal areas and labeled "Current Distribution" on Figure 3.2 (see stream segments highlighted blue) were determined utilizing the same methodology described above for the Louisiana Pigtoe (i.e., by identifying stream reaches with live or recent dead observations since the year 2000) with the exception of the inclusion of impoundments. Impoundments with live or recent dead observations since 2000 were considered occupied in their entirety (due to a paucity of reservoir survey data). No attempt was made to quantify a surrogate parameter for occupied habitat reach length for impoundments.

Table 3.2. Current known populations of Texas Heelsplitter and estimated length of occupied reach.

Texas Heelsplitter Focal Areas			
River Basin (Representation Area)	**State**	**Population (Focal Area)**	**Length of Occupied Reach (miles)**
Sabine	TX/LA	**Sabine River/Toledo Bend**	245.8
Neches	TX	**Neches R/B.A. Steinhagen**	240.9
	TX	**Lower Neches River**	74.2
Trinity	TX	**Grapevine Lake**	n/a
	TX	**Trinity River/Lake Livingston**	203.4

3.A.2.A. Sabine River/Toledo Bend Population (Sabine River Basin)

The Sabine River/Toledo Bend population includes Toledo Bend Reservoir, Sabine River upstream to Lake Tawakoni's Iron Bridge Dam, and 7.9 river miles of Lake Fork Creek upstream from its confluence with the Sabine River. From 2005 to 2019, 82 live and 25 recently dead Texas Heelsplitters were collected at 88 sites from this population (Randklev 2018, entire; Ford et al. 2016, p. 27). The Sabine River/Toledo Bend population occupies an estimated 245.8 river miles of the Sabine River in De Soto and Sabine Parishes, Louisiana, and Gregg, Harrison, Newton, Panola, Rains, Rusk, Sabine, Shelby, Smith, Upshur, Van Zandt, and Wood counties, Texas.

3.A.2.B. Neches River/B.A. Steinhagen Lake Population (Neches River Basin)

The Neches River/B.A. Steinhagen Lake population includes B.A. Steinhagen Lake, the mainstem Angelina River from B.A. Steinhagen Lake to Sam Rayburn Reservoir, and the mainstem of the Neches River upstream 225.5 river miles to approximately 0.5 miles upstream of the Farm-to-Market 320 bridge southwest of Cuney, Texas. The population is located in Anderson, Angelina, Cherokee, Houston, Jasper, Polk, Trinity, and Tyler counties, Texas. Surveys of this population from 2005 through 2019 recorded 57 live and 97 recently dead Texas Heelsplitter at 41 sites (Randklev 2018, entire).

3.A.2.C. Lower Neches River Population (Neches River Basin)

The Lower Neches River population in Hardin, Jasper, and Tyler counties, Texas, extends 74.2 river miles downstream from Lake B.A. Steinhagen's Town Bluff Dam to approximately 4.5 miles downstream of the Village Creek confluence. Texas Heelsplitter observations from this population include 382 live and 12 recently dead individuals collected from 2004 to 2019 at 60 sites (Randklev 2018, entire; Bio-West 2019, unpublished data).

3.A.2.D. Grapevine Lake Population (Trinity River Basin)

The Grapevine Lake population is contained completely within Grapevine Lake (an impoundment on Denton Creek, a tributary of Elm Fork of the Trinity River) in Denton and Tarrant counties, Texas. A sampling effort in 2014 found at least two gravid female Texas Heelsplitter from this population (Randklev 2018, entire).

3.A.2.E. Trinity River/Lake Livingston Population (Trinity River Basin)

The Trinity River/Lake Livingston population occupies a total of 203.4 river miles in portions of Anderson, Ellis, Freestone, Henderson, Houston, Leon, Madison, Navarro, Polk, San Jacinto, Trinity, and Walker counties, Texas. This population includes Lake Livingston, the Trinity River 193.1 river miles upstream to Ennis, Texas, and 10.3 river miles of Bedias Creek (a tributary of the Trinity River). From 2005 to 2017, 55 live and six recently dead Texas Heelsplitter were recorded at 21 sites within the Trinity River/Lake Livingston population (Randklev 2018, entire).

3.B. NEEDS OF THE LOUISIANA PIGTOE AND TEXAS HEELSPLITTER

3.B.1. POPULATION RESILIENCY

For these species to maintain viability, their populations or some portion thereof must be resilient to disturbance from stochastic events that vary in duration and intensity. Stochastic events that have the potential to affect mussel populations include 1) high flow events that result in scouring, mobilization of substrates, and burial of mussel beds by large amounts of sediment (these events include flash floods following heavy rains, bank collapse events, etc.), 2) extended droughts and other dewatering events, 3) changes to water quality, including the ongoing or episodic discharge of environmental pollutants or hazardous materials (e.g., oil spill), 4) large-scale depredation events (e.g., collection, natural predation), 5) disease outbreaks, and 6) changes to basic water chemistry (e.g., high water temperature, episodes of low dissolved oxygen). A number of factors influence the resiliency of populations, including occupied stream length, abundance, and recruitment. Elements of occupied habitat such as water quality and hydrologic conditions also influence resiliency by controlling whether mussel populations can grow to maximize habitat occupancy, thereby increasing the resiliency of populations. These factors that affect population resiliency and habitat utilization are discussed in greater detail below in the context of how they meet the needs of mussels, how they were defined for the purposes of our analysis, and how they were used to evaluate population resiliency.

POPULATION FACTORS INFLUENCING RESILIENCY

Occupied Stream Length – Most freshwater mussels, including the Louisiana Pigtoe and Texas Heelsplitter, are found in aggregations called mussel beds that vary in size from about 50 to >5000 square meters (m^2) and are separated by stream reaches in which mussels are absent or rare (Vaughn 2012, p. 2). Strayer recognized that “unionid populations in streams are highly patchy, especially on scales of 1 – 100 m” (1999, p. 468). As discussed above, we define a mussel population at a larger scale than a single mussel bed; it is the collection or series of mussel beds within a stream reach between which infested host fish may travel, allowing for ebbs and flows in mussel bed density and abundance over time throughout the population’s occupied reach. Therefore, resilient mussel populations must occupy stream reaches long enough such that stochastic events that adversely affect individual mussel beds do not eliminate the entire population. In other words, repopulation by glochidia-infested fish from other mussel beds within the reach allow the population to recover from the temporary loss of individuals due to occasional disruptive events. For our analysis, we consider populations extending greater than 50 miles to have a high probability of persistence to stochastic events because a single event is unlikely to affect the entire population. Populations occupying reaches between 20 and 50 river miles have a moderate probability of persistence to stochastic events, while populations occupying reaches less than 20 miles have a low probability of persistence (Table 3.3). We consider probability of persistence to be a reflection of species’ resiliency. Note that we define populations occupying a stream length at or approaching zero miles as being functionally extirpated or extirpated.

Table 3.3. Occupied stream length and corresponding rankings for probabilities of persistence for Louisiana Pigtoe and Texas Heelsplitter populations.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/Extirpated**
Occupied Stream Length	> 50 river miles	20-50 river miles	< 20 river miles	none

Abundance – Populations require a minimum number of individuals to ensure stability and persistence. This threshold is often referred to as the minimum viable population and is generally calculated through a population viability analysis that estimates extinction risk given a number of input variables. There are no published minimum viable population estimates for the Louisiana Pigtoe or Texas Heelsplitter; therefore, it is unknown how many individuals are required to sustain populations of these mussels. However, population health is dependent on species abundance as well as water availability and the ability for mussels to meet life history needs within their habitats, which can be assessed and was evaluated as part of this report.

It is important to recognize that Louisiana Pigtoe observations used to determine abundance in this report may include misidentified individuals. Inoue (2018, p. 1) has suggested that without genetic confirmation, identification of Louisiana Pigtoe in the field based on shell morphology is questionable, with seasoned experts accurately identifying the species only 76% of the time. Unfortunately, genetic testing was not available for the majority of reported Louisiana Pigtoe historical observations, which relied solely on shell morphological characteristics for species identification (Randklev 2018, entire). Since there is no way to know the margin of error or to otherwise account for potential misidentifications, we determined abundance based on reported observations (as is). We do not consider misidentification to be an issue for Texas Heelsplitter observations, since they are recognizable based on morphological characteristics observed in the field and not easily confused with other species.

Mussel abundance in a given stream reach is a product of the number of mussel beds and the density of mussels within those beds. For populations of Louisiana Pigtoe and Texas Heelsplitter to be healthy (i.e., resilient), mussel beds of sufficient number and density must be present to allow recovery from natural and local stochastic events, allowing the mussel bed to persist and the overall local population to survive within a stream reach. Mussel abundance is indicated by the number of individuals found during a sample event. Mussel surveys are rarely a complete census of the population, but density can be estimated by the number of individuals found during a survey effort using various statistical techniques (i.e., estimate the total population from a subset of surveyed individuals). Population estimates are not available for all Louisiana Pigtoe and Texas Heelsplitter populations, and techniques for available surveys are not always directly comparable (i.e., same area size searched, similar search time, etc.). When available, we used the number of individuals captured relative to the amount of time surveys were conducted to estimate population abundance, hereafter referred to as overall catch per unit effort (CPUE). Although overall CPUE was the preferred metric to estimate population abundance, when overall CPUE was not available, the number of individuals detected during the most recent comprehensive survey effort was used as a surrogate metric. Abundance was calculated per the following guidelines, 1) Overall CPUE, for each population was calculated by adding the total number of live individuals detected during surveys since 2000 divided by total survey effort (time searched in person-hours), 2) Negative surveys (i.e., where no Louisiana Pigtoe or Texas Heelsplitter were found) were not available in our dataset or considered in the calculations, nor were surveys that did not report effort (e.g., person-hours searched), and 3) individuals detected per survey were used to calculate abundance for populations where CPUE data were not available. For sites with survey data spanning several years, abundance was based on the number of individuals detected during the most recent year's comprehensive survey effort.

For example, since the year 2000, mussel surveys with Texas Heelsplitter detections had been performed at 88 sites within the Sabine River/Todelo Bend population of Texas Heelsplitter (Randklev 2018, entire). Eighteen of those 88 survey sites reported effort in time spent searching (person-hours). The total number of live Texas Heelsplitter observed during those 18 surveys was 25 individuals, and a total of 170.51 person-hours were spent searching. Overall CPUE was then calculated by dividing the sum of live individuals (25) by the sum of time spent searching (170.51 person-hours) for a value of 0.15 live Texas Heelsplitter per person-hour searched. The 0.15 value, when compared to our definitions for abundance, falls within an overall CPUE of < 0.5, so the population is rated as functionally extirpated/extirpated with an estimated probability of persistence of less than 10% (Table 3.4). However, if no survey effort (i.e., time expended) had been reported from any of the surveys within the Sabine River/Toledo Bend population of Texas Heelsplitter since 2000, overall abundance would be based on the most recent year's comprehensive survey effort. In this case, 32 live Texas Heelsplitter were observed at 17 sites during the 2012 survey season (most recent comprehensive survey), which corresponds to our definition of moderate abundance (i.e., from ≤25 to <100 individuals) with an estimated probability of persistence of between 60% to 90%. Since information on survey effort was available and we consider CPUE to be a more accurate assessment of abundance, we used CPUE to rate abundance for this population.

Calculation of abundance in this manner is intended to be an estimate and is considered the best available information when population trend data do not exist and precise population abundance cannot be determined. Using these methods, we are able to estimate if the species is currently (since year 2000) common or rare within populations. Table 3.4 displays how estimates of relative abundance for each species were defined and used to rank the probability of persistence for populations from high to functionally extirpated/extirpated.

Table 3.4. Population abundance and corresponding rankings for probability of persistence for Louisiana Pigtoe and Texas Heelsplitter populations. Catch per unit effort (CPUE) refers to the overall number of mussels observed per person-hour searched during surveys performed within a population since the year 2000.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/Extirpated**
Population Abundance	Overall CPUE ≥ 4.0 (or ≥ 100 individuals found per population survey)	Overall CPUE ≥ 2.0 and < 4.0 (or ≥ 25 and <100 individuals found per population survey)	Overall CPUE ≥ 0.5 and < 2.0 (or ≥ 3 and < 25 individuals found per population survey)	Overall CPUE < 0.5 (or < 3 individuals found per population survey)

Reproduction/Recruitment – Resilient Louisiana Pigtoe and Texas Heelsplitter populations must also be reproducing and recruiting young individuals into the population to replace individuals lost to old age, disease, or predation. Population size and abundance are a reflection of habitat conditions, environmental stressors, and other past influences on the population. The ability of populations to successfully reproduce and recruit will determine if a population may be stable, increasing, or decreasing over time. For example, a large, dense mussel population that contains mostly old individuals is not likely to remain large and dense into the future if there are few young individuals to sustain the population over time (i.e., death rates exceed birth rates resulting in negative population growth). Conversely, a population that is less dense but has many young and/or gravid individuals is likely to grow, becoming more densely populated in the future (i.e., birth rates, and subsequent recruitment of reproductive adults, exceed death rates resulting in positive population growth). Detection rates of very young juvenile mussels during routine abundance and distribution surveys are extremely low due to sampling bias because sampling involves tactile searches and mussels < 35 mm can be difficult to detect (Strayer and Smith 2003, pp. 47-48). For this evaluation, we concluded there was evidence of reproduction/recruitment for a population when surveys detected small-sized individuals (near the low end of the detectable range or approximately 35 mm in size) since the year 2000 or gravid females (eggs and/or glochidia visible) were observed during the reproductively active time of year (Table 3.5). Sites lacking survey information specific to the presence of gravid females or juveniles due to inadequate effort default to a ranking of low in Table 3.5.

Table 3.5. Reproduction/recruitment and corresponding rankings for probability of persistence for populations of Louisiana Pigtoe and Texas Heelsplitter.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/ Extirpated**
Reproduction/ Recruitment	50% or more sites with juveniles (< 35mm) or gravid females present during breeding season. Fish host(s) present.	25-50% of sites inhabited by juveniles (< 35mm) and or gravid females present during breeding season. Fish host(s) present in moderate abundance.	< 25% of sites inhabited by juveniles (< 35mm) or gravid females present during breeding season. Fish host(s) present in low numbers and/or ability to disperse is reduced.	No gravid or juvenile individuals present

HABITAT FACTORS INFLUENCING RESILIENCY

Habitat Structure/Substrate – Suitable habitat structure and substrates vary among species of freshwater mussels, including between the Louisiana Pigtoe and Texas Heelsplitter. Most mussel species need stable substrate in which to anchor. The Louisiana Pigtoe occurs primarily in stream segments composed of riffle and run habitats where suitable substrates are present. Typical substrates utilized by the Louisiana Pigtoe include gravel, sand, and cobble, but the species has also been observed in fine substrates including silt. Sedimentation can negatively impact Louisiana Pigtoe populations by burying individuals and degrading anchoring habitat. The Texas Heelsplitter occurs in river systems and lentic waters (lakes or other non-flowing systems) primarily in pools and backwater habitats. Substrates providing adequate anchoring habitat for the Texas Heelsplitter include mud, sand, and silt. Sedimentation can also negatively impact Texas Heelsplitter populations by burying individuals. The habitat structure and substrate needs of both species are displayed in Table 3.6.

Table 3.6. Habitat structure and substrate conditions and corresponding rankings for probability of persistence for populations of Louisiana Pigtoe and Texas Heelsplitter.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/ Extirpated**
Habitat Structure/Substrate for Louisiana Pigtoe	Riffle and run habitat common. Substrates are stable and sufficient to provide anchoring habitat. Low levels of sedimentation on substrate.	Riffle and run habitat uncommon. Substrates are mostly stable and sufficient to provide anchoring habitat with some mobilization of particles and light sedimentation on substrate.	Riffle and run habitat rare or absent; substrates are mostly unstable; habitat eroded, or being buried by mobilized sediments from upstream.	No suitable habitat present
Habitat Structure/Substrate for Texas Heelsplitter	Pool and backwater habitats common. Stable mud, sand, and silt substrates sufficient to provide anchoring habitat. Low levels of sedimentation on substrate.	Pool and backwater habitats uncommon. Mud, sand, and silt substrates mostly stable and sufficient to provide anchoring habitat with some mobilization of particles and light sedimentation on substrate.	Pool and backwater habitat absent; substrates mostly unstable, habitat eroded, or being buried by mobilized sediments from upstream.	No suitable habitat present

Hydrological Regime – Freshwater mussels need water for survival. Some species are more resilient to low-velocity water than others and inhabit lentic waters (lakes or other non-flowing systems) including the Texas Heelsplitter. Neither Louisiana Pigtoe nor Texas Heelsplitter are able to persist in or tolerate areas that are regularly dewatered. High stream flows can degrade mussel habitat by producing shear stress capable of dislodging mussels and scouring stream bed substrates. Low stream flows can reduce the amount of anchoring habitat and negatively influence water quality parameters necessary for freshwater mussel persistence. Both high and low flows can also influence the presence or absence of host fish. The hydrological needs of both mussel species are displayed in Table 3.7.

Table 3.7. Hydrological regimes and corresponding rankings for probability of persistence for populations of Louisiana Pigtoe and Texas Heelsplitter.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/ Extirpated**
Hydrological Needs of Louisiana Pigtoe	Flowing water present year-round. No recorded periods of zero flow days, even during droughts. High flows and shear stress capable of causing bed movement or dislocation of mussels minimally impacts population (or habitat).	Flowing water present year-round (no zero flow days). High flows and shear stress capable of causing bed movement or dislocation of mussels moderately impacts population (or habitat).	Flowing water is not present year-round. River may become isolated pools or dry river bed seasonally. Zero flow days occur and riffles become dry. High flows and shear stress capable of causing bed movement or dislocation of mussels significantly impacts population (or habitat).	Dry stream bed or zero flow days occur often enough to preclude survival. Substrates are mostly unstable; high flows and shear stress are routinely capable of causing bed movement or dislocation of mussels (i.e., occurs frequently), resulting in unsuitable habitat for mussels.
Hydrological Regime of Texas Heelsplitter	Slow to moderate flowing water present year-round. No recorded periods of zero flow days, even during droughts. Extremely high, low, and/or erratic (e.g. significant fluctuations in flow over a short time) flows are rare. Little fluctuation of water levels in occupied reservoirs.	Slow to moderate flowing water present year-round (no zero flow days), however, extremely high, low, and/or erratic flows occur infrequently. Moderate fluctuation of water levels in occupied reservoirs.	Slow to moderate flowing water is not present year-round. River may become isolated pools or dry river bed seasonally. Zero flow days occur and riffles become dry. Extremely high, low, and/or erratic flows are routine. High fluctuation of water levels in occupied reservoirs.	Dry stream bed or zero flow days occur often enough to preclude survival. Extremely high, low, and/or erratic flows are frequent, resulting in unsuitable habitat for mussels. Large magnitude reservoir drawdowns occur frequently.

Water Quality – Freshwater mussels, as a group, are very sensitive to changes in water quality, including parameters such as temperature, dissolved oxygen, salinity, ammonia, pH and a variety of environmental pollutants. Habitats with natural, high quality water that is free of pollutants and contains appropriate levels of necessary parameters are considered suitable, while habitats with levels outside of the appropriate range for mussels are considered unsuitable or degraded habitat. Basic water quality conditions for the Louisiana Pigtoe and Texas Heelsplitter as they relate to our estimates of probability of persistence are displayed in Table 3.8.

Table 3.8. Water quality conditions and corresponding rankings for probability of persistence for populations of Louisiana Pigtoe and Texas Heelsplitter.

	Probability of Persistence			
	High	**Moderate**	**Low**	**Functionally Extirpated/Extirpated**
Water Quality	Overall WQ is good or excellent. No known contaminants, dissolved oxygen sufficient, and no thermal extremes documented. Total dissolved solids (TDS) stable or decreasing.	Overall WQ is good to fair. Contaminants known, moderate to low dissolved oxygen, and occasional temperature extremes documented. Not believed to be at levels that threaten mussel survival. TDS stable or slightly increasing.	Overall WQ is fair to poor. Contaminants known, low dissolved oxygen, and temperature extremes documented. TDS increasing. Levels sufficient to threaten mussel survival.	Overall WQ is limiting for aquatic life. Water quality degraded enough to preclude mussel habitation.

3.B.2. SPECIES REPRESENTATION

Maintaining species representation in the form of genetic and ecological diversity is important in safeguarding the ability of Louisiana Pigtoe and Texas Heelsplitter populations to adapt to future environmental changes. Mussel species like the Louisiana Pigtoe and Texas Heelsplitter need to retain populations throughout their range to maintain their overall potential, both genetically and ecologically (i.e., across habitats with varying capacity to meet life history attributes), to appropriately buffer the species against stochastic events and maintain their ability to respond to environmental changes over time (Jones et al. 2006, p. 531). The genetic diversity of populations of Louisiana Pigtoe and Texas Heelsplitter is unknown, although both species may have lost genetic diversity as populations have contracted over time or been reduced or extirpated by human activities. As such, maintaining the remaining representation in the form of genetic and ecological diversity will be important to preserving the capacity of these populations to adapt to future environmental change.

The major river basins within the historical distribution of the Louisiana Pigtoe described in section 3.A.1. span across multiple states and ecoregions, including Blackland Prairie, East Central Plains, and South Central Plains in Texas, the Ouachita Mountains of Oklahoma and Arkansas, and the Rolling and Coastal Plains of Mississippi. The major river basins within the historical distribution of the Texas Heelsplitter described in section 3.A.2. span multiple ecoregions in Texas, including Cross Timbers, Blackland Prairie, East Central Plains, and South Central Plains. Maintaining this ecological and spatial diversity in the future will be important to preserve representation for both species. For our analysis, we considered each river basin to be a separate representation area (see Tables 3.1 and 3.2).

3.B.3. SPECIES REDUNDANCY

Both the Louisiana Pigtoe and Texas Heelsplitter need multiple resilient populations distributed throughout their range to provide adequate redundancy. The more populations that exist, particularly densely populated populations, and the wider the distribution of those populations, the more redundancy the species will exhibit. Redundancy reduces the risk that a large portion of the species' range will be negatively affected by a single catastrophic natural or anthropogenic-induced event at any given point in time. Species that are well-distributed across their historical range are considered less susceptible to extinction and more likely to remain viable compared to species that are confined to a small portion of their historical range (Carroll et al. 2010, entire; Redford et al. 2011, entire). Historically, populations of both mussel species were hydrologically connected by fish migration within each river basin including their tributaries. Impoundments and other barriers to fish movement, such as river reaches with unsuitable water quality (e.g., high salinity or temperature), effectively isolate populations from one another, making repopulation of extirpated locations from nearby populations unlikely without human intervention (i.e., active restocking).

Chapter 4 - Current Condition of Louisiana Pigtoe and Texas Heelsplitter

This assessment defines a mussel population as a stream reach that is occupied by a collection of mussel beds through which host fish infested with glochidia may travel freely, allowing for dispersal of juveniles among and within mussel beds. This chapter discusses the current condition of Louisiana Pigtoe and Texas Heelsplitter populations for each species and evaluates the resiliency of those populations.

4.A. Methodology for Population Resiliency Assessment

To evaluate the current condition for each species and each population, we developed and assigned condition categories for three population factors (Occupied Stream Length, Abundance, Reproduction/Recruitment) and three habitat factors (Habitat Structure/Substrate, Hydrological Regime, and Water Quality); see "Chapter 3.B. Needs of the Louisiana Pigtoe and Texas Heelsplitter." Occupied stream length was calculated for populations of both species using ArcGIS by summing the contiguous stream miles between locations known to be occupied since 2000 (based on available survey data). Scoring of the six factors was based solely on conditions within the populations as delineated by the contiguous occupied reaches shown in Figures 3.1 and 3.2. All six factors were scored by U.S. Fish and Wildlife Service and state wildlife agency biologists based on consensus using a combination of available empirical data and expert opinion. Empirical data (e.g. survey results) included information that was available from our files, directly provided to our office by other agencies and academia, and obtained from agency websites or online reports.

For each population, categories were assigned a numerical value: "3" for healthy (high condition), "2" for moderately healthy (moderate condition), "1" for unhealthy (low condition), and "0" for extirpated or functionally extirpated. Six categories were scored for stream (lotic) populations, while five categories were scored for the Grapevine Lake (lentic) population of Texas Heelsplitter (for which Occupied Stream Length did not apply). Values for population and habitat factors were averaged (i.e., scores for all six categories were summed and divided by six for lotic populations, and scores for five categories were summed and divided by five for the lentic population of Texas Heelsplitter), resulting in an overall condition value that was then compared back to the individual category value for population abundance. This comparison was necessary to ensure that the overall condition value did not exceed the population abundance value (i.e., overall population condition was capped at the population abundance condition value) because we consider abundance to be the most direct measure of the health and status of the species and therefore the best available information. For example, measures like water quality and hydrology might rank high, indicating high quality habitat, but that does not necessarily indicate the presence of the species, only the presence of suitable habitat. Capping by abundance ensures overall scoring is based on species-specific information. Of 18 focal areas we evaluated for both species, capping overall condition by abundance resulted in a lower condition for 11 populations (indicated by ** footnote on Tables 4.3 and 4.5). The resulting overall current condition value and category for each population is both a qualitative and quantitative estimate based on the analysis of the three population factors and three habitat factors. Table 4.1 displays our presumed probability of persistence and probability of extirpation over 20 years (approximate time needed for at least three to five generations of each species) for populations that fall into one of four current condition categories. For example, for our analysis we assumed that a mussel population rated as having a high overall current condition would have less than a 10% probability of becoming extirpated or functionally extirpated over 20 years into the future.

Table 4.1. Presumed probabilities of persistence and extirpation for each overall current condition category over 20 years.

Likelihood of Persistence:	High	Moderate	Low	Extirpated/Functionally Extirpated
Range of Presumed Probability of Persistence over ~20 years	**>90%**	**60 – 90%**	**10 – 60%**	**< 10%**
Range of Presumed Probability of Extirpation over ~20 years	**<10%**	**10 – 40%**	**40 – 90%**	**> 90%**

4.B. LOUISIANA PIGTOE

4.B.1. CURRENT CONDITIONS

Based on our analysis, the total combined stream length currently occupied by the 13 remaining Louisiana Pigtoe populations described in Chapter 3 is 1,142 river miles, which is approximately 17% of the presumed historical range for the species (6,775 river miles). The presumed historical range was calculated by adding the combined stream length of the mainstem and major tributaries for all river basins with records for the species, but did not include minor tributaries with no records even though these areas may have been occupied at one time. Although a precise historical range for the species is unknown, and occupied areas would likely fluctuate naturally over time due to a variety of environmental conditions, assuming only the mainstem and major tributaries were occupied, this would represent an 83% reduction to the range of the species.

To summarize the overall current conditions of Louisiana Pigtoe populations, we assigned each population to one of four condition categories (high, moderate, low, or extirpated/functionally extirpated) based on an evaluation of the six population and habitat factors discussed in Chapter 3. Table 4.2 provides the definitions we used to assign conditions for the six factors. It is important to note that although the definitions were developed by our team based on the best available science and were vetted by experts in the field of malacology, they are intended for the sole purpose of meeting the objectives of this SSA and should not be viewed as standards that are applicable for other purposes.

Table 4.3 presents the condition we assigned for all six factors as well as the overall condition for each of the 13 remaining Louisiana Pigtoe populations. The overall condition of each population is also displayed graphically within a map of the historical range of the species in Figure 4.1. To evaluate the overall condition for each population, Appendix B, Table B.1 was developed. Within Table B.1, the cause and effects of stressors for each factor were considered through a combination of literature pertinent to specific factors and the elicitation of subject matter experts within the SSA working group.

Table 4.2. Definitions for population and habitat characteristics used to assign the current condition of Louisiana Pigtoe populations (see Table 4.3).

	Population Factors			Habitat Factors		
Condition	**Occupied Habitat** (stream length)	**Abundance**	**Reproduction/ Recruitment**	**Habitat Structure/ Substrate**	**Hydrology**	**Water Quality**
High	> 50 river miles	Catch per unit effort (CPUE) ≥ 4.0 *(or ≥ 100 individuals found per population survey)	50% or more sites with juveniles (< 35mm) and gravid females present during breeding season. Fish hosts present (i.e., not limiting).	Riffle and run habitat common. Substrates are stable and sufficient to provide lodging. Low levels of sedimentation on substrate.	Flowing water present year-round. No recorded periods of zero flow days, even during droughts. High flows and shear stress capable of causing bed movement or dislocation of mussels minimally impacts population (or habitat).	Overall WQ is good or excellent. No known contaminants, dissolved oxygen sufficient, and no thermal extremes documented. Pollutants indicative of anthropogenic degradation, such as total dissolved solids (TDS), are stable or decreasing.
Moderate	20–50 river miles	4.0 > CPUE ≥ 2.0 *(or ≥ 25 and <100 individuals found per population survey)	25-50% of sites inhabited by juveniles (< 35mm) and gravid females present during breeding season. Fish hosts present in moderate abundance.	Riffle and run habitat uncommon. Substrates are mostly stable and sufficient to provide lodging with some mobilization of particles and light sedimentation on substrate.	Flowing water present year-round (no zero flow days). High flows and shear stress capable of causing bed movement or dislocation of mussels moderately impacts population (or habitat).	Overall WQ is fair. Contaminants known; low dissolved oxygen, and temperature extremes documented. Not believed to be at levels that threaten mussel survival. TDS stable or slightly increasing.
Low	< 20 river miles	2.0 > CPUE ≥ 0.5 *(or ≥ 3 and < 25 individuals found per population survey)	< 25% of sites inhabited by juveniles (< 35mm) and gravid females present during breeding season. Fish host present in low numbers and/or ability to disperse is reduced.	Riffle and run habitat rare or absent; Substrates are mostly unstable; habitat eroded or being buried by mobilized sediments from upstream.	Flowing water is not present year-round. River may become isolated pools or dry river bed seasonally. Zero flow days occur and riffles become dry. High flows and shear stress capable of causing bed movement or dislocation of mussels significantly impacts population (or habitat).	Overall WQ is poor. Contaminants known; low dissolved oxygen, and temperature extremes documented. TDS increasing. Pollutant levels sufficient to threaten mussel survival.
Extirpated/ Functionally Extirpated	none	CPUE < 0.5 *(or < 3 individuals found per population survey)	No gravid or juvenile individuals present	No suitable habitat present.	Dry stream bed or zero flow days occur often enough to preclude survival. Substrates are mostly unstable; high flows and shear stress are routinely capable of causing bed movement or dislocation of mussels (i.e., occurs frequently), resulting in unsuitable habitat for mussels.	Overall WQ is limiting for aquatic life. Water quality degraded enough to preclude mussel habitation.

*the number of individuals found per most recent comprehensive population survey were used to rank Abundance when CPUE information was not available.

Table 4.3. The estimated **c**urrent condition of known Louisiana Pigtoe populations*; where high condition = 3 (green box), moderate condition = 2 (yellow box), low condition = 1 (red box), and extirpated/functionally extirpated = 0 (grey box).

		Population Factors			Habitat Factors			
River Basin	**Population**	**Occupied Habitat**	**Abundance**	**Reproduction/ Recruitment**	**Habitat Structure/ Substrate**	**Hydrology**	**Water Quality**	**Overall Condition** (Viability)
Red	Little River/Rolling Fork	3	2	3	2	2	2	2
	Cossatot River	2	3	3	3	2	2	3
	Saline River	2	2	1	1	2	2	2
	Lower Little River	1	0	1	1	1	1	0**
Big Cypress	Big Cypress Bayou	2	2	1	2	2	2	2
Calcasieu-Mermentau	Calcasieu River	3	1	1	3	2	2	1**
Pearl	Pearl River	3	1	1	2	2	2	1**
Sabine	Sabine River	3	0	1	3	2	2	0**
	Bayou Anacoco	1	2	1	3	2	2	2
Neches	Angelina River	3	1	1	3	3	2	1**
	Neches River	3	3	2	3	3	2	3
	Lower Neches River	3	1	2	3	2	2	1**
San Jacinto	East Fork San Jacinto River	1	1	1	1	2	2	1

*See Appendix B, Table B.1 for supporting information used to score population and habitat factors.

**Indicates focal areas where overall condition was capped by abundance (i.e., scored lower).

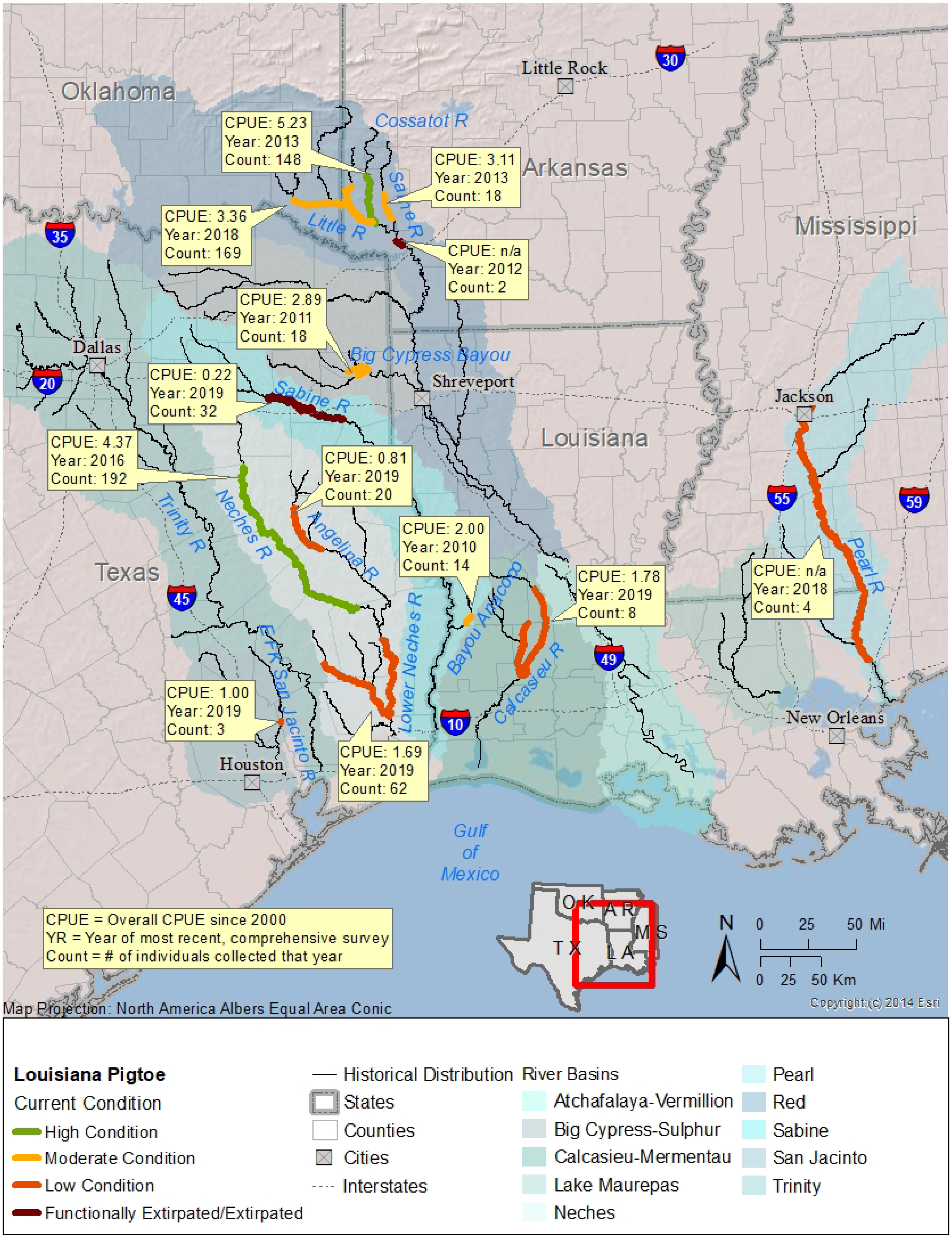


Figure 4.1. Location and estimated current condition of 13 remaining populations of louisiana pigtoe within the historical range of the species.

4.B.2. CURRENT POPULATION RESILIENCY

Resiliency describes the ability of a species to withstand stochastic disturbance. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Generally speaking, populations need abundant individuals within habitat patches of adequate area and quality to maintain survival and reproduction in spite of natural and anthropogenic disturbance. Resilient populations have the ability to rebound from events that cause mortality or otherwise temporarily reduce fecundity to restore the overall population back to pre-disturbance levels within a relatively short amount of time (e.g., 2-5 years, depending on the magnitude of the event). Based on our analysis, the Louisiana Pigtoe currently persists as 13 populations across five states and within portions of seven separate river basins (Big Cypress-Sulphur, Calcasieu-Mermentau, Neches, Pearl, Red, Sabine, and San Jacinto; Chapter 3). The predicted resiliency for populations is equal to the estimated current condition (e.g., high current condition = high resiliency).

Within the Big Cypress-Sulphur River basin in northeast Texas, Louisiana Pigtoe currently occupy portions of Big Cypress Bayou, a drainage that extends approximately 150 miles. The Big Cypress Bayou population occupies approximately 32 miles of river at the confluence of Big Cypress Bayou and Little Cypress Bayou located between Lake O' the Pines and Caddo Lake. Based on our analysis as defined by our estimates, the current condition evaluation for this population determined that occupied habitat reach length, abundance, habitat structure/substrate, hydrology, and water quality were in moderate condition (Tables 4.2 and 4.3). Reproduction/recruitment was determined to be in low condition due to a lack of reported juveniles or gravid females (Randklev 2018, entire). This single population is estimated to have a moderate overall current condition and, therefore, moderate resiliency (Table 4.3).

Louisiana's Calcasieu-Mermentau River basin has a single population on hydrologically connected portions of the mainstem Calcasieu River, Whiskey Chitto and Tenmile creeks. Louisiana Pigtoe are currently known to occur along an approximately 134 mile section of these streams in Allen, Rapides, and Vernon parishes. The current condition evaluation for this population determined that occupied habitat reach length and habitat structure/substrate were in high condition while hydrology and water quality were in moderate condition (Tables 4.2 and 4.3). Based on our analysis as defined by our estimates, abundance and reproduction/recruitment were found to be in low condition, primarily due to the low number and distribution of surveys performed within the Calcasieu River basin and resulting lack of data (LNHP 2018, entire; Kinney 2019, entire). Based on our analysis as defined by our estimates, this population has a low overall current condition, which corresponds to low resiliency.

The Neches River basin in Texas has three populations of Louisiana Pigtoe, one each in the Angelina, Neches (above B.A. Steinhagen reservoir), and Lower Neches rivers (below B.A. Steinhagen). These three populations combined encompass over 400 miles of river in a basin that many experts believe contains some of the best remaining habitat for freshwater mussels in Texas. The Neches River and Lower Neches River populations are hydrologically isolated from each other by an impoundment that forms B.A. Steinhagen Lake known as Town Bluff Dam, while the Angelina River population is isolated from the Neches River population by Sam Rayburn Dam and Reservoir. The Angelina River population current condition evaluation (Tables 4.2 and 4.3) found that occupied habitat reach length, habitat structure/substrate, and hydrology were high condition; water quality was in moderate condition; and abundance and reproduction/recruitment were in low condition, due to low CPUE and lack of juvenile or gravid female presence data, respectively (Randklev 2018, entire; Bio-West 2019, unpublished data). The Neches River population current condition evaluation determined that occupied reach habitat length, abundance, habitat structure/substrate, and hydrology were in high condition, while reproduction/recruitment and water quality were in moderate condition. No population or habitat current condition factors were determined to be low for the Neches River population. Based on our analysis as defined by our estimates, the Lower Neches River population current condition evaluation found occupied habitat reach length and habitat structure/substrate in high condition while reproduction/recruitment, hydrology, and water quality were moderate condition. The Lower Neches River population abundance was in low condition due to low CPUE (Randklev 2018, entire, Bio-West 2019, unpublished data). The Angelina River and Lower Neches

River populations have a low overall current condition, and the Neches River population has a high overall current condition; resiliency for these populations is low, low, and high, respectively.

The Pearl River basin in Louisiana and Mississippi has a single population of Louisiana Pigtoe within the mainstem Pearl River that extends approximately 150 miles below Ross Barnett Dam near Jackson MS to Picayune MS (upstream of Interstate 59). A new impoundment proposed by the Rankin-Hinds Pearl River Flood and Drainage Control District 9 miles downstream of Ross Barnett Reservoir intended for flood control is still under review. The current condition evaluation for the Pearl River population determined that occupied habitat reach length was in high condition; habitat structure/substrate, hydrology, and water quality were in moderate condition; and abundance and reproduction/recruitment were in low condition due to the few individuals observed and lack of juvenile or gravid female presence (Johnson et al. 2019, p.11). The Pearl River population has an estimated overall low current condition and low resiliency.

The Red River basin contains four distinct populations, all within the Little River drainage in Arkansas and Oklahoma, including populations in the Cossatot River, Little River/Rolling Fork, Lower Little River, and Saline River. Millwood Lake, located in southwest Arkansas, hydrologically separates the Cossatot River, Saline River, Little River/Rolling Fork, and Lower Little River populations from one another. The Cossatot River population current conditions evaluation found that abundance, reproduction/recruitment, and habitat structure/substrate were in high condition; occupied habitat reach length, hydrology, and water quality were in moderate condition; and no habitat or population factors were determined to be in low condition (Tables 4.2 and 4.3). The Little River/Rolling Fork population current condition evaluation determined occupied habitat reach length and reproduction/recruitment were high condition. All other population and habitat factors were in moderate condition. The Saline River population current condition evaluation found occupied habitat reach length, abundance, hydrology, and water quality in moderate condition while reproduction/recruitment and habitat structure/substrate were in low condition. The Lower Little River population current conditions evaluation determined that all population and habitat factors were in low condition except abundance, which was functionally extirpated/extirpated due low numbers of individuals observed in this focal area (AGFC 2018, entire). In summary, the Cossatot River population has a high overall current condition, the Little River/Rolling Fork and Saline River populations have a moderate overall current condition, and the Lower Little River population is considered functionally extirpated/extirpated.

There are two known Sabine River populations, one located along 85 miles of river between State Highway 14 near Hawkins, Texas downstream to above the State Highway 43 crossing near Tatum, Texas, and a second population within a 9 mile segment of Bayou Anacoco in Louisiana. These populations are hydrologically separated by Toledo Bend Dam and Reservoir. The Sabine River population current condition evaluation determined that occupied habitat reach length and habitat structure/substrate were high condition; hydrology and water quality were moderate condition; and reproduction/recruitment in low condition. However, abundance was functionally extirpated/extirpated due to low reported CPUE (Randklev 2018, entire). The Bayou Anacoco population current conditions evaluation found habitat structure/substrate was high condition; abundance, hydrology, and water quality were in moderate condition; and occupied habitat reach length and reproduction/recruitment were low condition due to the distribution of observed individuals and lack of reported juveniles or gravid females (Randklev 2018, entire). The Sabine River population is considered functionally extirpated/extirpated due to the very low number of individual mussels found during recent surveys, and therefore has little to no resiliency. The Bayou Anacoco population is in moderate current overall condition and has moderate resiliency.

The East Fork San Jacinto River population located, near Plum Grove, Texas, occupies a 1.3 mile segment of stream. The population current condition evaluation found hydrology and water quality were moderate condition while the other population and habitat factors were low condition (Tables 4.2 and 4.3). The East Fork San Jacinto River population was determined to be in overall low condition due to the limited number of individuals found. This population was estimated to have low resiliency.

4.B.3. CURRENT SPECIES REPRESENTATION

Representation describes the ability of a species to adapt to changing environmental conditions over time. It is characterized by the breadth of genetic and environmental diversity within and among populations. Our analysis explores the relationship between the species life history and the influence of genetic and ecological diversity and the species ability to adapt to changing environmental conditions over time.

We consider Louisiana Pigtoe to have representation in the form of genetic, ecological, and geographical diversity between each of seven river basins: Big Cypress-Sulphur, Calcasieu-Mermentau, Neches, Pearl, Red, Sabine, and San Jacinto. Because there are no un-impounded, freshwater connections that allow movement between the seven basins, for our analysis we treated each river basin as a separate area of representation.

4.B.4. CURRENT SPECIES REDUNDANCY

Redundancy describes the ability of a species to withstand and recover from catastrophic events. High redundancy is achieved through multiple populations that serve to spread risk, thereby reducing the impact that any one event might have in terms of overall loss to the species. Redundancy is characterized by having multiple healthy, resilient populations distributed across the range of the species. It can be measured by population number, resiliency, spatial extent, and degree of connectivity. Our analysis explored the influence of the number, distribution, and connectivity of populations on the species' ability to withstand catastrophic events.

Within identified representation areas, the Big Cypress-Sulphur, Calcasieu-Mermentau, Pearl, and San Jacinto River basins each have only one known current population and therefore lack redundancy. The Sabine River basin has two separate populations (Sabine River and Bayou Anacoco populations) but lacks redundancy due to the Sabine River population being functionally extirpated/extirpated. The Neches and Red River basin each currently have three viable populations (the Lower Little River population in the Red River basin is considered functionally extirpated/extirpated), however each population is hydrologically isolated within their respective river basins and are, therefore, considered to provide only limited redundancy.

4.C. TEXAS HEELSPLITTER

4.C.1. CURRENT CONDITIONS

Based on our analysis, the total combined stream length currently occupied by the five known Texas Heelsplitter populations described in Chapter 3 equals 764 river miles, including four reservoirs, which is approximately 24.3% of more than 3,146 river miles that the species may have occupied historically. This approximate range reduction assumes the species continuously occupied its entire historical range, which is unlikely given the species' specialized habitat preferences. However, our estimates of historical range are based solely on river miles within the mainstem and major tributaries, and therefore take a conservative approach since they do not include a significant number of minor tributaries for which we lack records but that may have been occupied at one time. Due to a lack of research into Texas Heelsplitter habitat needs in lacustrine environments and uncertainty whether those populations function as viable populations, no attempt was made to quantify occupied habitat in reservoirs.

To summarize the overall current conditions of Texas Heelsplitter populations, we assigned each population to one of four condition categories based on an assessment of six factors, as described in Section 4.B.1 above and as displayed in Table 4.4. Table 4.5 presents the estimated overall condition of Texas Heelsplitter populations, which is also displayed geographically across the range of the species in Figure 4.2. To evaluate the overall condition for each population, Appendix B, Table B.1 was developed. Within Table B.1, the cause and effects of stressors for each factor were considered through a combination of literature pertinent to specific factors and the elicitation of subject matter experts within the SSA working group.

Table 4.4. Definitions for population and habitat characteristics used to assign the current condition of Texas Heelsplitter populations (see Table 4.5)

	Population Factors			Habitat Factors		
Condition	**Occupied Habitat** (stream length)	**Abundance**	**Reproduction/ Recruitment**	**Habitat Structure/ Substrate**	**Hydrology**	**Water Quality**
High	> 50 river miles	Catch Per Unit Effort (CPUE) ≥ 4.0 *(or ≥ 100 individuals found per population survey)	50% or more sites with juveniles (< 35mm) and gravid females present during breeding season. Fish hosts present (i.e., not limiting).	Pool and backwater habitats common. Stable mud, sand, and silt substrates sufficient to provide lodging. Low levels of sedimentation on substrate.	Slow to moderate flowing water present year-round. No recorded periods of zero flow days, even during droughts. Extremely high, low, and/or erratic flows are rare. Little fluctuation of water levels in occupied reservoirs (i.e., little to no drying of occupied habitat).	Overall WQ is good or excellent. No known contaminants, dissolved oxygen sufficient, and no thermal extremes documented. Pollutants indicative of anthropogenic degradation, such as total dissolved solids (TDS) are stable or decreasing.
Moderate	20–50 river miles	4.0 > CPUE ≥ 2.0 *(or ≥ 25 and < 100 individuals found per population survey)	25-50% of sites inhabited by juveniles (< 35mm) and gravid females present during breeding season. Fish hosts present in moderate abundance.	Pool and backwater habitats uncommon. Mud, sand, and silt substrates mostly stable and sufficient to provide lodging with some mobilization of particles and light sedimentation on substrate.	Slow to moderate flowing water present year-round (no zero flow days), however, extremely high, low, and/or erratic flows occur infrequently. Moderate fluctuation of water levels in occupied reservoirs.	Overall WQ is fair. Contaminants known, low dissolved oxygen, and temperature extremes documented. Not believed to be at levels that threaten mussel survival. TDS stable or slightly increasing.
Low	< 20 river miles	2.0 > CPUE ≥ 0.5 *(or ≥ 3 and < 25 individuals found per population survey)	< 25% of sites inhabited by juveniles (< 35mm) and gravid females present during breeding season. Fish host present in low numbers and/or ability to disperse is reduced.	Pool and backwater habitat absent; substrates mostly unstable, habitat eroded, or being buried by mobilized sediments from upstream.	Slow to moderate flowing water is not present year-round. River may become isolated pools or dry river bed seasonally. Zero flow days occur and riffles become dry. Extremely high, low, and/or erratic flows are routine. High fluctuation of water levels in occupied reservoirs.	Overall WQ is poor. Contaminants known, low dissolved oxygen, and temperature extremes documented. TDS increasing. Pollution levels sufficient to threaten mussel survival.
Extirpated/ Functionally Extirpated	none	CPUE <0.5 *(or < 3 individuals found per population survey)	No gravid or juvenile individuals present	No suitable habitat present	Dry stream bed or zero flow days high enough to preclude survival. Extremely high, low, and/or erratic flows are frequent, resulting in unsuitable habitat for mussels. Large magnitude reservoir drawdowns occur frequently, resulting in drying of occupied habitat and morality.	Overall WQ is limiting for aquatic life. Water quality degraded enough to preclude mussel habitation.

*the number of individuals found per most recent comprehensive population survey were used to rank Abundance when CPUE information was not available.

Table 4.5. The estimated current condition of Texas Heelsplitter populations*; where high condition = 3 (green box), moderate condition = 2 (yellow box), low condition = 1 (red box), and extirpated/functionally extirpated = 0 (grey box).

		Population Factors			Habitat Factors			
River Basin	**Population**	**Occupied Habitat**	**Abundance**	**Reproduction/ Recruitment**	**Habitat Structure/ Substrate**	**Hydrology**	**Water Quality**	**Overall Condition** (Viability)
Sabine	**Sabine River/ Toledo Bend**	3	0	1	3	2	3	0**
Neches	**Neches River/ B.A. Steinhagen**	3	1	1	3	3	2	1**
	Lower Neches River	3	1	1	3	2	2	1**
Trinity	**Grapevine Lake**	na	0	1	3	2	2	0**
	Trinity River/ Lake Livingston	3	1	1	1	1	2	1**

*See Appendix B, Table B.1 for supporting information used to score population and habitat factors.

**Indicates focal areas where overall condition was capped by abundance (i.e., scored lower).

na = not applicable (i.e., not applicable to reservoirs).

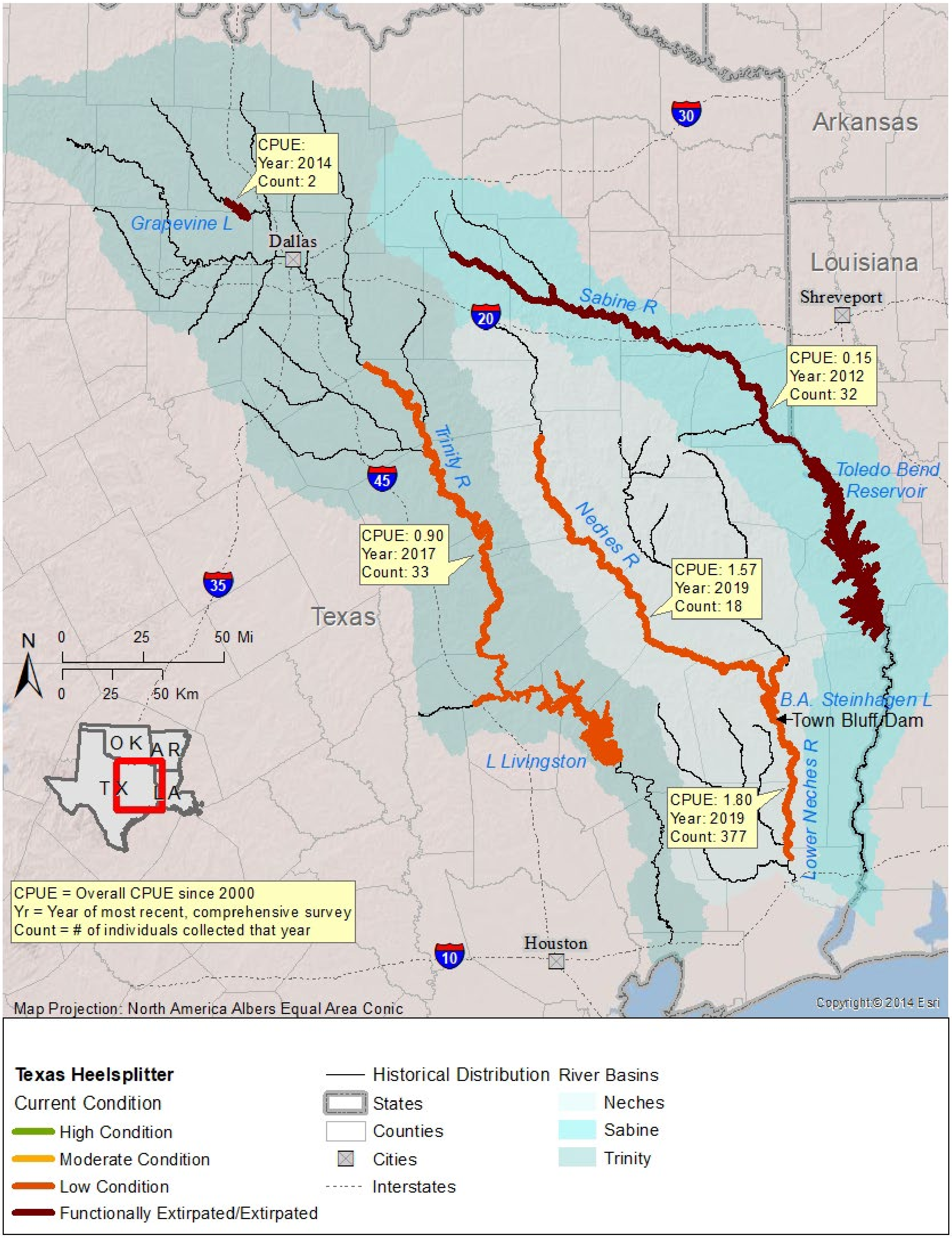


Figure 4.2. *Location and estimated condition of 5 remaining Texas Heelsplitter populations within the historical range of the species.*

4.C.2. CURRENT POPULATION RESILIENCY

Currently, Texas Heelsplitter are known to exist as five populations occurring in three adjacent river basins: the Neches, Sabine, and Trinity.

The Neches River basin in Texas has two populations of Texas Heelsplitter: Neches River/B.A. Steinhagen population and Lower Neches River population. The Neches River/B.A. Steinhagen and Lower Neches River populations are hydrologically isolated from each other by Town Bluff Dam, an impoundment that forms B.A. Steinhagen Reservoir. The Neches River population extends 225 miles on the mainstem from just below Lake Palestine to B.A. Steinhagen Reservoir and includes the portion of main stem Angelina River between B.A. Steinhagen and Sam Rayburn reservoirs. The Neches River population current condition evaluation determined occupied reach habitat length, habitat structure/substrate, and hydrology were high condition; water quality in moderate condition; and abundance and reproduction/recruitment in low condition due to low CPUE or lack of reported juvenile or gravid female observations (Randklev 2018, entire; Bio-West 2019, unpublished data). The Lower Neches River population current condition evaluation found that occupied habitat reach length and habitat structure/substrate were in high condition; hydrology and water quality were in moderate condition; and abundance and reproduction/recruitment were in low condition due to low reported CPUE and numbers of reported juveniles or gravid females (Tables 4.4 and 4.5). The Neches River population and the Lower Neches River population have a low overall current condition, resulting in low resiliency for both populations.

The Sabine River basin has one Texas Heelsplitter population in Texas, which marginally extends into Louisiana. The Sabine River population current conditions evaluation determined that water quality, habitat structure/substrate, and occupied habitat reach length were high condition; hydrology in moderate condition; reproduction/recruitment in low condition due to a lack of reported juvenile or gravid female presence data; and abundance condition was determined to be functionally extirpated/extirpated due to low CPUE (Tables 4.4 and 4.5; Randklev 2018, entire). The current condition of this population is functionally extirpated/extirpated and; therefore, has little to no resiliency.

The Grapevine Lake and Trinity River/Lake Livingston populations, located within the Trinity River basin in Texas, are hydrologically isolated from one another by the dam that forms Grapevine Lake. The Grapevine Lake population current condition evaluation found habitat structure/substrate to be in high condition; hydrology and water quality in moderate condition; reproduction/recruitment in low condition; and abundance was determined to be functionally extirpated/extirpated due to low number of individuals observed (Randklev 2018, entire). The Trinity River population current condition evaluation resulted in occupied habitat reach length found in high condition and habitat structure/substrate in moderate condition; the remaining population and habitat factors were determined to be low condition, primarily attributed to impacts associated with hydrology changes within the Trinity River basin and low reported CPUE and numbers of reported juveniles or gravid females (Tables 4.4. and 4.5). The Grapevine Lake population is considered functionally extirpated/extirpated, while the Trinity River/Lake Livingston population has a low overall current condition and low resiliency.

4.C.3. CURRENT SPECIES REPRESENTATION

We consider the Texas Heelsplitter to have representation in the form of genetic, geographic, and ecological diversity in the three currently occupied river basins. Because there are no freshwater connections between the three basins, we treated each river basin as separate areas of representation.

4.C.4. CURRENT SPECIES REDUNDANCY

Within the identified Texas Heelsplitter representation areas (Neches, Sabine, and Trinity River basins), only the Neches and Trinity River basins have at least one known current viable population (the Sabine River/Toledo Bend population in the Sabine River basin and Grapevine Lake in the Trinity River basin are considered functionally extirpated/extirpated). The Neches River basin has two currently viable populations (Neches River and Lower Neches River populations); however, these populations are hydrologically isolated, and therefore provide only minimal redundancy.

4.D. SUMMARY OF CURRENT CONDITIONS OF LOUISIANA PIGTOE AND TEXAS HEELSPLITTER

Louisiana Pigtoe and Texas Heelsplitter exhibit various levels of resiliency, redundancy, and representation across the major river basins in which they occur. However, no population seems to contain all of the habitat and population factors necessary to warrant strong, healthy mussel populations. Given our analysis of current condition, only two Louisiana Pigtoe populations were considered to have high current condition overall (i.e., Neches and Cossatot rivers; Table 4.3), and no Texas Heelsplitter populations are in high condition (Table 4.5). While other populations have aspects, or factors, that are in high condition (such as occupied habitat length or habitat structure/substrate) none of those populations have all of the factors necessary to support a highly resilient population. Four populations of the Louisiana Pigtoe and Texas Heelsplitter are considered functionally extirpated/extirpated, meaning abundance is too low to support viability of the population, including the Lower Little River (tributary to the Red River) and Sabine River populations for the Louisiana Pigtoe, and Sabine River/Toledo Bend and Grapevine Lake populations for the Texas Heelsplitter.

CHAPTER 5 - FACTORS INFLUENCING VIABILITY

This chapter evaluates the past, current, and future factors that may affect the long-term viability of Louisiana Pigtoe and Texas Heelsplitter. Each factor is discussed below and explored further in the "Cause and Effects Tables" attached to this report (Appendix B**)**. The Cause and Effects Tables analyze, in detail, the pathways through which each factor influences a species at both the individual and population level. Each factor is also examined temporally to determine the magnitude of potential impacts on the status of the species from a historical, current, and future perspective. These factors include: 1) water quality changes, 2) altered hydrology, 3) substrate changes, 4) habitat fragmentation, 5) direct mortality, and 6) invasive species. Climate change, which has the unique ability to influence all six factors, is also briefly mentioned toward the end of the chapter and is a key component of our analysis in Chapter 6 where we take a closer look at future conditions.

The current and potential future effects of the six factors, along with current estimates of distribution and abundance, determine present viability, and therefore future vulnerability to extinction. The factors we chose to examine are based on known stressors that either influence Louisiana Pigtoe and Texas Heelsplitter directly or influence the resources upon which mussels rely for survival, growth, and reproduction, as well as a discussion on the sources of those stressors. For more information about how each factor influences species survival, see Appendix B. Environmental stressors that are not known to affect Louisiana Pigtoe and Texas Heelsplitter populations are not discussed in this SSA report.

5.A. CHANGES IN WATER QUALITY

Freshwater mussels require water in sufficient quantity and quality on a consistent basis to complete their life cycles and those of their host fishes. Like many rare species, along with natural perturbations that exert pressure on populations and influence survival, habitat for freshwater mussels is impacted by a myriad of anthropogenic activities. These activities, such as residential development and agriculture, place increasing demands on natural resources, particularly water, which can have deleterious effects on both water quality and quantity.

Water quality can be degraded through contamination or alteration of water chemistry. Environmental contaminants include a broad array of natural, synthetic, and chemical substances introduced to the environment that can be hazardous to living organisms. Chemical contaminants are ubiquitous throughout the environment and are a major contributor to the current declining status of freshwater mussel species nationwide (Augspurger et al. 2007, p. 2025). Contaminants that enter the environment are generally categorized by their origin as either coming from point sources such as hazardous spills, industrial wastewater, and municipal effluents, or non-point sources such as urban stormwater and agricultural runoff. These discharges can introduce a variety of pollutants to air, water and soil, including organic compounds, trace metals, pesticides, plastics, petroleum hydrocarbons, flame retardants, and a wide variety of emerging contaminants (e.g., pharmaceuticals and personal care products) that comprise some 85,000 chemicals in commerce today and are routinely released into the aquatic environment (EPA 2018, p. 1). The extent to which environmental contaminants adversely affect aquatic biota can vary depending on many site-specific variables (e.g., the concentration of the pollutant, the volume discharged, and the timing of the release), but species diversity and abundance consistently ranks lower in waters that are known to be polluted or otherwise impaired by contaminants. For example, freshwater mussels are not generally found for many miles downstream of municipal wastewater treatment plants (WWTP)(Gillis et al. 2017, p. 460; Goudreau et al. 1993, p. 211; Horne and McIntosh 1979, p. 119). Transplanted common freshwater mussels (*Amblema plicata* and *Corbicula fluminea*) showed reduced growth and survival below a WWTP outfall relative to sites located upstream of the WWTP in Wilbager Creek (a tributary to the Colorado River in Travis County, Texas); water chemistry was altered by the wastewater flows at downstream sites, with elevated constituents in the water column that included copper, potassium, magnesium, and zinc (Nobles and Zhang 2015, p.11; Duncan and Nobles 2012, p. 8).

Although municipal wastewater effluents are nutrient rich and contain a variety of pollutants that can affect water quality, ammonia is of particular concern below wastewater treatment plant outfalls because freshwater mussels have been shown to be particularly sensitive to increases in ammonia levels (Augspurger et al. 2003, p. 2569). Elevated concentrations of un-ionized ammonia (NH_3) in the interstitial spaces of benthic habitats (> 0.2 parts per billion) have been implicated in the reproductive failure of Eastern Elliptio (*Elliptio complanata*) freshwater mussel populations (Strayer and Malcom 2012, pp. 1787-8), and sub-lethal effects (valve closures) have recently been described as TAN approaches 2.0 milligrams per liter (mg/L or ppm; Bonner et al. 2018, p. 186). Waters near intensive agricultural operations such as poultry farms, processing plants, and confined animal feeding operations that house large concentrations of animals producing ammonia waste are also at risk of contamination. Quantitative estimates of the effects of un-ionized ammonia in the water column are currently unknown, and relationships between TAN and un-ionized ammonia (NH_3) are dependent on pH and temperature (see inset). Recent laboratory studies suggest that for Pimpleback (*Cyclonaias pustulosa*; a species native to the eastern United States and entire Mississippi drainage), the revised EPA ammonia benchmarks are sufficient to protect from short-term effects of ammonia on resting metabolic rate and ability to extract oxygen even under low oxygen conditions (Bonner et al. 2018, p. 151). However, some sources are continuous and the long-term effects of chronic ammonia exposure (i.e., years or decades) to freshwater mussels has yet to be experimentally investigated. Although a comprehensive review of ammonia related impacts to Louisiana Pigtoe and Texas Heelsplitter is beyond the scope of this document, municipal wastewater is known to contain both ionized and un-ionized ammonia and wastewater discharge permits issued by Texas Commission on Environmental Quality (TCEQ) do not always impose limits on ammonia, particularly for smaller volume dischargers. Thus, at a minimum there are likely to be elevated concentrations of ammonia in the immediate mixing zone of some WWTP outfalls, and in some cases, impacts will persist for some distance downstream. To give insight into the potential scope of WWTP related impacts, there are approximately 386 discharge permits issued for the Trinity River basin alone from its headwaters above the Dallas-Fort Worth metroplex down to the Gulf of Mexico (TCEQ 2018b, entire). The San Jacinto Basin, although geographically smaller than most other basins in Texas, has approximately 1,052 WWTP outfalls, while the Neches and Sabine Rivers have 218 and 191

Ammonia toxicity explained *in* Bonner et al. 2018, p. 147-8:

"Ammonia in surface waters is typically reported as total ammonia nitrogen (TAN). This refers to the combined concentration of nitrogen (mg/L) occurring in the two co-existing forms of ammonia, ionized (NH_4^+) and un-ionized (NH_3). Un-ionized ammonia is the most toxic form. The proportion of un-ionized to ionized (NH_3:NH_4^+) ammonia increases with increasing pH and temperature. Thus, ammonia becomes more toxic with increases in temperature and/or pH even if the concentration of ammonia, measured as TAN, remains the same. The U.S. EPA 2013 ammonia benchmark is 17 mg TAN/L for acute (1 hour average) exposure and 1.9 mg TAN/L for chronic (30 day rolling average) exposure. These benchmarks are referred to as "criterion minimum concentrations" (CMC) and represent a concentration that is expected to be lethal to < 50% of individuals in sensitive species. They specifically apply to a pH of 7 and a temperature of 20°C during the summer months. The toxicity of 17 (acute) and 1.9 (chronic) mg TAN/L benchmark concentrations would therefore increase and may no longer be sufficiently protective of unionid mussels. The EPA is cognizant of this issue and provides tables to adjust benchmark concentrations for specific temperature and pH values. Un-ionized ammonia can affect organisms such as mussels via multiple mechanisms that increase ventilation rates (volume of water passing through gills per unit time), gill damage, and a reduction in the ability of blood (hemolymph) to carry oxygen."

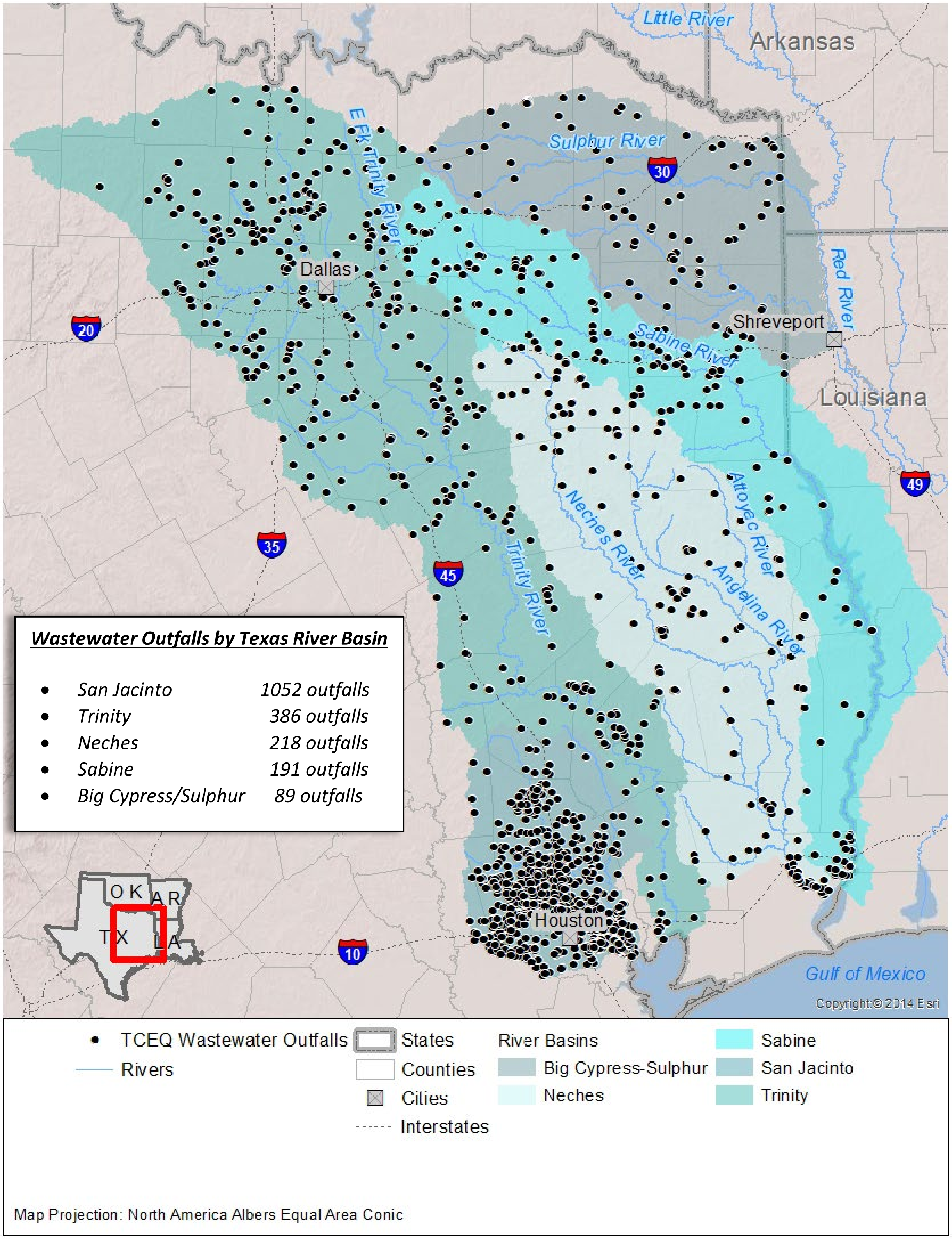


*Figure 5.1. Wastewater discharge permits issued by the Texas Commission on Environmental Quality within the range of Louisiana Pigtoe and Texas Heelsplitter (analysis limited to Texas; TCEQ 2018b, entire)***.**

outfalls respectively (Figure 5.1). In addition, some industrial permits such as animal processing facilities can discharge millions of gallons per day and have ammonia limits in the range of 4 mg/L, which exceeds levels that inhibited growth in juvenile Fatmucket (*Lampsilis siliquoidea*) and Rainbow Mussel (*Villosa iris*) during 28 day chronic tests (0.37 to 1.2 mg total ammonia N/L; no-observed-effect concentration and lowest-observed-effect concentration, respectively) (Wang et al. 2007, entire). Immature mussels (i.e., juveniles and glochidia) are especially sensitive to water quality degradation and contaminants (Cope et al. 2008, p. 456, Wang et al. 2017, p. 791-792; Wang et al. 2018, p. 3041).

Another common type of water quality degradation is the alteration of basic water chemistry, including changes to water quality parameters such as dissolved oxygen, temperature, total dissolved solids (TDS) and salinity. Dissolved oxygen levels are influenced by temperature (i.e., as temperatures increase, dissolved oxygen levels decrease) and may be reduced from increased nutrient inputs or other sources of organic matter that increase the biochemical oxygen demand in the water column as microorganisms decompose waste. Organic waste can originate from stormwater, agriculture, irrigation runoff or wastewater effluent, and juvenile mussels seem to be particularly sensitive to low dissolved oxygen with sub-lethal effects evident at 2 ppm and lethal effects at 1.3 ppm after just 48 hours (Sparks and Strayer 1998, pp. 132-133). Although some aquatic organisms tolerate dissolved oxygen levels below 3 ppm, most prefer levels somewhere between 4 ppm and supersaturation (i.e., excessively high dissolved oxygen). Increases in water temperature ($\geq$27° C for sensitive species) resulting from water diversions, climate change, or low flows during droughts can increase the toxicity of many pollutants and exacerbate low dissolved oxygen levels, in addition to other drought-related effects on both juvenile and adult mussels.

Total dissolved solids, a measure of the mineral content of water (i.e., inorganic salts, metals, cations or anions dissolved in water, including calcium, magnesium, potassium, sodium, bicarbonates, chlorides, and sulfates), is commonly elevated in watersheds impacted by a variety of industrial, commercial, urban and agricultural activities, and has been associated with acute and chronic toxicity to aquatic organisms. Total dissolved solids are a good overall indicator of water quality and can be measured indirectly using conductivity; therefore, watersheds with increasing trends in conductivity or TDS are experiencing declines in water quality that can be harmful to mussels and other aquatic organisms. Increasing trends in TDS are not uncommon in watersheds impacted by anthropogenic activities. For example, water quality samples taken on Segment 0402 of Big Cypress Bayou near the confluence with Little Cypress Bayou showed a significant increasing trend in conductivity, with values rising from 120 uS/cm in 1998 to 190 uS/cm in 2012, likely due to changing land uses and subsequent increases in point and non-point source pollution (TCEQ 2014, pp. 20-21).

Mussels are also sensitive to elevated salinity, which is a measure of dissolved salts like chloride and sodium that are a component of TDS, such that, the distribution of mussels is naturally limited in the lower basins where conditions become unfavorable from the intrusion of brackish and saline water near the coast. Freshwater areas within these lower basins can be affected by storm surges or inclement weather, such as hurricanes, as saline water is carried inland. These salt water deposits can harm freshwater biota, including mussels, depending largely on the volume introduced and the amount of time saline conditions persist. Salinity in river water is diluted by surface flow and as surface flow decreases the influence of salt concentrations increase, resulting in adverse effects on freshwater mussels. Even low levels of salinity (2-4 parts per thousand (ppt)) can have substantial negative effects on reproductive success, metabolic rates, and survival of freshwater mussels (Blakeslee et al. 2013, p. 2853). Bonner et al. (2018, pp. 155-6) suggest that the behavioral response of valve closure to high salinity concentrations (> 2 ppt) is the likely mechanism for reduced metabolic rates, reduced feeding, and reduced reproductive success based on reported sub-lethal effects of salinity > 2 ppt for Texas Pimpleback, which closed tightly when exposed to salinity > 4 ppt for 7 days. The extent to which salinity currently affects freshwater mussel survival and reproduction near coastal areas is unknown, but the impacts will likely increase with climate change as weather related events increase the frequency and intensity of storms.

Contaminants released during accidental spills of chemicals, crude oil, or other hazardous materials are also a concern to water quality, as they often impact adjacent rivers, streams and waterbodies. Texas leads the nation in crude oil and natural gas production with more than 270,000 active oil and gas wells, in addition to 448,446 miles of pipelines and associated infrastructure needed to move product from wells to refineries for processing (Figure 5.2; RRC 2021a, 2021b, and 2021c, entire). Various chemicals, refined fuels like diesel, and wastewater related to oil and natural gas exploration are also routinely transported along Texas highways. These facilities and equipment used for extraction, transportation and refinement of hazardous materials are all potential sources of hazardous spills, which occur with regularity throughout the state and can originate from human error, equipment failure, or catastrophic events like industrial accidents, fires or floods. Although spills are relatively short-term events and may be localized, depending on the types of substances and volume released, water resources nearby can be severely impacted and degraded for years after the incident along with the biological resources that inhabit the area.

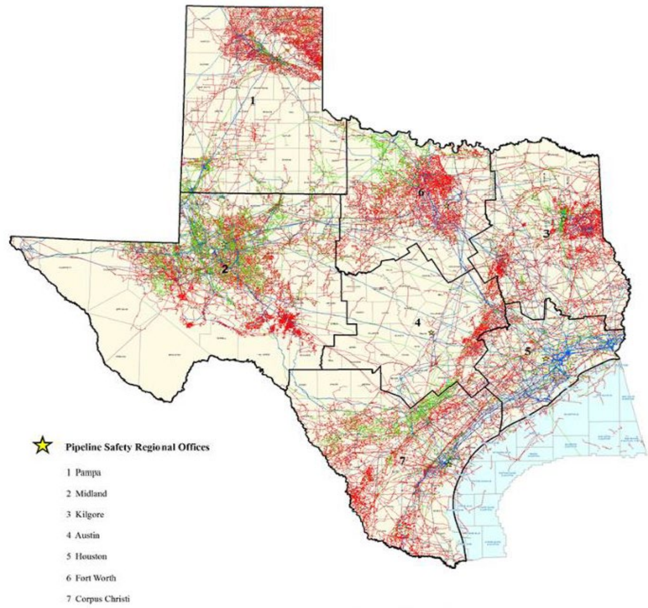


Figure 5.2. *Texas Railroad Commission map showing extensive pipeline network used to carry natural gas (red), crude oil (green), and hazardous liquids (blue) throughout the state (as of January 2018).*

Water quality and quantity are interdependent, so reductions in surface flow caused by drought, instream diversions, or groundwater extraction serve to concentrate contaminants from point and non-point source pollution that would otherwise be diluted. For example, point source discharges of industrial or municipal wastewater inherently pose a greater risk to aquatic biota under low flow conditions as concentrations of pollutants and water temperatures increase. Drought conditions can place additional stressors on stream systems beyond reduced flows by exacerbating contaminant related effects to aquatic biota, including freshwater mussels. Not only can temperature be a biological, physical, and chemical stressor, the toxicity of many pollutants to aquatic organisms increases at higher temperatures (e.g., ammonia, mercury), which is further exacerbated by the increased metabolic activity (e.g., higher respiration rates) experienced by organisms as they try to adapt to hotter conditions within the water column (Noyes *et al.* 2009, p. 979; Ganser *et al.* 2013, p. 1172; Patra *et al.* 2015, p. 1814). We foresee threats to water quality increasing into the future due to the effects of climate change as demand and competition for limited water resources grows. For additional information and a more comprehensive discussion of water quality requirements for aquatic species in Texas, the reader is referred to USFWS (2006, entire).

5.B. ALTERED HYDROLOGY

In this report, altered hydrology refers to anthropogenic changes to historical flow regimes that result in degradation of Louisiana Pigtoe and Texas Heelsplitter habitat. The changes to flow originate from a variety of activities, resulting in either an increase or decrease in flows (e.g., magnitude, duration, intensity) beyond natural fluctuations that occurred historically, and in some cases these changes exceed levels tolerated by mussels. While we recognize changes to flow occur naturally, such as floods and droughts, the focus of our discussion is related to changes to flow that are directly or indirectly related to human activity. Altered hydrology (leading to inundation, low flow, or high flow conditions) may reduce the quality of affected habitats to the point where they are no longer suitable for freshwater mussels. While Louisiana Pigtoe and

Texas Heelsplitter have adapted to survive natural fluctuations in flows, populations that experience sustained higher than normal flows, prolonged flooding or unnatural fluctuations in the frequency or intensity of high/low flows, or extended (or repeated) drying events, will not persist. Although some watersheds have been more heavily impacted than others, virtually every watershed within the range of these two freshwater mussels has experienced some level of anthropogenic-induced change to the hydrology during the 20th century, a trend that will likely continue into the 21st century, particularly in areas with rapid population growth.

Inundation of previously free-flowing rivers and streams by impoundments has arguably had the single largest human-related impact on the distribution of freshwater mussels. The construction of reservoirs and other impoundments permanently alters the hydrology, and hence, the ecology of rivers, often with deleterious effects to water quality, water quantity, host fish movement and dispersal of mussel glochidia, nutrient cycling, sediment deposition, fate and transport of contaminants, and numerous other changes to the physical, chemical, and biological characteristics of affected areas (upstream and downstream). In this section, we discuss how the close relationship of flow to mussels makes them uniquely vulnerable to hydrology changes.

Both mussel species are adapted to flowing water (lotic habitats) rather than standing water (lentic habitats). Louisiana Pigtoe require free-flowing water to survive and prolonged inundation in non-flowing conditions is not suitable habitat for the species. Like the Louisiana Pigtoe, the Texas Heelsplitter evolved in flowing conditions but they have also been observed in lentic habitats and appear to be tolerant of reservoir conditions. There is, however, uncertainty about whether populations that occur in lacustrine environments function in the same manner as those in lotic habitats, and the mechanisms that allow the Texas Heelsplitter to tolerate reservoirs are poorly understood (Randklev 2019a, p. 2). Some have suggested Texas Heelsplitter may occur in higher densities, and hence favor, areas of reservoirs that are influenced by stream inflows where conditions more closely resemble their preferred riverine habitat (Whisenant 2019, p. 1; Neck and Howells 1995, p. 15).

Inundation of mussel habitat has primarily occurred upstream of dams, including large structures on public land such as Toledo Bend Reservoir and other major flood control and water supply reservoirs, and smaller structures like low water vehicle crossings and diversion dams typically found along tributaries on privately-owned land. These structures alter the hydrology of rivers by slowing, impeding or diverting normal flow patterns, causing a myriad of other changes to the aquatic environment. Inundation alters natural sediment deposition by increasing deposition in some areas and eliminating the interstitial spaces that Louisiana Pigtoe and Texas Heelsplitter inhabit. Inundation also includes the effects of reservoir releases where the frequency and magnitude of flows and variations in surface water elevation can make habitat unsuitable for these species. In large reservoirs that release water from the hypolimnion, the deeper water is cold and often devoid of oxygen and necessary nutrients, which can adversely affect mussel survival. Cold water can stunt mussel growth and delay or hinder spawning (Vaughn and Taylor 1999, p. 917). Reservoirs like Broken Bow Lake in southeast Oklahoma that release cold water from the bottom of the reservoir (in part to support a non-native rainbow and brown trout recreational fishery), can affect water temperatures for miles downstream. These cold releases create an extinction gradient, where freshwater mussels are absent or presence is low near the dam, and abundance does not rebound until some distance downstream where ambient conditions raise the water temperature to within the tolerance limits of mussels (Davidson et al. 2014, p. 29; Vaugh and Taylor 1999, pp. 915, 916).

The construction of dams for flood control and drinking water supply, and the subsequent management of water releases from those reservoirs (e.g., timing, intensity, and duration), has significant impacts on the natural function and hydrology of rivers and streams. For example, dams trap sediment in reservoirs and managed releases typically do not conform to the natural flow regime, often resulting in higher base flows, and peak flows of reduced intensity but longer duration. The additional shear stress caused by these sustained high base flows can incise channels, erode river banks, scour mussel beds, and remove substrate preferred by mussels. Over time, the physical force of these higher base flows can dislodge mussels from the sediment and permanently alter the geomorphology of rivers. Rivers transport not only water but also sediment, which is transported mostly as solids suspended in the water column. The majority of sediment transport occurs during floods (Kondolf 1997, p.533; Clark and Mangham 2019, pp. 6-7). The increase in flooding severity results in greater sediment transport, with important effects to substrate stability and benthic habitats for freshwater mussels, as well as other organisms that are dependent on stable benthic habitats. Further, water released by dams is usually clear due to reduced sediment load, and is considered “hungry water because the excess energy is typically expended on erosion of the channel bed and banks…resulting in incision (downcutting of the bed) and coarsening of the bed material until a new equilibrium is reached” (Kondolf 1997, p.535). The extent to which downcutting and erosion occurs as a result of dam releases varies depending on the volume of flows and geomorphology of the river downstream, but in some cases leads to bank collapse, burial of mussel beds, and mortality. Conversely, depending on how dam releases are conducted, reduced flood peaks can lead to accumulations of fine sediment in the river bed (i.e., loss of flushing flows, Kondolf 1997, pp. 535, 548).

Operation of reservoirs for flood-control, water-supply, and recreation results in altered hydrologic regimes, including an attenuation of both high- and low-flow events. Flood control dams store flood waters and then release them in a controlled manner. Extended release of these flood waters can result in significant scour, and loss of substrates that provide mussel habitat. The changes to flood flows also alter sediment dynamics, as sediments are trapped above and scoured below major impoundments. These changes in water and sediment transport negatively affect freshwater mussels and their habitats (Gascho-Landis and Stoeckel 2016, p. 234; Ford 2013, p.3). Evidence that Texas Heelsplitter are able to tolerate reservoir conditions leads us to believe the overall impacts of reservoirs may be more pronounced for Louisiana Pigtoe (Howells 2010b, p. 3); however, this is speculative since to our knowledge there have been no studies to elucidate this issue.

Flow loss and scour - Very low flows and water levels are also detrimental to Louisiana Pigtoe and Texas Heelsplitter populations. Droughts that occurred in the recent past led to extremely low flows in several east Texas rivers. Some rivers, or portions thereof, are resilient to drought because they are spring-fed (Calcasieu, Neches), contain large volumes of water (Trinity), have large reservoirs in the upper reaches that release water for downstream users (all, excluding Calcasieu), or have significant return flows (Pearl, Sabine, Trinity); however, drought in combination with increasing trends in groundwater extraction may lead to lower river flows of longer duration than previously recorded. Reservoir releases can be managed to some extent during drought conditions to prevent complete dewatering below reservoirs, but in many cases dam operators must stop releases during droughts to conserve water and protect water supplies, leaving mussels vulnerable to desiccation. The same limitation applies during major floods, where dam operators have little choice but to maximize flood releases to protect public safety and property, which can negatively affect mussels downstream.

Streamflow and overall discharge for rivers inhabited by Louisiana Pigtoe and Texas Heelsplitter are expected to decline due to climate change and projected increases in temperatures and evaporation rates, resulting in more frequent and intense droughts (Lafontaine et al. 2019, entire). Return flows, consisting primarily of treated municipal wastewater, are projected to continue to increase in areas with population growth and may serve to ameliorate some of the effects of climate change downstream of metropolitan areas, albeit with notable impacts to water quality; however, these benefits may become less significant as municipalities increase wastewater reuse as a conservation measure. The Trinity River, for example, has been a significantly modified, highly controlled and regulated system since the 1960s, with low flows steadily increasing as the population has grown, resulting in base flows that are significantly higher compared to historical flows (Clark and Mangham 2019, p. 9). The increase in base flows can be attributed to substantial return flows from Dallas/Fort Worth metropolitan area wastewater treatment plants and are projected to continue to increase in the future. Surface and alluvial aquifer groundwater withdrawals will likely increase in the future due to the effects of more intense droughts, with reductions in streamflows putting an additional strain on aquatic resources. With the exception of stream segments where municipal effluent return flows supplement base flows, most streams experience lower base flows and reduced high flow events after major reservoirs are constructed (USGS 2008, pp. 964, 966).

Many streams within the range of these two freshwater mussel species receive significant groundwater inputs from multiple springs associated with aquifers. As spring flows decline due to drought, climate change, or groundwater pumping, habitat for freshwater mussels in affected streams is reduced and could eventually cease to exist. While Louisiana Pigtoe and Texas Heelsplitter may survive short periods of low flow, as low flows persist, mussels can be subjected to oxygen deprivation, increased water temperature, and, ultimately, stranding, which leads to reduced survivorship, reproduction, and recruitment to the population. Likewise, high-flow events can lead to increased risk of mortality through physical removal, transport, or burial of mussels as unstable substrates are transported downstream by flood waters (entrainment) and dislodged mussels are later redeposited in locations that may not be suitable habitat. Low flow events also lead to an increased risk of desiccation (physical stranding and drying) and exposure to elevated water temperature and other water quality degradations, such as more concentrated contaminants, as well as to predation.

The distribution of mussel communities and their habitats is affected by large floods returning at least once during the typical life span of an individual mussel (generally from 3 to 30 years), as mediated by the presence of flow refuges, where shear stress is relatively low, sediments are relatively stable, and mussels "must either tolerate high-frequency disturbances or be eliminated and can colonize (only) areas that are infrequently disturbed between events" (Strayer 1999, pp. 468-9). Shear stress and relative shear stress (RSS) are limiting to mussel abundance and species richness (Randklev et al. 2017a, p. 7) and riffle habitats may be more resilient to high flow events than littoral (bank) habitats.

Louisiana Pigtoe and Texas Heelsplitter undoubtedly evolved in the presence of extreme hydrological conditions to some degree, including severe droughts leading to dewatering, and heavy rains leading to damaging scour events and movement of mussels and substrate, although the frequency, duration, and intensity of these events may be different from what is observed today. The natural drought/flood cycle in east Texas can be characterized by long periods of time with little or no rain, interrupted by short periods of heavy rain that often result in flooding. These same patterns led to the development of flood control and storage reservoirs throughout Texas in the twentieth century. Howells (2000) provides a summary of drought conditions in Texas from 1995-1999, characterized by prolonged drought conditions punctuated by severe floods, and their impacts on native unionids, reporting that "although no sampling efforts were mounted to document [the] impact on rare endemic unionids…, [some] species… were almost certainly reduced in numbers, especially at sites that dried completely" (p.ii). It follows that given the variable climate of east Texas; mussels must have life history strategies, and other adaptations, that allow them to persist by withstanding severe conditions, and/or repopulating during more favorable conditions. However, there are limits to the ability of mussels to respond to increasing variability, frequency, and severity of extreme weather events, which is believed to be a contributing factor to the contraction of populations for both species.

Another source of alteration to hydrology is from sand and gravel mining. Sand and gravel can be mined directly from rivers or from adjacent alluvial deposits, and instream gravels often require less processing and are thus more attractive from a business perspective (Kondolf 1997, p. 541). Instream mining directly impacts river habitats by removal of substrates used by mussels, and can indirectly affect river habitats through channel incision, bed coarsening, and lateral channel instability (Kondolf 1997, p. 541). Excavation of pits in or near to the channel can create a knickpoint, which can contribute to erosion (and mobilization of substrate) associated with head cutting (Kondolf 1997, p. 541). Pits associated with off-channel mining of the floodplain can become involved during floods, such that the pits become hydrologically connected, and thus can affect sediment dynamics in the stream or river (Kondolf 1997, p. 545). Sand and gravel mines occurred historically and continue to operate in some basins throughout the range of Louisiana Pigtoe and Texas Heelsplitter, including two operations noted within the Bayou Anacoco focal area and one within the San Jacinto focal area during our review.

Due to the importance of hydrology to Louisiana Pigtoe and Texas Heelsplitter, in 2018 the Service contracted the Texas A&M University's Natural Resources Institute to conduct research on hydrologic changes that have occurred in east Texas rivers and examine potential impacts to freshwater mussels. This two year study entitled "Assessment and Review of Hydrological Relationships for Mussels in East Texas" utilized historical U.S. Geological Survey stream gage data to evaluate changes to eleven flow parameters assessed using Indicators of Hydrologic Alteration (IHA) at 43 gages over a 50 year period (1968 – 2018)(Figure 5.3). Preliminary findings contained in the 2019 Interim Report indicate significant changes to specific measured hydrologic parameters in all four river basins reviewed, with basins experiencing change ranked from high to low as follows: Trinity River, Sabine River, Big Cypress Bayou, and Neches River (see Figure 5.4). To determine the influence these changes to flow had on Louisiana Pigtoe and Texas Heelsplitter (i.e., clarify mussel-flow relationships), the gage data were paired with records from approximately 500 mussel surveys conducted within 20 kilometers of the 43 gages (24 gages for the Trinity, nine for the Neches, six for the Sabine, and four for Big Cypress). Although evaluation of mussel-flow relationships is ongoing and a final report is not due until the Fall of 2020, based on quantile regression models there are flow parameters that appear to be limiting to Louisiana Pigtoe and Texas Heelsplitter. Specifically, changes to the number of days with zero flow was limiting for Louisiana Pigtoe, and the number of high pulses was limiting for Texas Heelsplitter. In summary, results to date indicate natural flow regimes have been altered in east Texas rivers, as was expected, which has led to modification of instream habitats and contributed to declines in freshwater mussels. These findings agree with the opinion of many experts who believe (1) portions of the Trinity River have been significantly modified and may no longer support mussels (particularly in the upper basin where stream hydrology and geomorphology have been permanently altered), and (2) the Neches River is least altered and has some of the best remaining mussel habitat, along with the most abundant and diverse mussel populations, left in east Texas.

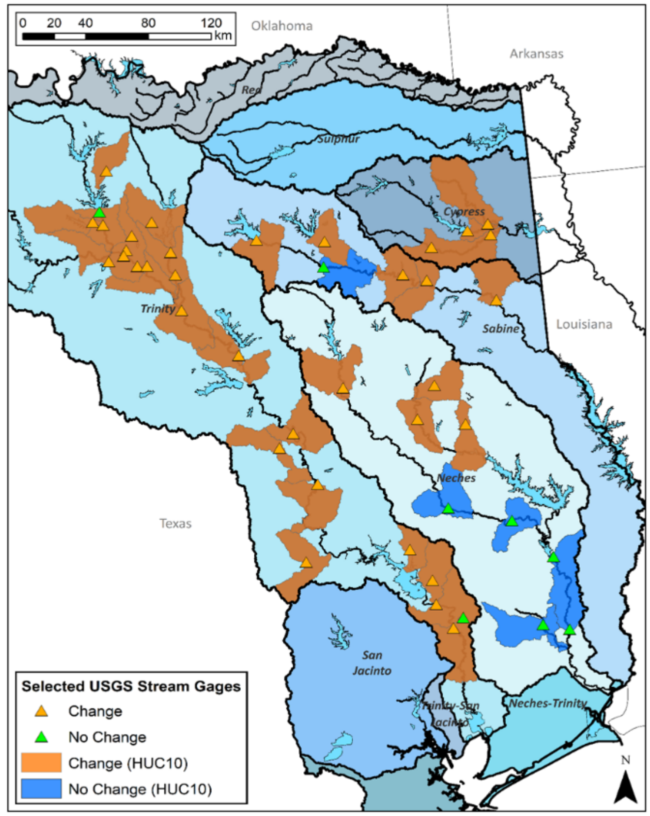


Figure 5.3. *Map of USGS stream gages evaluated for changes to flow from 1968-2018 based on HUC10 watersheds. HUCs highlighted in orange indicate at least one gaging station showed a significant change over time in one or more of the 11 flow parameters analyzed.(HUCs in blue show no change in any of the 11 flow parameters). (Khan and Randklev, 2019 Interim Report, pg. 7).*

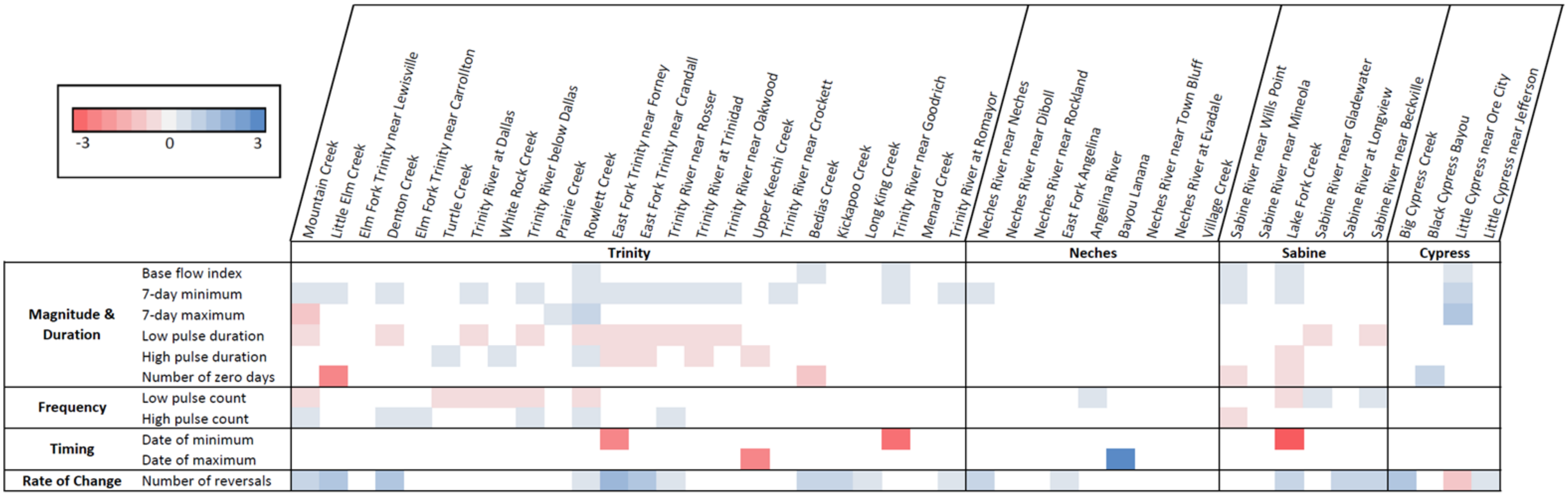


Figure 5.4. *Changes to eleven flow metrics over time at 43 gages within multiple drainage basins of four east Texas rivers. Red denotes a statistically significant ($p \leq 0.05$) decrease in flow metric over time, and blue denotes a significant ($p \leq 0.05$) increase in flow metric over time. The degree of the slope is indicated by a color gradient, with darker colors representing more substantial changes to flow. White spaces indicate no significant change in flow parameter over time, representing mussel habitat that has been least impacted by hydrologic alterations (Khan and Randklev 2019, Interim Report, pg. 8). See Appendix D for definitions of flow metrics.*

5.C. CHANGES TO HABITAT STRUCTURE/SUBSTRATE

Juvenile and adult Louisiana Pigtoe and Texas Heelsplitter inhabit microhabitat along river stream beds that have abundant interstitial spaces or small openings in an otherwise closed matrix of substrate, created by gravel, cobble, boulders, bedrock crevices, tree roots, and other vegetation, with some amount of fine sediment (i.e., clay and silt) necessary to provide appropriate shelter. However, excessive amounts of fine sediments can reduce available microhabitat in an otherwise suitable mussel bed by filling in these interstitial spaces, effectively smothering mussels in place. Louisiana Pigtoe and Texas Heelsplitter generally require stable substrates, and loose silt deposits do not generally provide adequate substrate stability. Interstitial spaces provide essential habitat for juvenile mussels in particular, offering protection from predation and vital nutrients. Juvenile freshwater mussels burrow into interstitial substrates, making them particularly susceptible to degradation of this habitat feature. When clogged with sand or silt, interstitial flow rates and spaces may become reduced (Box and Mossa 1999, p. 100) and no longer provide suitable habitat for juveniles. While adult mussels can be physically buried by excessive sediment, "the main impacts of excess sedimentation on unionids are often sublethal" and include interference with feeding mediated by valve closure (Box and Mossa 1999, p. 101). Many land use activities can result in excessive erosion, sediment production and channel instability, including, but not limited to oil and gas development, logging, crop farming, ranching, mining, and urbanization (Arm et al. 2014, p. 114; Howells 2010b, p. 14; Arbuckle and Downing 2002, p. 311; Box and Mossa 1999, p. 102).

Under a natural flow regime, a river or stream is in equilibrium in the context of sediment load, so that sediments are naturally washed away from one microhabitat to another, the amount of sediment in the substrate is relatively stable, and different reaches within a river or stream may be aggrading or degrading sediment at any given time (Poff et al. 1997, pp. 770-772). Current (and past) human activities often result in enhanced sedimentation in river systems, including legacy sediment from past land disturbances and reservoir construction. These activities continue in many basins occupied by Louisiana Pigtoe and Texas Heelsplitter, influencing river processes and sediment dynamics (Wohl 2015, p. 31, pp. 39), with legacy effects that can result in degradation of mussel habitat. Fine sediments collect on the streambed and in crevices during low flow events, and much of the sediment is washed downstream during high flow events (also known as cleansing flows) and deposited elsewhere. However, increased frequency of low flow events (from groundwater extraction, instream surface flow diversions, and/or drought) combined with a decrease in cleansing flows (from reservoir management and drought) causes sediment to accumulate. Sediments deposited by large scale flooding or other disturbance may persist for several years until adequate cleansing flows can redistribute that sediment downstream. When water velocity decreases, which can occur from reduced streamflow or inundation, water loses its ability to carry sediment in suspension and sediment falls to the substrate, eventually smothering mussels not adapted to soft substrates (Watters 2000, p. 263). Sediment accumulation can be exacerbated when there is a simultaneous increase in the sources of fine sediments in a watershed. Within the range of Louisiana Pigtoe and Texas Heelsplitter, these sources include streambank erosion from development, agricultural activities, livestock and wildlife grazing, in-channel disturbances, roads, and crossings, among others (Poff et al. 1997, p. 773). In areas with ongoing development, runoff can transport substantial amounts of sediment from ground disturbance related to construction activities with inadequate or absent sedimentation controls. While these construction impacts can be transient (lasting only during the construction phase), the long-term effects of development on water quantity and quality are long lasting and can result in hydrological alterations as increased impervious cover increases run off and resulting shear stress causes streambank instability and additional sedimentation.

5.D. HABITAT FRAGMENTATION

Historically, the Louisiana Pigtoe and Texas Heelsplitter were likely distributed throughout the river basins described in Chapter 3. Given the reproductive ecology of both species, new areas of suitable habitat would have been colonized through movement of infested host fish, as newly metamorphosed juveniles would excyst from host fish and become established in new locations.

Today, the remaining Louisiana Pigtoe and Texas Heelsplitter populations are isolated from one another by major reservoirs such that natural recolonization of areas previously extirpated is extremely unlikely, if not impossible, due to barriers to host fish movement. With the exception of the Louisiana Pigtoe populations in the Red River basin in Arkansas and Oklahoma, there is currently no opportunity for substantial interaction among extant Louisiana Pigtoe and Texas Heelsplitter populations as over time they have become isolated from one another by reservoirs, habitat alteration, and de-watering events, among other reasons.

Instream barriers, such as reservoirs, low water crossings, and sections of dry stream bed during periods of prolonged drought, have multiple impacts on stream ecosystems. The impacts of reservoirs in particular are significant, causing permanent changes to fish movement, water quality, and hydrology, with cascading effects to river ecology and aquatic species that utilize areas downstream. Reductions in the diversity and abundance of mussels are primarily attributed to habitat shifts caused by impoundments (Neves et al. 1997, p.63), including the drastic alteration in resident fish populations and the inability of host fish to move freely between mussel populations resulting in genetic isolation. The overall distribution of mussels is, in part, a function of the dispersal of their host fish. There is limited potential for immigration between populations other than through attached glochidia being transported to a new area or to another population. Small (or fragmented) populations are more affected by this limited immigration potential because they are susceptible to genetic drift (random loss of genetic diversity) and inbreeding depression overtime (Abernathy et al. 2013, p. 25). Fuller and Doyle (2018, p. 1445-1446) suggested mussel population genetic response to habitat fragmentation may be a function of lifespan of the species; where those with "long lifespans [e.g., Louisiana Pigtoe] benefit from generational overlap that insulates their population from the genetic impacts of habitat fragmentation more than species with shorter lifespan life history strategies [e.g., Texas Heelsplitter]". At the species level, populations that are eliminated due to stochastic events cannot be recolonized naturally, leading to reduced overall redundancy and representation.

The confirmed or assumed primary host fish species for both the Louisiana Pigtoe and Texas Heelsplitter are known to be common and widespread throughout the range of both mussel species, and are therefore not believed to be a limiting factor to dispersal at this time (Nico and Sturtevant 2022, entire; Nico et al. 2022, entire; Nico and Fuller 2022, entire; Fuller et al. 2022, entire). Each of the identified fish hosts are known to tolerate lacustrine environments and may utilize impoundments as corridors to facilitate migration between hydrologically connected tributaries, thus aiding mussel dispersal. If fish host species are indeed abundant, existing dams and the construction new major dams and reservoirs, and other barriers to fish movement are the primary mechanism in which remaining populations are isolated. Furthermore, reservoir impacts to river ecosystems can be difficult and costly to manage or minimize. For instance, it is possible to manage dam releases to more closely mimic natural fluctuations in flows to benefit wildlife; however, most reservoirs function primarily to provide water supply and/or flood control, and meeting those objectives typically involves holding on to as much water as possible (i.e., not releasing); this limits the ability of reservoir managers to modify releases for the purpose of meeting wildlife conservation or recovery goals. Although dams have been managed to allow fish passage for spawning, to our knowledge, fish passage has not been facilitated specifically to allow movement of host fish for the benefit of freshwater mussels, nor would this be cost-effective considering host fish for Louisiana Pigtoe and Texas Heelsplitter are believed to be abundant. Nevertheless, reservoirs represent a permanent barrier to freshwater mussel dispersal. The overall impact of reservoirs is believed to be greater for the Louisiana Pigtoe relative to the Texas Heelsplitter, which is able to persist in reservoir conditions although questions remain about their reproductive success in lacustrine environments.

5.E. DIRECT MORTALITY

Direct mortality includes any activity or event, whether human induced or natural, that results in the death of mussels within a localized area due to removal, crushing, burying, consumption, desiccation, or poisoning. Potential activities or events causing direct mortality include, but are not limited to, development projects (such as bridge replacement, stream channelization, and impoundment construction), undeveloped low-water crossings with vehicular traffic that intersect mussel beds, bank collapse, accidental release of hazardous materials, predation, vandalism, and collection (whether for scientific purposes, recreation, or by collectors). Although we expound on only a subset of possible activities and events that may cause direct mortality in this report, the above activities, and others not mentioned, are presumed to occur with some regularity in most watersheds occupied by Louisiana Pigtoe and Texas Heelsplitter and impact populations from time to time. The frequency, intensity, and magnitude of these impacts likely vary in time and by location, and are difficult to quantify with any certainty other than to acknowledge that they exist and negatively affect mussel survival to some degree.

In addition to these anthropogenic activities causing direct mortality, predation on freshwater mussels is a natural ecological interaction. Raccoons, muskrats, snapping turtles, and fish are known to prey upon mussels (East et al. 2013, p. 692; Walters and Ford 2013, p. 480; Neves and Odom 1989, p. 939). Under natural conditions, the level of predation occurring within these mussel species populations is not likely to pose a significant risk to any given population. However, during periods of low flow, terrestrial predators have increased access to portions of the river that are otherwise too deep under normal flow conditions, resulting in unnaturally high levels of predation that can decimate mussel populations. Predation during drought has been observed for the Texas Heelsplitter on the Sabine River (Walters and Ford 2013, p. 479). Drought and low flow conditions are predicted to occur more often and for longer periods due to the effects of future climate change; therefore, the tributaries and upper portions of focal areas for Louisiana Pigtoe and Texas Heelsplitter are expected to experience additional predation pressure into the future. Increased predation pressure may become especially problematic during summer months due to projected reductions in summer minimum base flows (Lafontaine et al. 2019, entire). Predation is expected to be less of a problem for the lower portions of the main stem river populations where the rivers are significantly larger than the tributary streams and these species are less likely to be located in exposed or very shallow habitats.

Additionally, certain mussel beds within some populations, due to ease of access, are vulnerable to over-collection and vandalism. These areas have well known and well documented mussel beds that are often sampled multiple times annually by various researchers for various scientific projects. Populations subjected to repeated sampling or monitoring may experience increased stress or higher rates of mortality. Mortality may also occur in areas with intense recreation where local fishing enthusiasts have been observed using freshwater mussels as bait. The risk of direct mortality from recreation or over collection for scientific purposes are compounded by the additional stressors discussed in this chapter, which can influence mussel survival in a cumulative manner. Service biologists recently hosted a meeting with State biologists, consultants, and academia who are involved in mussel research to discuss ongoing monitoring and scientific collections and to reduce the likelihood of over harvesting mussels from any given population (USFWS 2018, p.1). We anticipate this collaboration among researchers will continue into the future with ongoing coordination and annual meetings.

5.F. INVASIVE SPECIES

Invasive species, such as Asian Clam (*Corbicula fluminea*), Zebra Mussel (*Dreissena polymorpha*), feral hog (*Sus scrofa*), floating water hyacinth (*Eichhornia crassipes*), giant salvinia (*Salvinia molesta*), and hydrilla (*Hydrilla verticillata*), occur throughout the range of Louisiana Pigtoe and Texas Heelsplitter and can negatively impact mussel survival. These impacts include predation (feral hog), habitat destruction or modification (feral hog, floating water hyacinth, giant salvinia, hydrilla), changes to water quality (feral hog, Zebra Mussel), increased resource competition (Asian Clam, Zebra Mussel), or physical impairment (Zebra Mussel, hydrilla) (Howells 2010a, p. 13; Howells 2010b, pp. 14-15; Kaller and Kelso 2007, pp. 172-174).

Zebra Mussels have attached to this young Higgins Eye Pearlymussel, an endangered species found in the Mississippi river. Photo by USFWS

Asian Clam are common in river basins across the range of Louisiana Pigtoe and Texas Heelsplitter, often at high densities, and likely compete with native unionids for food, oxygen, physical space, and other environmental resources (USGS 2019a, entire; Howells 2010a, p. 13; Howells 2010b, p. 14; Cherry et al. 2005, p. 369). However, they are sensitive to low flow, increased silt loads, temperature extremes, and low dissolved oxygen, and can experience rapid die-offs (Cherry et al. 2005, p. 369). Tissue decomposition associated with Asian Clam die-offs can cause spikes of ammonia in the water column and impact native mussels, especially during early life stages (Cherry et al. 2005, pp. 376, 378); Cooper et al. (2005, p. 392) concluded concentrations of ammonia in substrate pore water (i.e., water contained in the interstitial spaces located between particles comprising the substrate) can be greater than that of the water column during Asian Clam die-offs, especially under low flow conditions, potentially impacting glochidia survival.

Although Zebra Mussel infestations occur in several Texas reservoirs, including Lewisville Lake and Lake Livingston, populations have not become established in nearby river habitats occupied by Louisiana Pigtoe and Texas Heelsplitter (TPWD 2019, entire; USGS 2019e, entire; Ford et al. 2016, p. 47). The distribution of Zebra Mussels may be limited to lacustrine environments in part due to the fragility of Zebra Mussel veligers (larval stage) and the higher turbulence and velocities associated with reservoir discharge (Churchill and Quigley 2018, p. 1123). Where native mussels and Zebra Mussels co-occur, Zebra Mussels compete with native mussels for dissolved oxygen and food resources, although the extent to which this competition limits the growth or survival of native mussels is poorly understood. Zebra Mussels reproduce prolifically and attach to virtually any surface, including the shells of native mussels, which impedes mobility and further reduces resource uptake (Baker and Levinton 2003, p. 98). Native mussels and Zebra Mussels prefer the unicellular cyanobacteria *Microcystis* as a food source; however, native mussels are less efficient at selecting *Microcystis* over less nutritious detritus particles than Zebra Mussels. Therefore, where Zebra Mussels are present, food quality available to native mussels decreases, contributing to native mussel mortality (Baker and Levinton 2003, pp. 103-104).

Feral hogs occur throughout the range of both mussel species and are known to engage in a variety of activities that disturb soils and degrade water quality, including the contribution of waste (i.e., excrement) that elevates nutrient and fecal coliform levels within streams and rivers (USDA 2019, entire; Gregory et al. 2014, p. 35; Kaller et al. 2007, p. 173). Feral hogs may also consume native mussels in shallow waters (Kaller et al. 2007, p. 174). Bank and stream bed damage from feral hogs contributes to erosion and increased sedimentation, and their presence appears to cause native mussel diversity and abundance to decrease through organic enrichment of the water column and unfavorable changes to microbial community composition (Howells 2010b, p. 10; Kaller et al. 2007, p. 174).

Invasive macrophyte infestations of floating water hyacinth, hydrilla, and giant salvinia negatively impact native mussels and their host fish throughout the southern half of the ranges of Louisiana Pigtoe and Texas Heelsplitter by creating hypoxic conditions through respiration and during decay (USGS 2019b, entire; USGS 2019c, entire; USGS 2019d, entire; Karateyev and Burlakova 2007, p. 298). Dense mats of hydrilla, an aquatic plant rooted to substrate, can also impede native mussel movement during periods of fluctuating surface water levels, leaving them stranded as water levels recede. In Texas, attempts to control these exotic species has led to periodic partial drawdowns of B.A. Steinhagen Lake, a reservoir known to be occupied by Texas Heelsplitter (Howells 2010b, p. 14), which likely led to mussel mortalities in areas where substrates were exposed for extended periods.

5.G. CLIMATE CHANGE

Experts agree climate change has been underway for decades with mounting impacts to humans, wildlife, infrastructure, and communities, particularly in coastal areas; continued greenhouse gas emissions at or above current rates will cause further warming with broad implications for living organisms across the planet and the habitat on which they depend (Intergovernmental Panel on Climate Change (IPCC) 2013, pp. 11-12, IPCC 2021, pp. 1-13—1-15). Warming in Texas is expected to be greatest in the summer (Maloney et al. 2014, p. 2236, Fig. 3), with the number of extremely hot days (high temperatures exceeding 95º Fahrenheit) projected to double by around 2050 (Kinniburgh et al. 2015, p. 83). The effects of climate change are expected to be more pronounced in the naturally dry climates of west Texas (Diffenbaugh et al. 2008, p. 3), although impacts to water resources are projected throughout the state. Changes in stream temperatures are expected to reflect changes in air temperature, at a rate of approximately 0.6 – 0.8°C increase in stream water temperature for every 1°C increase in air temperature (Morrill et al. 2005, pp. 1-2, 15), with implications for temperature-dependent water quality parameters such as dissolved oxygen and ammonia toxicity. Given that freshwater mussels in Texas exist at or near the ecophysiogical edge of climate and habitat gradients of unionid biogeography in North America, they may be particularly vulnerable to future climate changes in combination with current and future stressors (Burlakova et al. 2011a, pp. 156, 161, 163; Burlakova et al. 2011b, pp. 395, 403).

While projected changes to rainfall in Texas may seem relatively small (USGCRP 2017, p. 217), higher temperatures caused by anthropogenic activity will lead to increased soil water deficits because of higher rates of evapotranspiration. In turn, higher evapotranspiration rates will likely result in increasing drought severity in future climate scenarios at a time when “extreme precipitation, one of the controlling factors in flood statistics, is observed to have generally increased and is projected to continue to do so across the United States in a warming atmosphere” (USGCRP 2017, p. 231). Even if precipitation and groundwater recharge remain at current levels, increased groundwater pumping and resulting aquifer shortages due to increased temperatures are nearly certain (Loaiciga et al. 2000, p. 193; Mace and Wade 2008, pp. 662, 664-665; Taylor et al. 2013, p. 3).

Higher temperatures are also expected to lead to increased evaporative losses from reservoirs, diminishing overall water supply and negatively affecting downstream releases and flows (Friedrich et al. 2018, p. 167). Effects of climate change, such as changes to seasonal rainfall patterns, air temperature increases, and increases in drought frequency and intensity, have been shown to be occurring throughout the range of Louisiana Pigtoe and Texas Heelsplitter (USGCRP 2017, p. 188; Andreadis and Lettenmaier 2006, p. 3); these effects are expected to exacerbate several of the stressors discussed above, such as water temperature and flow loss (Wuebbles et al. 2013, p. 16). A recent review of future climate projections for Texas concludes that both droughts and floods could become more common in east Texas, with droughts like 2011 (the warmest on record) becoming commonplace by the year 2100 (Mullens and McPherson 2017, pp. 3, 6). This trend of more frequent droughts is driven by increases in hot temperatures (e.g., daily maximum) and the number of days projected to be at or above 100°F, which is set to “increase in both consecutive events and the total number of days" (Mullens and McPherson 2017, p. 14-15). Similarly, floods and extreme runoff are projected to become more common and severe in the 21st century as the frequency, magnitude and intensity of heavy precipitation events increase (Mullens and McPherson 2017, p. 20, USGCRP 2017, p. 224).

In the analysis of the future condition for Louisiana Pigtoe and Texas Heelsplitter, in Chapter 6, climate change is considered further under various plausible future scenarios, serving to exacerbate already deteriorating conditions through an increase of fine sediments, changes to water quality, loss of flowing water, and predation, among others.

5.H. SUMMARY

Our analysis of the past, current, and future variables that influence Louisiana Pigtoe and Texas Heelsplitter needs for long-term viability revealed that there are four factors that pose the largest risk to future viability, namely degradation of water quality, altered hydrology, substrate changes, and habitat fragmentation; all of which are exacerbated by climate change.

All the factors affecting viability, including degradation of water quality, altered hydrology, changes to substrate, habitat fragmentation, direct mortality, and invasive species, are carried forward in Chapter 6 where we assess the future condition of Louisiana Pigtoe and Texas Heelsplitter populations and the viability of each species as the influence of each factor changes into the foreseeable future.

Chapter 6 – Species Viability in the Future

This report has considered what Louisiana Pigtoe and Texas Heelsplitter need for viability and the current condition of those needs (Chapters 2, 3 and 4), and reviewed the risk factors that are driving the historical, current, and future conditions of the species (Chapter 5 and Appendix B). In this Chapter we consider potential changes to risk factors in the foreseeable future, and the implications of those changes on the viability of each species. In keeping with the SSA framework, we will apply our forecasts using the concepts of species resiliency, redundancy, and representation to describe future viability of Louisiana Pigtoe and Texas Heelsplitter.

6.A. Introduction

Relative to historical conditions (i.e., historical range), Louisiana Pigtoe and Texas Heelsplitter have declined significantly in terms of overall distribution and abundance over the past 100 or more years. Most known populations are isolated and currently exist in very low numbers (i.e., low abundance), have limited evidence of recruitment, and are believed to occupy much less habitat than in the past (range contraction). Furthermore, existing available habitats are experiencing additional stressors and are reduced in terms of water quality and quantity relative to historical conditions.

Efforts to create new infrastructure for flood control and water supply continued throughout the mid-20th century, and by 1975 major dams and reservoirs had been constructed in every river basin occupied by Louisiana Pigtoe and Texas Heelsplitter; in some cases, multiple reservoirs were established along the same river. Only the upper most reaches of a few rivers were spared, including the Calcasieu River population of Louisiana Pigtoe, which is currently free of large upstream impoundments. The inundation and subsequent alteration of hydrology and sediment dynamics associated with the operation of these flood-control, hydropower, and municipal supply reservoirs has resulted in irreversible changes to the natural flow regime of these rivers and ultimately re-shaped the aquatic ecosystems they provide, including the fisheries and invertebrate communities that depend on them, as well as populations of Louisiana Pigtoe and Texas Heelsplitter.

With the advent of the industrial revolution and before Congress enacted laws like the Clean Water Act to protect the environment, adverse water quality impacts were common in many rivers within the range of these two freshwater mussel species. Prior to the implementation of modern sanitation, impacts could be severe, leading the Texas Department of Health to call the Trinity a "mythological river of death" in 1925 (USGS 1998, p. 19). Fortunately today, water quality has improved dramatically utilizing enhanced treatment technology and centralized wastewater treatment, and fish populations have rebounded, although not to historical levels (Perkin and Bonner 2016, p. 97). Nevertheless, water quality in many watersheds remains largely altered from pre-industrial revolution condition, and degradation continues to affect mussels and their habitats. These impacts become more pronounced during low flow conditions, when water chemistry and geomorphological constraints diminish instream habitats. The timing, frequency, and intensity of high flow events have also been altered, generating greater shear stress that mobilizes substrates, scours mussel beds, and erodes river banks.

Additionally, while host fish may still be adequately represented in contemporary fish assemblages, access to fish hosts can be reduced during critical reproductive times by barriers such as low-water crossings, reservoirs, and low-head dams that are relatively common on the landscape. Low flows can lead to dewatering of habitats, desiccation of individuals, elevated water temperatures (above 30°C and approaching 40°C) and other water quality degradations (low dissolved oxygen and elevated TAN), as well as increased exposure to predation. Diminished access to host fish leads to reduced reproductive success just as barriers to fish passage impede the movement of fish, and thus compromise the ability of mussels to disperse and colonize new habitats following a disturbance (Schwalb et al. 2013, p. 446). Lastly, freshwater mussels have long been utilized by humans, for food and bait, for pearls and buttons, for scientific collection, and to create artificial pearls; even today rare mussels are vulnerable to human collection (Bogan 1993, pp. 604-5), although other threats like habitat modification pose a greater risk.

Populations of Louisiana Pigtoe and Texas Heelsplitter are faced with a myriad of stressors from natural and anthropogenic sources that pose a risk to their survival in both large and small river segments. In Texas, as elsewhere, climate change has the noteworthy distinction of being able to directly or indirectly exacerbate the most relevant stressors to freshwater mussels wherever they occur. Climate projections suggest persistent droughts over the continental United States that are longer, cover more area, and are more intense than what has been experienced in the 20th century (APA 2019, pg. 4; Terando et al. 2018, p. 786; Wehner et al. 2017, p. 237). Humans are likely to respond to climate change in predictable ways to meet their needs, such as increased groundwater pumping and surface water diversions, and increased use of reverse osmosis to treat sources of water that are of poor quality (thereby generating increasing volumes of reject wastewater). These activities will increase overall demand for freshwater resources at a time when those very resources are strained and less abundant (reviewed in Banner et al. 2010, entire). We expect climate change impacts to occur throughout the range of both Louisiana Pigtoe and Texas Heelsplitter.

These risks, acting alone or in combination with each other and climate change, could result in the extirpation of additional mussel populations, further reducing the overall redundancy and representation of Louisiana Pigtoe and Texas Heelsplitter. Historically, each species, bolstered by large, interconnected populations (i.e., with meta-population dynamics), would have been more resilient to stochastic events such as drought, excessive sedimentation, and scouring floods. As locations became extirpated by catastrophic events, they could be recolonized over time by dispersal from nearby surviving populations, facilitated by movements of "affiliate species" of host fish (Douda et al. 2012, p. 536). This connectivity across potential habitats made for highly resilient species overall, as evidenced by the long and successful evolutionary history of freshwater mussels as a taxonomic group, and in North America in particular. However, under current conditions, restoration of that connectivity on a regional scale is not feasible. As a consequence of these current conditions, the viability of Louisiana Pigtoe and Texas Heelsplitter now primarily depends on maintaining the remaining isolated populations and potentially restoring new populations where feasible.

6.B. FUTURE SCENARIOS AND CONSIDERATIONS

Because of significant uncertainty regarding the location, magnitude, and duration of impacts related to flow loss, water quality degradation, extreme flooding and scour/substrate mobilizing events, or new impoundment construction, we began forecasting future viability for Louisiana Pigtoe and Texas Heelsplitter in terms of resiliency, redundancy, and representation under three plausible future scenarios (maintain current trends, moderate increase in stressors, and severe increase in stressors). However, during our evaluations it became apparent that our approach lacked the resolution to distinguish any meaningful difference between the "maintain current trends" and the "moderate increase in stressors" scenarios. As a result, the SSA team decided to limit the future forecasts analyzed in this report to two scenarios, a moderate increase in stressors (Scenario 1) and a severe increase in stressors (Scenario 2)(Table 6.1). Both scenarios were evaluated at three time intervals into the future, where future risks were considered to determine the biological status of mussel populations and their habitats in 10, 25, and 50 years. Ten years represents one to two generations of mussels, assuming an average reproductive life span of five to 10 years. Twenty-five years similarly represents at least two to four mussel generations and 50 years represents at least five or more generations of mussels.

Table 6.1. Two future scenarios (moderate and severe increase in stressors) evaluated under associated lower and higher climate change emission scenarios (i.e., 4.5 and 8.5 Representative Concentration Pathways (RCP*), respectively), at each of three time steps.

Future Scenario	**RCP***	**10–years**	**25-years**	**50-years**
Scenario 1: moderate increase in stressors	4.5	0–10 yrs	10–25 yrs	25–50 yrs
Scenario 2: severe increase in stressors	8.5	0–10 yrs	10–25 yrs	25–50 yrs
*RCP = Representative Concentration Pathway Scenario (IPCC 2014, pp. 9, 57)				

The future scenarios included the interactive effects of future climate change through the use of the RCP 4.5 (lower greenhouse gas emissions trajectory) and RCP 8.5 (higher greenhouse gas emissions trajectory) scenarios contributed by the Working Group III to the Fifth Assessment Report and described in the most recent Synthesis Report of the Intergovernmental Panel on Climate Change (IPCC 2014, pp. 9, 22, 57). The IPCC Report describes four pathways that are representative of alternate trajectories of greenhouse gas emissions and the resulting atmospheric concentrations (RCPs) from the year 2000 to 2100 (van Vuuren et al. 2011, p.5). Scenario 1 assumed RCP 4.5, a medium stabilization scenario where CO_2 emissions continue to increase through mid-21st century, but then decline and atmospheric carbon dioxide concentrations are between 580 and 720 ppm CO_2 from 2050 to 2100, representing an approximate +2.5 ºC temperature change relative to 1861-80 (IPCC 2014, p. 9, Figure SPM.5). Scenario 2 assumed RCP 8.5 where atmospheric carbon dioxide concentrations are above 1000 ppm CO_2 between 2050 and 2100, representing an approximate +4.5 ºC temperature change relative to 1861-80 (IPCC 2014, p. 9, Figure SPM.5). The most recent IPCC Synthesis Report projects global temperature change to 2100 and beyond (IPCC 2014, p. 8). A recent study suggests that, because of uncertainty in long-run economic growth rates, there is "a greater than 35% probability that emissions concentrations will exceed those assumed in the most severe of the available climate change scenarios (RCP8.5)" by 2100 (Christensen et al. 2018, p. 1).

This SSA is based on the following assumptions, which are from the most recent Synthesis Report of the IPCC (IPCC 2014, entire) and other scientific studies. The IPCC Synthesis Report considers RCP 4.5 as an intermediate scenario and RCP 8.5 as having "very high" greenhouse gas emissions (IPCC 2014, p. 8). Under RCP 4.5, current conditions, including a continued trend towards increased warming, frequency and severity of extreme weather events, such as droughts and floods, are expected to continue. Global mean surface temperature change is projected "*more likely than not*" to exceed 1.5 ºC by 2100, relative to 1850-1900 (IPCC 2014, p. 60). Under RCP 8.5, future conditions include a more dramatic increasing trend with more significant increases in the frequency and severity of extreme weather events, such as droughts and floods, under future climate projections. Global mean surface temperature change is projected "*likely*" to exceed 2.0 ºC by 2100, perhaps as high as 4.8 ºC, relative to 1850-1900 (IPCC 2014, p. 60). It is important to remember that two of the most powerful environmental forces that influence the presence of living organisms in any given area are temperature and the presence of water; therefore, even minor shifts in global temperatures can have dramatic effects on species distribution and abundance. Because of the influence of temperature on water, including evapotranspiration, climate change is expected to result in drier soils with less runoff and under RCP 8.5 by 2100, "no region of the planet is projected to experience significantly higher levels of annual average surface soil moisture…even though much higher precipitation is projected in some regions" (USGCRP 2017, pp. 232-8).

For all IPCC RCP scenarios, extreme precipitation events over most mid-latitude land masses (like North America) will very likely become more intense and frequent as global mean surface temperatures increase (IPCC 2014, p. 60) and, as such, future temperature and precipitation patterns are likely to become more variable and extreme, with drought and flooding events occurring more frequently and with higher severity in the southwestern United States (Seager et al. 2007, pp. 1183-4). In the southeastern United States, most rivers are projected to experience lower annual minimum 7-day base flows and summer minimum base flows with fewer high flow events of longer duration (Lafontaine et al. 2019, entire). The magnitude of these changes is expected to increase with time even without increasing greenhouse gas emissions as even steady-state (i.e., no change in greenhouse emissions) or slightly reduced emissions would produce increased atmospheric concentrations. Given the inertia of the climate system and regardless of future emissions, the risk of flooding is expected to increase over the next 25-50 years. These increases in the severity of extreme floods are expected to affect human systems (reviewed in Willner et al. 2018, entire; Hirabayashi et al. 2013, entire), as well as marine and freshwater ecosystems and the aquatic organisms that depend on them, including freshwater mussels and their host fishes.

Future human demand for water resources, due to projected human population growth and limitations of existing supplies, is expected to increase and interact with climate effects to exacerbate the effects of drought on surface water resources in Texas. These effects are expected to occur throughout the range of Louisiana Pigtoe and Texas Heelsplitter, and are likely to impact the ability of water managers to provide "environmental flows" that are designed to provide the minimum flow needed by freshwater mussels and other aquatic dependent organisms (Wolaver et al. 2014, pp. 1-2).

The upper portions of the basins, including tributaries, will be more sensitive to changes in precipitation patterns and withdrawals, relative to the lower portions of the basins, where flows are generally larger and are supplemented by municipal wastewater (or other) return flows; senior water rights located at the "bottom" of the basin also help protect flows in the lower reaches. However, while minimum flows may be maintained, other artifacts of altered hydrology may have deleterious effects to mussels and their habitats through altered water quality. Changes to sediment transport (more extreme deposition and scour) will also lead to reductions in habitat quality and quantity.

This SSA report evaluates two plausible future scenarios (Table 6.1). Scenario 1 considers a moderate increase in stressors resulting in a moderate decline of current conditions projected across the next 10, 25, and 50 years. Scenario 1 is based on the RCP 4.5 emissions trajectory and associated model projections, and represents medium-term increases in emissions followed by a decline through the rest of the century. The resulting climate impacts are greater than today but less than under Scenario 2, and indicate an overall moderate decline in current Louisiana Pigtoe and Texas Heelsplitter population trends. Scenario 2 projects a severe decline in current Louisiana Pigtoe and Texas Heelsplitter population trends and condition categories in the future under RCP 8.5 predictions. Further, Scenario 2 also includes anthropogenic actions, such as the construction of new reservoirs, wastewater treatment plants, and other currently proposed projects. Scenario 2 manifests as a future where the hydrological conditions of many of the rivers and streams currently occupied by Louisiana Pigtoe and Texas Heelsplitter are altered such that base flows are diminished, floods are more severe if not more frequent, and mussels and their habitats are adversely affected through degradation of water quality and quantity. These altered hydrological conditions are primarily caused by a combination of increasing anthropogenic stressors and climate change.

We examined the resiliency, representation, and redundancy of Louisiana Pigtoe and Texas Heelsplitter under two plausible future scenarios for each of the three time periods. The resiliency of mussel populations depends on future conditions providing water of sufficient quality and quantity to meet the life history needs of Louisiana Pigtoe and Texas Heelsplitter and their host fishes. Resiliency requires good water quality, flowing water, and suitable substrates because these habitat factors directly influence species reproduction and abundance, which determines the amount of occupied habitat. We expect the extant populations of these mussel species to experience changes to critical aspects of their habitat in different ways under the different scenarios. We projected the future resiliency of each population based on events that were likely to occur under each scenario. We then projected the overall condition for each population based on expert opinion and anticipated changes to habitat and population factors. For these projections, populations in high (healthy) condition are expected to have high resiliency at that time period (i.e., they occupy habitat of sufficient size to allow for ebbs and flows in density of mussel beds within the population over time without significantly impacting the overall health of the population). Populations in high condition are expected to persist into the future (> 90 % chance of persistence beyond 20 years), and they have the ability to withstand stochastic events that may occur. Populations in moderate (moderately healthy) condition have lower resiliency than those in high condition, but the majority (60–90 %) are expected to persist beyond 20 years. Populations in moderate condition are smaller and less dense than those in high condition. Populations in low (unhealthy) condition have low resiliency and are not necessarily able to withstand stochastic events. As a result, they are less likely to persist beyond 20 years (10–60 % chance). Finally, we considered populations functionally extirpated/extirpated when they either lacked individuals (i.e., surveys yielded no observations) or there was no evidence of reproduction (functionally extinct); these populations have very low resiliency and have less than a 10 % chance of persistence beyond 20 years.

In an effort to maintain consistency throughout the scenario evaluation process for each Louisiana Pigtoe and Texas Heelsplitter population, the SSA team developed a population resiliency model to determine the direction and magnitude of change to population resiliency under each future scenario and time step. This unweighted additive model, based on the effects pathway flowchart, shows how threats under the different scenarios influence habitat factors (habitat structure/substrate, hydrological regime, and water quality), population factors (occupied habitat reach length, abundance, and reproduction/recruitment), host fish availability, and survival (Figure 6.1). However, if two of the three habitat factors (habitat structure/substrate, hydrological regime, or water quality) were determined to be in severe decline, we considered population resiliency to also be in severe decline regardless of model output. The final output value represented the impact of all forecasted threats to population resiliency.

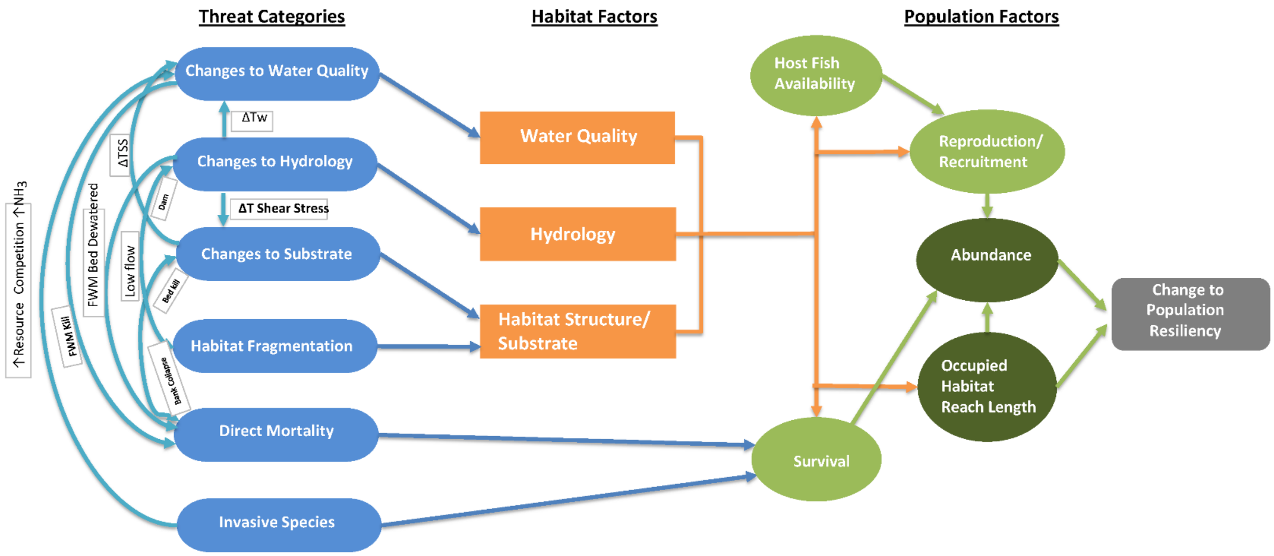


Figure 6.1. *Effects pathway flowchart for Louisiana Pigtoe and Texas Heelsplitter. Threats (blue elliptical circles on the left side of the chart) influence Habitat Factors (orange boxes in middle) and Population Factors (green circles at right), which ultimately determines population resiliency (grey box at far right). White boxes (on far left) provide examples of how a change to one threat category can influence other threat categories.*

Inputs to the population resiliency model were determined by SSA team consensus on the projected magnitude of change to the six threat categories (water quality, hydrology, habitat structure/substrate, fragmentation, direct mortality, and invasive species) and classified as either significant improvement, moderate improvement, maintain current trend, moderate decline, or severe decline (see Appendix C, tables C.1 and C.2 for classification criteria). Each threat category projection was then assigned a numerical value corresponding to the previous classifications. Input values ranged from 2 to -2: where 2 represents significant improvement; 1, moderate improvement; 0, maintain current trend; -1, moderate decline; and -2, severe decline. The algorithm for the population resiliency model was expressed as follows:

$$\mathbf{\triangle Resilience = 5(\triangle wq + \triangle hr + \triangle s + \triangle f) + \triangle m + \triangle i}$$

Where: $\triangle$Resilience = change to population resilience
$\triangle$wq = threat of changes to water quality
$\triangle$hr = threat of changes to hydrological regime
$\triangle$s = threat of changes to substrate
$\triangle$f = threat of changes to habitat fragmentation
$\triangle$m = threat of changes to direct mortality
$\triangle$i = threat of changes to invasive species

Population Resiliency Model assumptions:

- All threat categories are equal in importance (unweighted); however, those threats (or their products) used more frequently in the algorithm have more influence on model output than those used less.
- Each threat category can influence one or many other threat categories (see fig. 6.1).
- Current condition was considered to follow a continuing declining trend and additional conservation, if implemented, would at best negate the current decline in future scenarios.

Model output values ranged from 44 to -45, with positive numbers indicative of an overall improvement in population resiliency, 0 indicating no change from current trend, and negative values showing an overall decline in population resiliency. Scenarios with two of the three habitat factors (water quality, hydrology, and substrate) projected to be in severe decline from the current trend were considered to result in a severe decline in population resilience and identified with an output value of -45. Output values were categorized as shown in Table 6.2.

Table 6.2. Population resiliency model output classifications

Model output	Classification
44 ≥ △Resiliency > 22	Significant improvement in population resiliency
22 ≥ △Resiliency > 0	Moderate improvement in population resiliency
△Resiliency= 0	Maintain current population resiliency
0 > △Resiliency ≥ (-22)	Moderate decline in population resiliency
(-22) > △Resiliency > (-44)	Severe decline in population resiliency

Note: △Resiliency = (-45) indicates two of the three habitat factors are severely declining; therefore, △Resiliency = severe decline.

For each future scenario and time step, the population resiliency model output was compared to the population's current condition, as described in Chapter 4. SSA team consensus was then used to evaluate the effect of the projected change in population resilience over time to the current population condition, resulting in a projected population condition for each future scenario and time step.

6.B.1. Future Scenario 1 – Moderate Increase in Stressors

Scenario 1 considers a future with a moderate increase in stressors where conditions moderately decline from present trends under current population conditions. Scenario 1 assumes intermediate climate effects, including more frequent and intense droughts, where droughts are broken by major flooding. Scenario 1 also considers additional groundwater and surface water demands associated with human population growth and decreased water availability that is compounded by intermediate climate effects. Reductions in streamflow, due to decreased inputs and enhanced evapotranspiration, are expected to occur in all streams and rivers, and those effects will likely be more pronounced in the upper basins.

Scenario 1 considers additional water projects, like new wastewater treatment plant outfalls or proposed new reservoirs, only if currently proposed or planned. Under Scenario 1, proposed new reservoirs are constructed in the next 10–25 years, and any effects from completion of the associated dams are manifest in the next 25–50 years. Necessary routine maintenance as well as repair and replacement of existing old dams occurs in the next 10–25 years, and any effects from those repairs are manifest in the next 25–50 years.

6.B.2. Future Scenario 2 – Severe Increase in Stressors

Scenario 2 considers a future with a severe increase in stressors where conditions severely decline from the status quo (i.e., current conditions). Scenario 2 considers severe climate effects, including more frequent and intense droughts, where droughts are broken by major flooding. Scenario 2 considers additional groundwater and surface water demands associated with increased human demand and decreased water availability due to severe climate effects. Scenario 2 considers additional water projects, like new wastewater treatment plant outfalls, even if not currently proposed, as well as possible new reservoirs and other construction projects affecting water quality or quantity.

6.C. Future Viability (Resiliency, Redundancy, and Representation)

This section generally reviews the viability of Louisiana Pigtoe and Texas Heelsplitter under each of the two scenarios. The output of the scenarios at each time step (10-years, 25-years, and 50-years into the future) for each species, as well as a synopsis of the projected effects to the populations over time are included in Appendix C.

6.C.1. Future Scenario 1 – Moderate Increase in Stressors

Resiliency

Under Scenario 1, populations of Louisiana Pigtoe and Texas Heelsplitter decline in resiliency over time as the factors that are having an influence on populations moderately decline from the present trajectory of the estimated current condition (Table 6.4). The effects of current levels of climate change continue to result in low streamflows, which lead to increased sedimentation, reduced water quality, and occasional desiccation. Population extirpations occur to both species, with only the Cossatot River population of Louisiana Pigtoe in moderate condition in 50 years. The remaining populations of both species are in low condition and are therefore particularly vulnerable to extirpation.

Table 6.4. Condition of Louisiana Pigtoe and Texas Heelsplitter populations under Future Scenario 1 (Moderate Increase in Stressors). FE/E = Functionally Extirpated/Extirpated.

SPECIES	Representation Areas (River Basin)	POPULATIONS (Focal Areas)	Current Condition	10-yrs	25-yrs	50-yrs
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	FE/E	FE/E	FE/E	FE/E
Texas Heelsplitter	Neches	Neches R/BA Steinhagen	Low	Low	Low	Low
Texas Heelsplitter	Neches	Lower Neches R	Low	Low	Low	FE/E
Texas Heelsplitter	Trinity	Grapevine LK	FE/E	FE/E	FE/E	FE/E
Texas Heelsplitter	Trinity	Trinity R/Livingston	Low	Low	Low	FE/E
Louisiana Pigtoe	Red	Little R/Rolling FK	Moderate	Moderate	Low	Low
Louisiana Pigtoe	Red	Cossatot R	High	High	High	Moderate
Louisiana Pigtoe	Red	Saline R (Little)	Moderate	Moderate	Moderate	Low
Louisiana Pigtoe	Red	Lower Little R	FE/E	FE/E	FE/E	FE/E
Louisiana Pigtoe	Big Cypress	Big Cypress Bayou	Moderate	Moderate	Moderate	Low
Louisiana Pigtoe	Calcasieu	Upper Calcasieu R	Low	Low	Low	FE/E
Louisiana Pigtoe	Pearl	Pearl R	Low	Low	Low	Low
Louisiana Pigtoe	Sabine	Sabine R	FE/E	FE/E	FE/E	FE/E
Louisiana Pigtoe	Sabine	Bayou Anacoco	Moderate	Low	Moderate	Low
Louisiana Pigtoe	Neches	Angelina R	Low	Low	Low	FE/E
Louisiana Pigtoe	Neches	Neches R	High	High	Low	Low
Louisiana Pigtoe	Neches	Lower Neches R	Low	Low	Low	Low
Louisiana Pigtoe	San Jacinto	E FK San Jacinto R	Low	Low	Low	FE/E

Redundancy

Both Louisiana Pigtoe and Texas Heelsplitter lose redundancy under Scenario 1 (Tables 6.4 and 6.5). Under our projections, the Louisiana Pigtoe would have one population in moderate condition, seven in low condition, and five functionally extirpated or extirpated populations across five representation areas in 50 years. Of the five populations evaluated for Texas Heelsplitter, all but one (Neches River/B.A. Steinhagen population) are projected to become extirpated or functionally extirpated in 50 years under this scenario.

Representation

Under Scenario 1, both Louisiana Pigtoe and Texas Heelsplitter lose two areas of representation, diminishing the overall adaptive capacity of each species to future environmental change in the next 50 years (Tables 6.4 and 6.5). The Louisiana Pigtoe would lose the Upper Calcasieu River and San Jacinto River populations, and the Texas Heelsplitter would lose the Sabine River and Trinity River populations.

Table 6.5. Summary of condition for Louisiana Pigtoe and Texas Heelsplitter populations under Future Scenario 1 (Moderate Increase in Stressors).

Projected condition	Number of Louisiana Pigtoe populations (n=13 within 7 representation areas)			Number of Texas Heelsplitter populations (n=5 within 3 representation areas)		
	10-year	25-year	50-year	10-year	25-year	50-year
High	2	1	0	0	0	0
Moderate	3	3	1	0	0	0
Low	6	7	7	3	3	1
Extirpated/ functionally extirpated	2	2	5	2	2	4
Number of representation areas	7	7	5	2	2	1

6.C.2. Future Scenario 2 – Severe Increase in Stressors

Resiliency

Under Scenario 2, populations of Louisiana Pigtoe and Texas Heelsplitter would decline in resiliency over time as the effects of severe climate change begin to impact populations (Table 6.6). The effects of severe climate change result in even lower stream flows, with a proportionally severe increase in sedimentation, reduction in water quality, and increase in potential for desiccation of habitat. All Texas Heelsplitter populations are projected to become extirpated or remain functionally extirpated in 50 years. A total of seven populations of Louisiana Pigtoe are expected to remain functionally extirpated or become extirpated in 50 years, with the remaining six populations in low condition. The populations that remain in low condition are particularly vulnerable to extirpation.

Table 6.6. Condition of Louisiana Pigtoe and Texas Heelsplitter populations under Future Scenario 2 (Severe Increase in Stressors). FE/E = Functionally Extirpated/Extirpated.

SPECIES	Representation Areas (River Basin)	POPULATIONS (Focal Areas)	Current Condition	10-yrs	25-yrs	50-yrs
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	FE/E	FE/E	FE/E	FE/E
Texas Heelsplitter	Neches	Neches R/BA Steinhagen	Low	Low	Low	FE/E
Texas Heelsplitter	Neches	Lower Neches R	Low	Low	Low	FE/E
Texas Heelsplitter	Trinity	Grapevine LK	FE/E	FE/E	FE/E	FE/E
Texas Heelsplitter	Trinity	Trinity R/Livingston	Low	Low	Low	FE/E
Louisiana Pigtoe	Red	Little R/Rolling FK	Moderate	Moderate	Low	Low
Louisiana Pigtoe	Red	Cossatot R	High	High	High	Low
Louisiana Pigtoe	Red	Saline R (Little)	Moderate	Moderate	Low	Low
Louisiana Pigtoe	Red	Lower Little R	FE/E	FE/E	FE/E	FE/E
Louisiana Pigtoe	Big Cypress	Big Cypress Bayou	Moderate	Moderate	Moderate	Low
Louisiana Pigtoe	Calcasieu-Mermentau	Upper Calcasieu R	Low	Low	Low	FE/E
Louisiana Pigtoe	Pearl	Pearl R	Low	Low	Low	FE/E
Louisiana Pigtoe	Sabine	Sabine R	FE/E	FE/E	FE/E	FE/E
Louisiana Pigtoe	Sabine	Bayou Anacoco	Moderate	Low	Moderate	FE/E
Louisiana Pigtoe	Neches	Angelina R	Low	Low	Low	FE/E
Louisiana Pigtoe	Neches	Neches R	High	High	Low	Low
Louisiana Pigtoe	Neches	Lower Neches R	Low	Low	Low	FE/E
Louisiana Pigtoe	San Jacinto	E FK San Jacinto R	Low	Low	FE/E	FE/E

Redundancy

Both Louisiana Pigtoe and Texas Heelsplitter lose redundancy under Scenario 2 with a particularly severe outcome for Texas Heelsplitter populations, which are functionally extirpated/extirpated throughout the range of the species (Table 6.6 and 6.7). Under our projections, Louisiana Pigtoe would have three remaining populations within the Red River basin, one in the Big Cypess-Sulphur basin, and one in the Neches River basin in 50 years. The remaining five Louisiana Pigtoe populations are projected to be in low condition and vulnerable to extirpation.

Representation

Under Scenario 2, Louisiana Pigtoe lose four of the seven current representation areas in 50 years (Table 6.6 and 6.7), with eight of 13 populations remaining or becoming functionally extirpated/extirpated; therefore, the adaptive capacity and representation of this species is projected to be severely reduced from future environmental change. The five populations of Louisiana Pigtoe projected to remain in 50 years are in low condition. Texas Heelsplitter are projected to be functionally extirpated/extirpated throughout their range in 50 years (i.e., extinct), and the remaining Louisiana Pigtoe populations are extremely vulnerable to extinction under Scenario 2.

Table 6.7. Summary of condition of Louisiana Pigtoe and Texas Heelsplitter populations under Future Scenario 2 (Severe Increase in Stressors).

Projected condition	Number of Louisiana Pigtoe populations (n=13 within 6 representation areas)			Number of Texas Heelsplitter populations (n=5 within 3 representation areas)		
	10-year	25-year	50-year	10-year	25-year	50-year
High	2	1	0	0	0	0
Moderate	3	2	0	0	0	0
Low	6	7	5	3	3	0
Extirpated/ functionally extirpated	2	3	8	2	2	5
Number of representation areas	7	6	3	2	2	0

6.D. STATUS ASSESSMENT SUMMARY

Using the best available information, this report used scenario planning to develop forecasts of likely future conditions of Louisiana Pigtoe and Texas Heelsplitter populations across their current ranges. The goal of this report is to describe the viability of each species in terms of resiliency, representation, and redundancy. This report considers the possible future condition of each species, and a range of potential scenarios that include important influences on the current and future status of Louisiana Pigtoe and Texas Heelsplitter. The results of this analysis describe a range of possible future conditions to assess whether or not populations of these species are likely to persist into the future.

Both of these species face a variety of risks from a variety of environmental stressors, including hydrological alterations to their habitat (loss of flow leading to dewatering, excessive flows leading to scouring), water quality degradation, loss of suitable substrates due to excessive sedimentation and other processes, and inundation leading to habitat fragmentation and population isolation. Other factors contribute, or exacerbate exposure, to these risks but are not directly driving population condition. These secondary factors include: depredation, invasive species, over-collection and/or vandalism, and host fish interactions, among others.

These risks together substantially affect the future viability of Louisiana Pigtoe and Texas Heelsplitter. If population resiliency (the ability to withstand stochastic events and described by demographic factors including population size and growth rate) is diminished, populations are more vulnerable to extirpation. Population extirpations result in losses to redundancy (the ability of a species to withstand catastrophic events) and diminished species representation (important breadth of genetic and ecological diversity).

Louisiana Pigtoe is currently represented by two high condition populations, four moderate condition populations, five low condition populations, and two functionally extirpated/extirpated populations. Given the likelihood of climate change and other anthropogenic effects in the foreseeable future, within 50 years we estimate five populations will become (or remain) functionally extirpated/extirpated, seven will be in low condition, and one population will be in moderate condition under Scenario 1 (moderate increase in stressors; Table 6.5). Under Scenario 2, we estimate eight populations will become (or remain) functionally extirpated/extirpated, with five low condition populations remaining within 50 years (severe increase in stressors; Table 6.7).

Texas Heelsplitter is currently represented by three low condition populations, and two functionally extirpated/extirpated populations. Given the ongoing effects of climate change and human activities on hydrology and habitat quality, within 50 years we estimate only one population will remain in low condition while four become (or remain) functionally extirpated or extirpated under Scenario 1 (moderate increase in stressors; Table 6.5). Under Scenario 2 we estimate all Texas Heelsplitter populations will become (or remain) functionally extirpated/extirpated within 50 years (severe increase in stressors; Table 6.7).

We recognize our forecasted future conditions under Scenarios 1 and 2 are based on dozens of variables that may change in unpredictable ways moving forward, therefore our projections may overestimate or underestimate the severity of threats or the actual real-world condition of either species in the future. However, we believe our future forecasts are likely to come to fruition without a considerable investment in mussel conservation, and perhaps more importantly, a significant reduction in global greenhouse gas emissions in the coming decades.

See Figures 6.2 and 6.3 for a series of maps that represents the forecasted future condition of each population by species relative to current condition. Larger maps are provided in Appendix C.

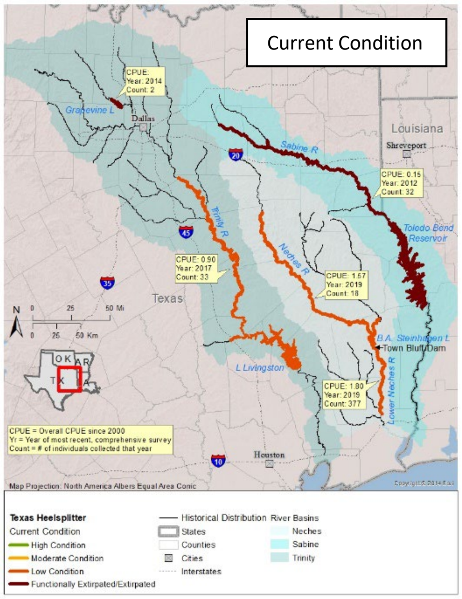
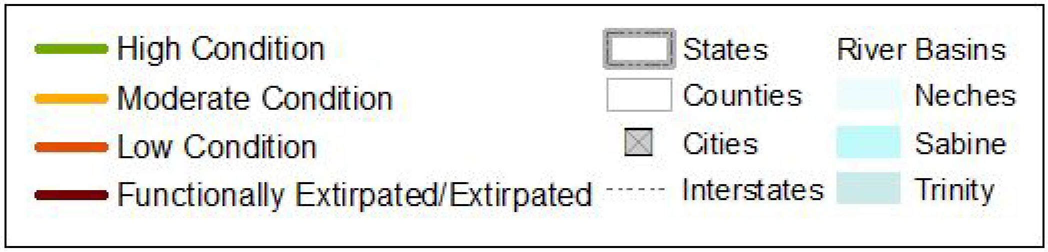
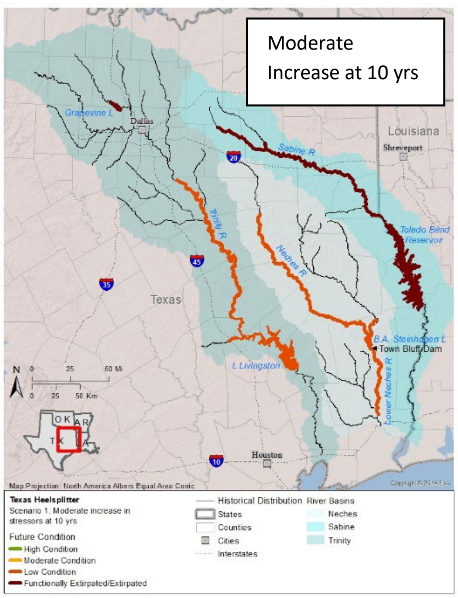
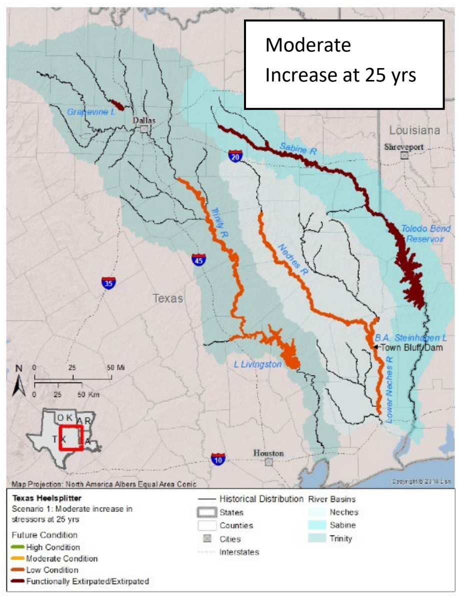
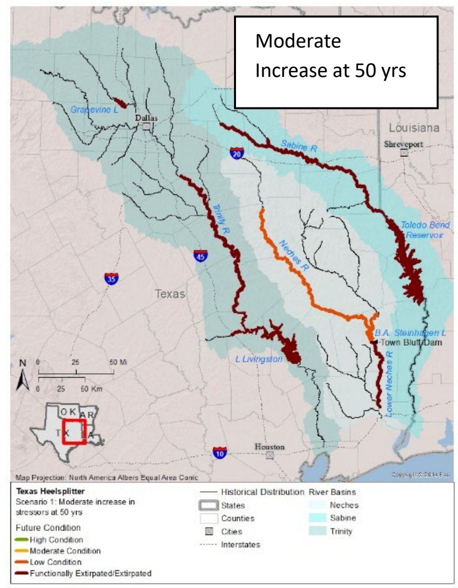


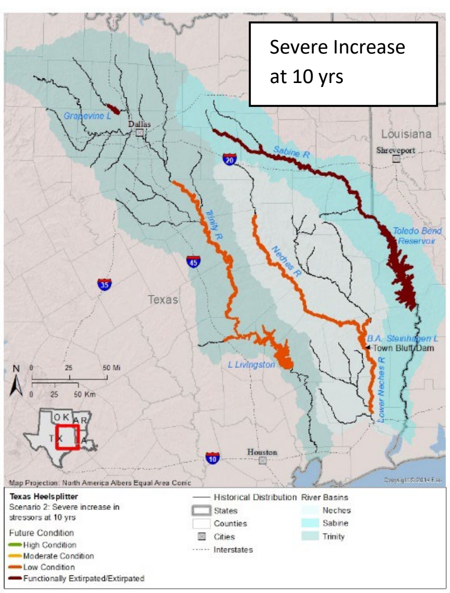
FIGURE 6.2. Summary of current and future population conditions for Texas Heelsplitter (See Appendix C for larger maps)

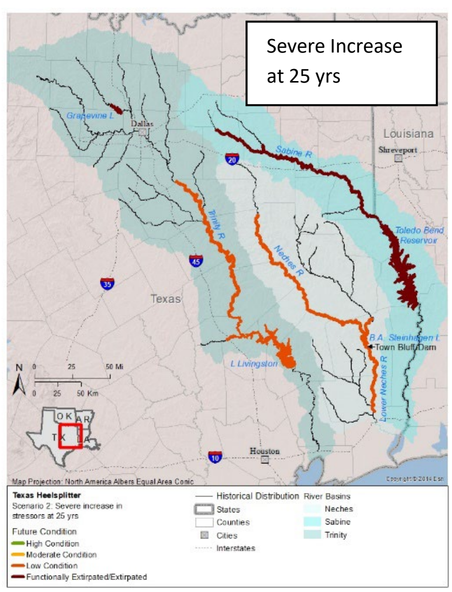


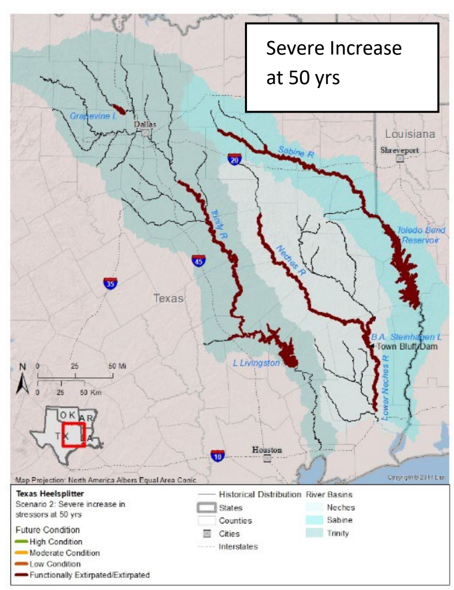












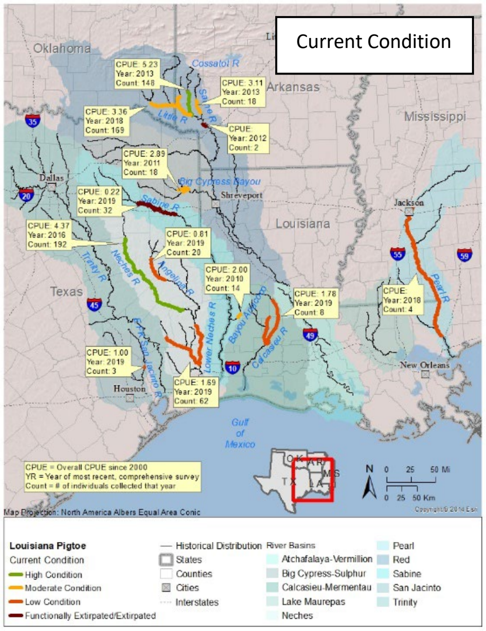
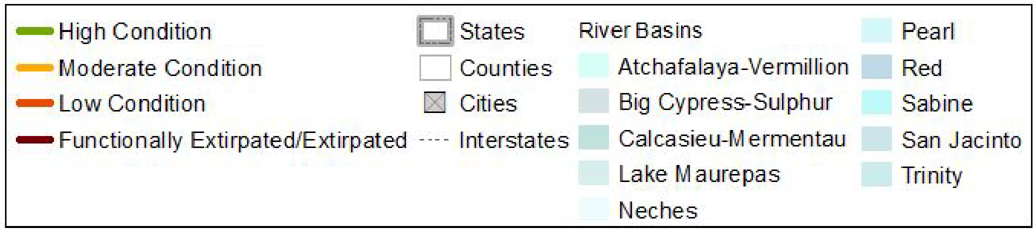
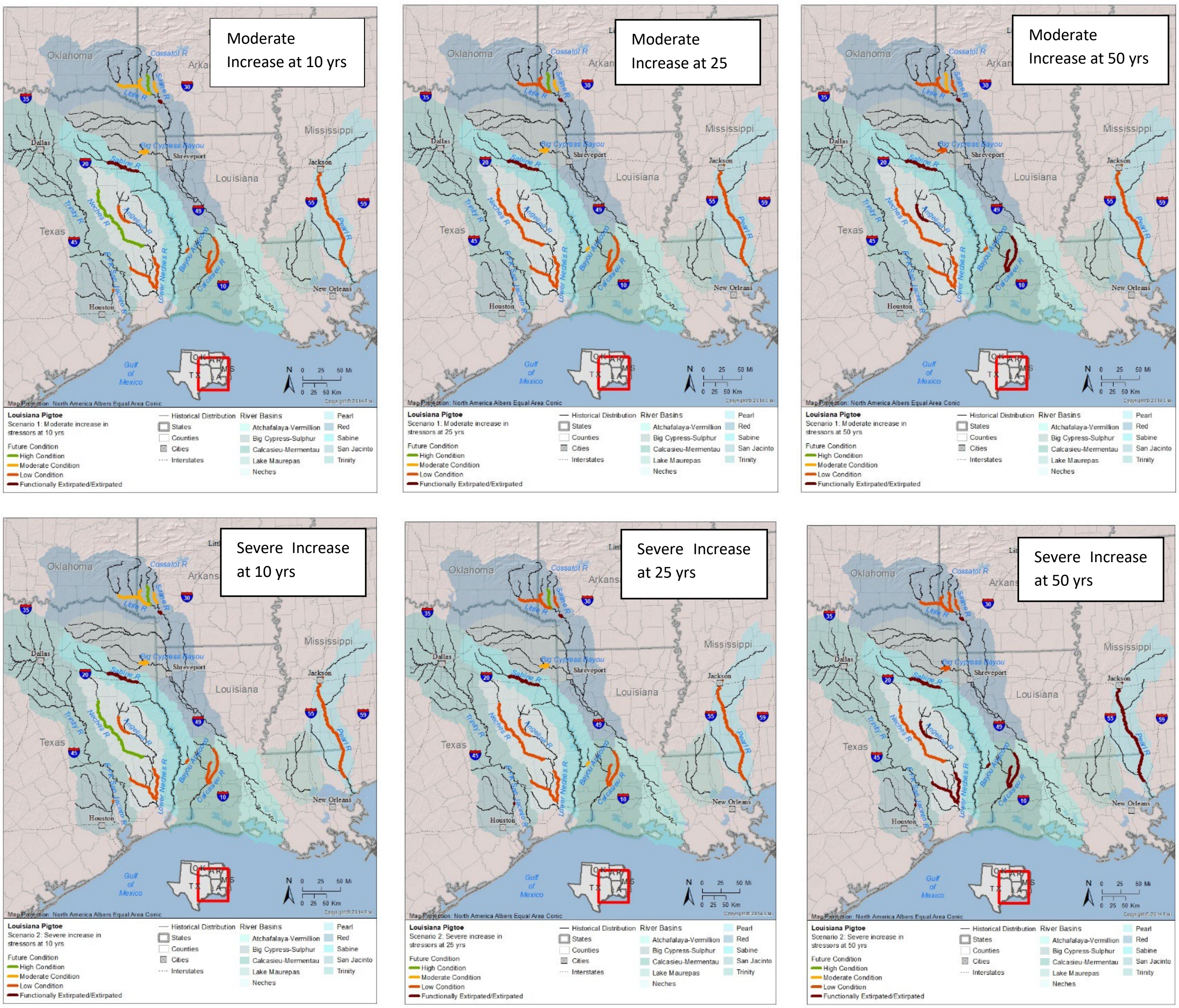


FIGURE 6.3. Summary of current and future population conditions for Louisiana Pigtoe (See Appendix C for larger maps)





APPENDIX A. LITERATURE CITED

Appendix A – Literature Cited

Abernathy, E., E. McCombs, L. Siefferman, and M. Gangloff. 2013. Effect of small dams on freshwater mussel population genetics in two southeastern USA streams. Freshwater Mollusk Biology and Conservation 16(1):21-28.

[AGFC]. Arkansas Game and Fish Commission. 2018. Email from Bill Posey regarding Louisiana pigtoe observation data in Arkansas. February 8, 2018. 2 pp.

Allen, D.C. and C.C. Vaughn. 2010. Complex hydraulic and substrate variables limit freshwater mussel species richness and abundance. Journal of the North American Benthological Society 29(2): 383-394.

Andreadis, K.M., and D.P. Lettenmaier. 2006. Trends in 20th century drought over the continental United States. Geophysical Research Letters 33. 4 pp.

[APA]. American Planning Association. 2019. Falling Dominoes: A Planner's Guide to Drought and Cascading Impacts. Report available at https://planning-org-uploaded-media.s3.amazonaws.com/publication/download_pdf/Falling-Dominoes-Planners-Guide-to-Drought-and-Cascading-Impacts.pdf. Accessed on February 12, 2020.

Arbuckle, K.E. and J.A. Downing. 2002. Freshwater mussel abundance and species richness: GIS relationships with watershed land use and geology. Canadian Journal of Fisheries and Aquatic Science 59: 310-316.

Archambault, J.M., W.G. Cope, and T.J. Kwak. 2013. Burrowing, byssus, and biomarkers: behavioral and physiological indicators of sublethal stress in freshwater mussels (Unionidae). Marine and Freshwater Behaviour and Physiology 46(4): 229-250.

Arey, L.B. 1932. The formation and structure of the glochidial cyst. Biological Bulletin 62:212-221.

Arm, J., T. Carlin, N. Kahal. N. Riseley-White, and E. Ross. 2014. A water budget analysis to support sustainable water in the Black River Basin, New Mexico. Group Project Report, Masters of Environmental Science and Management Program, University of California, Santa Barbara. 170 pp.

Arnold, W.R., D.W. Peterson, and K.W. Farrish. 2013. A survey of unionid mussels inhabiting streams of the Sabine National Forest, Texas. Texas Journal of Science 65(2): 21-39.

Augspurger, T., F. J. Dwyer, C. G. Ingersoll, and C. M. Kane. 2007. Advances and opportunities in assessing contaminant sensitivity of freshwater mussel (Unionidae) early life stages. Environmental Toxicology and Chemistry 26: 2025-2028.

Augspurger, T., A.E. Keller, M.C. Black, W.G. Cope, and F.J. Dwyer. 2003. Water quality guidance for protection of freshwater mussels (Unionidae) from ammonia exposure. Environmental Toxicology and Chemistry 22: 2569-2575.

Baker, S.M. and J.S. Levinton. 2003. Selective feeding by three North American freshwater mussels implies food competition with zebra mussels. Hydrobiologia 505: 97-105.

Bakken, D.S. 2013. Recruitment and survival of post-parasitic juvenile mussels in an East Texas river. University of Texas at Tyler, Tyler, Texas. 52 pp.

Banner, J.L., Jackson, C.S., Yang, Z., Hayhoe, K., Woodhouse, C., Gulden, L., Jacobs, K., North, G., Leung, R., Washington, W., Jiang, X., and R. Casteel. 2010. Climate change impacts on Texas water: a white paper assessment of the past, present, and future and recommendations for action. Texas Water Journal 1:1-19.

Barnhart, M.C., Haag, W.R., and R.N., William. 2008. Adaptations to host infection and larval parasitism in Unionida. Journal of the North American Benthological Society 27: 370-394.

Bertram, E.P. 2015. Confirmation of potential cyprinid hosts for a state threatened freshwater mussel of East Texas. Thesis. University of Texas at Tyler, Tyler, Texas. 65 pp.

Bio-West, Inc. 2019. Aquatic resource relocation report: Lower Neches Valley Authority canal mussel relocation, Beaumont, Texas. San Marcos, Texas. 10 pp.

Bio-West, Inc. 2021. Native mussel surveys in the LNVA canal system. Preliminary report to Lower Neches Valley Authority, April 2021. San Marcos, Texas. 13 pp.

Blakeslee, C.J., H.S. Galbraith, L.S., Robertson, and B.S.J., White. 2013. The effects of salinity exposure on multiple life stages of a common freshwater mussel, *Elliptio complanata*. Environmental Toxicology and Chemistry 32:2849-2854.

Bogan, A.E. 1993. Freshwater bivalve extinctions (Mollusca: Unionoida): a search for causes. American Zoologist 33:599-609.

Bonner, T.H., E.L. Oborny, B.M. Littrell, J.A. Stoeckel, B.S. Helms, K.G. Ostrand, P.L. Duncan, and J. Conway. 2018. Multiple freshwater mussel species of the Brazos River, Colorado River, and Guadalupe River basins. CMD 1 - 6233CS. Final Report to Texas Comptroller of Public Accounts. Texas State University, San Marcos, Texas. 306 pp.

Bosman, B,B. Christies, M. Hart, J. Morton, and C. Randklev. 2015. Host confirmed for *Potamilus amphichaenus* and *Potamilus metnecktayi*. Ellipsaria 17:15.

Bouldin, J., W.R. Posey II, and J.L. Harris. 2013. Status survey of Ouachita rock pocketbook, *Arkansia wheeleri* Ortmann and Walker 1912, scaleshell, *Leptodea leptodon* (Rafinesque 1820), and rabbitsfoot, *Quadrula c. cylindrica* (Say 1817), in the Little River Basin, Arkansas. Final report for Arkansas Game and Fish Commission, Perryton, Arkansas. 30 pp.

Box, J., and J. Mossa. 1999. Sediment, land use, and freshwater mussels: prospects and problems. Journal of the North American Benthological Society 18:99 117.

Burch, J.B. 1975. Freshwater Unionacean Clams (Mollusca, Pelecypoda) of North America (No. 11). Malacological publications. Accessed on June 16, 2020.

Burlakova, L.E. and A.Y. Karatayev. 2007. The effect of invasive macrophytes and water level fluctuations on unionids in Texas impoundments. Hydrobiologia 586:291-302. DOI 10.1007/s10750-007-0699-1.

Burlakova, L.E., A.Y. Karatayev, V.A. Karatayev, M.E. May, D.L. Bennett, and M.J. Cook. 2011a. Endemic species: contribution to community uniqueness, effect of habitat alteration, and conservation priorities. Biological Conservation 14: 155-165.

Burlakova, L.E., A.Y. Karatayev, V.A. Karatayev, M.E. May, D.L. Bennett, and M.J. Cook. 2011b. Biogeography and conservation of freshwater mussels (Bivalvia: Unionidae) in Texas: patterns of diversity and threats. Diversity and Distributions 17: 393-407.

Burlakova, L.E., D. Campbell, A.Y. Karatayev, and D. Barclay. 2012. Distribution, genetic analysis and conservation priorities for rare Texas freshwater molluscs in the genera *Fusconaia* and *Pleurobema* (Bivalvia: Unionidae). Aquatic Biosystems 2012 8:12. 15 pp.

Carroll, C., J.A. Vucetich, M.P. Nelson, D.J. Rohlf, and M.K. Phillips. 2010. Geography and recovery under the U.S. Endangered Species Act. Conservation Biology 24:395-403.

Cherry, D.S., J.L. Scheller, N.L. Cooper, and J.R. Bidwell. 2005. Potential effects of Asian clam (*Corbicula fluminea*) die-offs on native freshwater mussels (Unionidae) I: water-column ammonia levels and ammonia toxicity. Journal of the North American Benthological Society 24:369-380.

Christensen, P., K. Gillingham, and W. Nordhaus. 2018. Uncertainty in forecasts of long-run economic growth. Proceedings of the National Academy of Sciences May 2018, 115 (21) 5409-5414; DOI: 10.1073/pnas.1713628115. Available at https://www.pnas.org/content/115/21/5409.abstract

Churchill, C.J. and D.P. Quigley. 2018. Downstrean dispersal of zebra mussels (*Dreissena polymorpha*) under different flow conditions in a coupled lake-stream ecosystem. Biological Invasions 20:1113-1127.

Clark, B. and W. Mangham. 2019. Middle Trinity River research, 2018. White paper. Trinity River Authority, Arlington, Texas. 49 pp.

Cooper, N.L., J.R. Bidwell, and D.S. Cherry. 2005. Potential effects of Asian clam (*Corbicula fluminea*) die-offs on native freshwater mussels (Unionidae) II: porewater ammonia. Journal of the North American Benthological Society 24:381-394.

Cope, W.G, R.B. Bringolf, D.B. Buchwalter, T.J. Newton, C.G. Ingersoll, N. Wang, T. Augspurger, F.J. Dwyer, M.C. Barnhart, R.J. Neves, and E. Hammer. 2008. Differential exposure, duration, and sensitivity of unionoidean bivalve life stages to environmental contaminants. Journal of the North American Benthological Society 27:451-462.

Corbett, T. 2020. Email correspondence regarding depth of occupied Louisiana Pigtoe habitat in Neches River below B.A. Steinhagen Dam. March 30, 2020. 9 pp.

Daraio, J.A., L.J. Weber, S.J. Zigler, T.J. Newton, and J.M. Nestler. 2012. Simulated effects of host fish distribution on juvenile unionid mussel dispersal in a large river. USGS Staff -- Published Research. Paper 591. http://digitalcommons.unl.edu/usgsstaffpub/591

Dascher, E.D., L.E. Burlakova, A.Y. Karatayev, D.F. Ford, and A.N. Schwalb. 2017. Distribution of unionid freshwater mussels and host fishes in Texas. A study of broad-scale spatial patterns across basins and a strong climate gradient. Hydrobiologia 810(1):315-331. doi:10.1007/s10750-017-3168-5

Davidson, C.L. 2017. Population structure of selected freshwater mussel (Bivalvia: Unionoida) beds in the Little River, Pond Creek National Wildlife Refuge - Phase I Final Report. USFWS, Arkansas Ecological Services Field Office, Conway, Arkansas. 38 pp.

Davidson, C.L., T. Brady, and T. Fotinos. 2014. Status and distribution of freshwater mussels (Bivalvia: Unionacea) inhabiting Little River from the Arkansas - Oklahoma state line to Millwood Lake, Arkansas. U.S. Fish and Wildlife Service, Arkansas Ecological Services Field Office, Conway, Arkansas. 34 pp.

Dickson, J. 2018. Habitat associations and detectability of three unionid species along the upper Sabine River in East Texas. Biology Theses. Paper 54. University of Texas at Tyler, Tyler, Texas. 41 pp. http://hdl.handle.net/10950/1166

Diffenbaugh, N.S., F. Giorgi, and J.S. Pal. 2008. Climate change hotspots in the United States. Geophysical Research Letters 35: L16709. 5 pp. doi:10.1029/2008GL035075

Douda, K., Horky, P., and M., Bily. 2012. Host limitation of the thick-shelled river mussel: identifying the threats to declining affiliate species. Animal Conservation 15:536-544.

Duncan, A., and T. Nobles. 2012. Effects of municipal wastewater effluent on freshwater mussel growth and survival. Report SR-13-02. City of Austin Watershed Protection. 24 pp.

East, M.B, B.M. Fillmore, and D.B. Ligon. 2013. Feeding behavior of captive-reared juvenile alligator snapping turtles (Macrochelys temminckii). Southeastern Naturalist 12:692–702.

[EPA]. Environmental Protection Agency. 2007. Risks of atrazine use to eight federally listed endangered freshwater mussels: Pink Mucket Pearly (*Lampsilis abrupta*), Rough Pigtoe (*Pleurobema plenum*), Shiny Pigtoe Pearly (*Fusconaia edgariana*), Fine-rayed Pigtoe (*Fusconaia cuneolus*), Heavy Pigtoe (*Pleurobema taitianum*), Ovate Clubshell (*Pleurobema perovatum*), Southern Clubshell (*Pleurobema decisum*), and Stirrup Shell (*Quadrula stapes*). Pesticide effects determination. Environmental Fate and Effects Division, Office of Pesticide Programs, Washington, D.C. 200 pp.

[EPA]. Environmental Protection Agency. 2018. TSCA Chemical Substance Inventory. Available at https://www.epa.gov/tsca-inventory/about-tsca-chemical-substance-inventory. Accessed June 28, 2018.

Ford, D.F. 2013. Ground-truthing Maxent in East Texas Rivers. Biology Thesis. Paper 12. University of Texas at Tyler, Tyler, Texas. 94 pp.

Ford, D.F. and A.M. Oliver. 2015. The known and potential hosts of Texas mussels: Implications for future research and conservation efforts. The Journal of Freshwater Mollusk Conservation Society 18:1-14.

Ford, N.B. 2013. Surveys of freshwater mussels in three units of the Big Thicket Preserve. University of Texas at Tyler, Tyler, Texas. 16 pp.

Ford, N.B., J. Gullett, and M.E. May. 2009. Diversity and abundance of unionid mussels in three sanctuaries on the Sabine River in northeast Texas. Texas Journal of Science 61(4): 279-294.

Ford, N.B., L. Williams, and M.G. Williams, and M. May. 2010. Final report: Surveys for rare freshwater unionid mussels and fish in upper reaches of the Sabine River to gather population information on threatened species. Prepared for Texas Parks and Wildlife Department, State Wildlife Grants Program, Austin, Texas. 17 pp.

Ford, N.B., K. Heffentrager, D.F. Ford, A.D. Walters, and N. Marshall. 2014. Significant recent records of Unionid mussels in Northeast Texas. Walkerana 17(1): pp. 8 – 15.

Ford, N.B., L. Williams, M.G. Williams, J. Banta, J. Placyk, and H. Hawley. 2016. Final report: endangered species research projects for freshwater mussels, Region 2, East Texas for Texas Comptroller of Public Accounts. University of Texas at Tyler, Tyler, Texas. 58 pp.

Ford, N.B., L. Williams, M. Williams, and C. Robertson. 2018. Final report: Population ecology of two state listed species, the Southern Hickorynut (*Obovaria arkansasensis*) and the Louisiana Pigtoe (*Pleurobema riddellii*). University of Texas at Tyler, Tyler, Texas. 21 pp.

Francis-Floyd, R. 2011. Dissolved oxygen for fish production. Publication FA 27. Fisheries and Aquatic Sciences Department, Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida. 3 pp.

Friedrich, K., R.L. Grossman, J. Huntington, P.D. Blanken, J. Lenters, K.D. Holman, D. Gochis, B. Livneh, J. Prairie, E. Skeie, N.C. Healey, K. Dahm, C. Pearson, T. Finnessey, S. J. Hook, and T. Kowalski. 2018. Reservoir evaporation in the western United States. Bulletin of the American Meteorological Society, January 2018: 167-187.

Frierson, L.S., 1898. *Unio (Lampsilis) amphichaenus*, n. sp. The Nautilus, 11: 109-110, plate 1. Available at https://www.biodiversitylibrary.org/page/1774677#page/121/mode/1up

Frierson, L. S. 1927. A classification and annotated check list of the American naiades. Baylor Univeristy Press, Waco, Texas. Available at https://babel.hathitrust.org/cgi/pt?id=coo.31924002812125&view=1up&seq=2

Fuller, M.R. and M.W. Doyle. 2018. Gene flow simulations demonstrate resistance of long-lived species to genetic erosion from habitat fragmentation. Conservation Genetics 19:1439-1448.

Fuller, P., K. Dettloff, and R. Sturtevant, 2022, Aplodinotus grunniens Rafinesque, 1819: U.S. Geological Survey, Nonindigenous Aquatic Species Database, Gainesville, FL, https://nas.er.usgs.gov/queries/FactSheet.aspx?SpeciesID=946, Revision Date: 8/3/2021, Peer Review Date: 8/3/2021. Accessed on January 12, 2022.

Gagnon, P.M., Golladay, S.W., Michener, W.K., and M.C. Freeman. 2004. Drought responses of freshwater mussels (Unionidae) in coastal plain tributaries of the Flint River Basin, Georgia. Journal of Freshwater Ecology, 19:4, 667-679, DOI: 10.1080/02705060.2004.9664749

Galbraith, H.S., D.E. Spooner, and C.C. Vaughn. 2010. Synergistic effects of regional climate patterns and local water management on freshwater mussel communities. Biological Conservation 143:1175-1183.

Galbraith, H.S, and C.C. Vaughn. 2009. Temperature and food interact to influence gamete development in freshwater mussels. Hydrobiologia 636:35-47.

Ganser A.M., T.J. Newton, and R.J. Haro. 2013. The effects of elevated water temperature on native juvenile mussels: implications for climate change. Freshwater Science 32(4):1168–1177. Available at https://www.journals.uchicago.edu/doi/10.1899/12-132.1

Gascho-Landis, A.M., W.R. Haag, and J.A. Stoeckel. 2013. High suspended solids as a factor in reproductive failure of a freshwater mussel. Freshwater Science 32:70-81.

Gascho-Landis, A.M., and J.A. Stoeckel. 2016. Multi-stage disruption of freshwater mussel reproduction by high suspended solids in short- and long-term brooders. Freshwater Biology 61:229-238. doi:10.1111/fwb.12696

Gillis, P.L., R. McInnis, J. Salerno, S.R. de Solla, M.R. Servos, and E.M. Leonard. 2017. Municipal wastewater treatment effluent-induced effects on freshwater mussel populations and the role of mussel refugia in recolonizing an extirpated reach. Environmental Pollution 225: 460-468.

Glen, A.R. 2017. Examining the relationship between mesohabitats and freshwater mussels in an east Texas river. Biology Theses. Paper 48. University of Texas at Tyler, Tyler, Texas. 37 pp. http://hdl.handle.net/10950/586

Golladay, S.W., P. Gagnon, M. Kearns, J.M. Battle, and D.W. Hicks. 2004. Response of freshwater mussel assemblages (Bivalvia: Unionidae) to a record drought in the Gulf Coastal Plain of southwestern Georgia. Journal of the North American Benthological Society 23:494-506.

Goudreau, S.E., R.J. Neves, and R.J. Sheehan. 1993. Effects of wastewater treatment plant effluents on freshwater mollusks in the upper Clinch River, Virginia, USA. Hydrobiologia 252: 211-230.

Gregory, L., N. Boitnott, and A. Castilaw. 2014. Attoyac Bayou watershed protection plan. Texas Water Resource Institute Technical Report - 458. 100 pp.

Haag, W.R. 2012. North American freshwater mussels: natural history, ecology, and conservation. Cambridge University Press. New York. 505 pp.

Haag W.R. and A.L. Rypel. 2010. Growth and longevity in freshwater mussels: evolutionary and conservation implications. Biological Reviews. 86:225-247.

Haag, W.L. and M.L. Warren. 2008. Effects of severe drought on freshwater mussel assemblages. Transactions of the American Fisheries Society 137:1165-1178.

Haas, F. 1969. Superfamilia Unionacea. Das Tierreich, 88: 1–663.

Heffentrager, K.B. 2013. Utilizing Maxent to improve and explain a species distribution model for freshwater mussel species in East Texas. University of Texas at Tyler, Tyler, Texas. 102 pp.

Hinkle, E. 2018. Suitable host fish, population structure, and life-history characteristics for the state-listed, Louisiana pigtoe, *Pleurobema riddellii*. Biology Theses. Paper 56. University of Texas at Tyler, Tyler, Texas. 35 pp. http://hdl.handle.net/10950/1187

Hirabayashi, Y., R. Mahendran, S. Koirala, L. Konoshima, D. Yamazaki, S. Watanabe, H. Kim, and S. Kanae. 2013. Global flood risk under climate change. Nature Climate Change 3: 816-821.

Hoggarth, M.A. 1988. The use of glochidia in the systematics of the Unionidae (Mollusca: Bivalvia). Doctoral dissertation, The Ohio State University. 367 pp.

Hollis, L. 2013. Freshwater mussel (Bivalvia: Unionidae) assemblages in the Sabine River tributaries in the Sabine National Forest during an exceptional drought in East Texas. Master's Thesis: Stephen F. Austin State University, Nacogdoches, Texas. 228 pp.

Horne, R.H., and S. McIntosh. 1979. Factors influencing distribution of mussels in the Blanco River of Central Texas. The Nautilus 94:119-133.

Howells, R.G. 1996. Distributional surveys of freshwater bivalves in Texas: Progress report for 1994. Management Data Series No. 120. Texas Parks and Wildlife Department, Inland Fisheries Division, Austin, Texas. 57 pp.

Howells, R. 1997. Status of freshwater mussels (Bivalvia: Unionidae) of the Big Thicket region of east Texas. Texas Journal of Science 49(3):21-34.

Howells, R.G. 2000. Distributional surveys of freshwater bivalves in Texas: Progress Report for 1999. Texas Parks and Wildlife Management Data Series 170. Austin, Texas. 56 pp.

Howells, R.G. 2006. Final report: statewide freshwater mussel survey. Federal Aid Grant number T-15-P. 106 pp.

Howells, R.G. 2010a. Louisiana pigtoe (*Pleurobema riddellii*): summary of selected biological and ecological data for Texas. Biostudies, Kerrville, Texas. 18 pp.

Howells, R.G. 2010b. Texas Heelsplitter (*Potamilus amphichaenus*): summary of selected biological and ecological data for Texas. Biostudies, Kerrville, Texas. 21 pp.

Howells, R.G. 2014. Field Guide to Texas Freshwater Mussels, 2nd edition. BioStudies, Kerrville, Texas. 141 pp.

Howells, R.G., R.W. Neck, and H.D. Murray. 1996. Freshwater Mussels of Texas. Texas Parks and Wildlife Department Inland Fisheries Division. Austin, Texas. 281 pp.

Howells, R.G., C. M. Mather, and J.A.M. Bergmann. 1997. Conservation status of selected freshwater mussels in Texas. Pages 117 – 127 in K.S. Cummings, A.C. Buchanan, C.A. Mayer, and T.J. Niamo. Conservation and management of freshwater mussels II: initiatives for the future. Proceedings of a UMRCC symposium, October 1995, St. Louis Missouri. Upper Mississippi River Conservation Committee, Rock Island, Illinois.

[IEc]. Industrial Economics, Incorporated. 2021. Screening analysis of the likely economic impacts of critical habitat designation for the Louisiana Pigtoe and Texas Heelsplitter [Memorandum]. Prepared for U.S. Fish and Wildlife Service, September 1, 2021. 32 pp.

Inoue, K. 2018. Email correspondence regarding genetic confirmation of Louisiana Pigtoe in Little River, Oklahoma. June 11, 2018. 3 pp.

Inoue, K., D.M. Hayes, J.L. Harris, N.A. Johnson, C.L. Morrison, M.S. Eackles, T.L. King, J.W. Jones, E.M. hallerman, A.D. Christian, and C.R. Randklev. 2018. The Pleurobemini (Bivalvia : Unionida) revisited:

molecular species delineation using a mitochondrial DNA gene reveals multiple conspecifics and undescribed species. Invertebrate Systematics 32:689-702.

[IPCC]. Intergovernmental Panel on Climate Change. 2013. Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, New York, USA.

[IPCC]. Intergovernmental Panel on Climate Change. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

[IPCC]. Intergovernmental Panel on Climate Change. 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou, eds.] Cambridge University Press. In Press.

Johnson, P. M. 2001. Habitat associations and drought responses of freshwater mussels in the lower Flint River basin. Master's thesis. University of Georgia, Athens, Georgia. 30 pp.

Johnson, N.A., C.H. Smith, C.E. Beaver, P.D. Hartfield, and C.R. Randklev. 2019. Integrating molecular and morphological data to confirm new records of *Pleurobema riddellii*, a species being considered for ESA protection. Web presentation by Nathan Johnson to SSA core team, May, 5, 2019. 12 pp.

Jones, J.W., R.J. Neves, M.A. Patterson, C.R. Good, and A. DiVittorio. 2001. A status survey of freshwater mussel populations in the upper Clinch River, Tazewell County, Virginia. Banisteria 17:20-30.

Jones, J.W., E.M. Hallerman, and R.J. Neves. 2006. Genetic management guidelines for captive propagation of freshwater mussels (Unionoidea). Journal of Shellfish Research 25:527-535.

Kaller, M.D. and W.E. Kelso. 2003. Effects of feral swine on water quality in a coastal bottomland stream. In Proceedings of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies (No. 57, pp. 291-298).

Kaller, M.D. and W.E. Kelso. 2006. Swine activity alters invertebrate and microbial communities in a coastal plain watershed. The American Midland Naturalist, 156:163-178.

Kaller, M.D., J.D. Hudson III, E.C. Achberger, and W.E. Kelso. 2007. Feral hog research in western Louisiana: expanding populations and unforeseen consequences. Human-Wildlife Conflicts 1:168-177.

Karatayev, A.Y. and L.E. Burlakova. 2007. East Texas mussel survey. State Wildlife Grants Program report submitted to Texas Parks and Wildlife Department, Austin, Texas. 177 pp.

Karatayev, A.Y. and L.E. Burlakova. 2008. Distributional survey and habitat utilization of freshwater mussels. Final report submitted to Texas Water Development Board, Austin, Texas. (Grant # 434135). 47 pp.

Kelso, W.E, M.D. Kaller, M. Fries, and B. Trumbo. 2011. A survey of fishes and mussels inhabiting the Calcasieu, Mermentau, Vermillion, and lower Sabine River systems in Louisiana. Final report for Louisiana Department of Wildlife and Fisheries, Coastal Non-game Division, Baton Rouge, Louisiana. 34 pp.

Khan, J., C.H. Smith, K. Inoue, M. Hart, and C. Randklev. 2018. A survey and assessment of taxonomy, phylogeny and population genetics of critically endangered freshwater mussels in east Texas to assess their conservation status. Annual report. Texas A&M University, College Station, Texas. 22 pp.

Khan and Randklev. 2019. Assessment and review of hydrological relationships for mussels in east Texas. Interim Report. Texas A&M University, College Station, Texas. 13 pp.

Khan, J., M. Hart, J. Dudding, C. Robertson, R. Lopez, and C. Randklev. 2019. Evaluating the upper thermal limits of glochidia for select freshwater mussel species (Bivalvia:Unionidae) in central and east Texas and the implications for their conservation. Aquatic Conservation: Marine and Freshwater Ecosystems 29: 1202-1215.

Kinney, S. 2019. Email correspondence regarding new Louisiana pigtoe records from Calcasieu River, Louisiana. October 7, 2019. 3 pp.

Kinniburgh, F., M.G. Simonton, and C. Allouch. 2015. Come heat and high water: climate risk in the southeastern U.S. and Texas. A product of the risky business project. 114 pp. Available at http://riskybusiness.org/site/assets/uploads/2015/09/Climate-Risk-in-Southeast-and-Texas.pdf

Kondolf, G.M. 1997. Hungry Water: Effects of Dams and Gravel Mining on River Channels. Environmental Management 21:533-551.

LaFontaine, J.H., R.M. Hart, L.E. Hay, W.H. Farmer, A.R. Bock, R.J. Viger, S.L. Markstrom, R.S. Regan, and J.M. Driscoll. 2019. Simulation of water availability in the Southeastern United States for historical and potential future climate and land-cover conditions. U.S. Geological Survey Scientific Investigations Report 2019–5039, 83 pp. https://doi.org/10.3133/sir20195039.

Lea, I. 1862. New Unionidæ of the United States and Arctic America. Journal of the Academy of Natural Sciences 5: 193-4. Available at https://biodiversitylibrary.org/page/35217896

Loaiciga, H.A., D.A. Maigment, and J.B. Valdes. 2000. Climate-change impacts in a regional karst aquifer, Texas, U.S.A. Journal of Hydrology 227:173-194.

Lopes-Lima, M., L.E. Burlakova, A.Y. Karatayev, K. Mehler, M. Seddon, and R. Sousa. 2018. Conservation of freshwater bivalves at the global scale: diversity, threats and research needs. Hydrobiologia 810:1-14.

[LNHP]. Louisiana Natural Heritage Program. 2018. The LNHP database provides information on the occurrence and distribution of rare species in Louisiana. http://www.wlf.louisiana.gov/wildlife/louisiana-natural-heritage-program

Mace, R. E. and S. C., Wade. 2008. In hot water? How climate change may (or may not) affect groundwater resources of Texas. Gulf Coast Association of Geological Societies Transaction 58:655-668.

Maloney, E.D., S.J. Camargo, E. Chang, B. Colle, R. Fu, K.L. Geil, Q. Hu, X. Jiang, N. Johnson, K.B. Karnauskas, J. Kinter, B. Kirtman, S. Kumar, B. Langenbrunner, K. Lombardo, L.N. Long, A. Mariotti, J.E. Meyerson, K.C. Mo, J.D. Neelin, Z. Pan, R. Seager, Y. Serra, A. Seth, J. Sheffield, J. Stroeve, J. Thibeault, S.P. Xie, C. Wang, B. Wyman, and M. Zhao. 2014. North American Climate in CMIP5 Experiments: Part III: Assessment of Twenty-First-Century Projections. Journal of Climate 27: 2230-2269. Available at https://journals.ametsoc.org/doi/full/10.1175/JCLI-D-13-00273.1

Mantua, N., I. Tohver, and A. Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Climate Change 102:187-223.

March, F.A., F.J. Dwyer, T. Augspurger, C.G. Ingersoll, N. Wang, and C.A. Mebane. 2007. An evaluation of freshwater mussel toxicity data in the derivation of water quality guidance and standards for copper. Environmental Toxicology and Chemistry: An International Journal, 26(10), pp.2066-2074.

Marshall, N.T. 2014. Identification of potential fish hosts from wild populations of state-threatened east Texas freshwater mussels using a molecular identification dataset. Thesis. University of Texas at Tyler, Tyler, Texas. 112 pp.

McMahon, R.F., and A.E. Bogan. 2001. Mollusca:Bivaliva. Chapter 11 (pp. 331-249) in Ecology and Classification of North American Freshwater Invertebrates, Second Edition. (Thorp, J.H., and A.P. Covich, editors). 1056 pp.

Mersinger, R.C. and N.J. Silvy. 2007. Range size, habitat use, and dial activity of feral hogs on reclaimed surface-mined lands in east Texas. Human-Wildlife Conflicts, 1(2), pp.161-167.

Mitchell, Z., J. McGuire, J. Abel, B. Hernandez, and A. Schwalb. 2018. Move on or take the heat: Can life history strategies of freshwater mussels predict their physiological and behavioral responses to drought and dewatering? Freshwater Biology: *in prep*.

Morrill, J.C., R.C. Bales, and M.H. Conklin. 2005. Estimating stream temperature from air temperature: implications for future water quality. Journal of Environmental Engineering 131:139-146.

Mullens, E.D., and R.A. McPherson. 2017. Texas: A weather and climate trends roadmap. South Central Climate Science Center, Norman, Oklahoma. 37 pp. Available at https://climateprojections.wixsite.com/transportation/texas

Neck, R.W. and R.G. Howells. 1995. Status survey of the Texas Heelsplitter, *Potamilus amphichaenus* (Frierson, 1898). Report on file with Texas Parks and Wildlife Department, Austin, Texas. p. 15

Neves, R.J. 1991. Mollusks. Pp. 251-319 in: K. Terwilliger, coordinator. Virginia's endangered species. Proceedings of a symposium, April 1989, Blacksburg, Virginia. McDonald & Woodward Publishing Co., Blacksburg.

Neves, R.J. and M.C. Odom. 1989. Muskrat predation on endangered freshwater mussels in Virginia. Journal of Wildlife Management 53: 934–941.

Neves, R.J., A.E, Bogan, J.D. Williams, S.A. Ahlstedt, and P.W. Hartfield. 1997. Status of aquatic mollusks in the southeastern United States: A downward spiral of diversity. In G. W. Benz & D. E. Collins (Eds.), Aquatic fauna in peril: The southeastern perspective (pp. 43–85). Decatur, GA: Southeast Aquatic Research Institute.

Newton, T.J., D.A. Woolnough, and D.L. Strayer. 2008. Using landscape ecology to understand and manage freshwater mussel populations. Journal of the North American Benthological Society 27(2):424-439.

Nico, L. and P. Fuller, 2022, *Cyprinella venusta* Girard, 1856: U.S. Geological Survey, Nonindigenous Aquatic Species Database, Gainesville, FL, https://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=521, Revision Date: 4/11/2006, Peer Review Date: 4/11/2006. Accessed on January 12, 2022.

Nico, L. and R. Sturtevant, 2022, *Pimephales vigilax* (Baird and Girard, 1853): U.S. Geological Survey, Nonindigenous Aquatic Species Database, Gainesville, FL, https://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=623, Revision Date: 9/12/2019, Peer Review Date: 2/6/2015. Accessed on January 12, 2022.

Nico, L., P. Fuller, M. Neilson, and W. Daniel. 2022, *Cyprinella lutrensis* (Baird and Girard, 1853): U.S. Geological Survey, Nonindigenous Aquatic Species Database, Gainesville, FL, https://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=518, Revision Date: 3/16/2021, Peer Review Date: 3/16/2021. Accessed on January 12, 2022.

Nobles, T., and Y. Zhang. 2015. Survival, growth and condition of freshwater mussels: effects of municipal wastewater effluent. PLoS One 10: 1:19.

Noyes, P.D., M.K. McElwee, H.D. Miller, B.W. Clark, L.A. Van Tiem, K.C. Walcott, K.N. Erwin, and E.D. Levin. 2009. The toxicology of climate change: environmental contaminants in a warming world. Environment International, 35(6), 971–986. Available at https://doi.org/10.1016/j.envint.2009.02.006

Pandolfo, T.J., W.C. Cope, C. Arellano, R.B. Bringolf, M.C. Barnhart, and E. Hammer. 2010. Upper thermal tolerances of early lifestages of freshwater mussels. Journal of the North American Benthological Society 29:959-969.

Pappas, E.A., D.R. Smith, C. Huang, W.D. Shuster, and J.V. Bonta. 2008. Impervious surface impacts to runoff and sediment discharge under laboratory rainfall simulation. Catena 72:146-152.

Patra, R.W., J.C. Chapman, R.P. Lim, P.C. Gehrke, and R.M. Sunderam. 2015. Interactions between water temperature and contaminant toxicity to freshwater fish. Environmental Toxicology and Chemistry, 34:1809–1817. https://doi.org/10.1002/etc.2990

Perkin, J.S. and T.H. Bonner. 2016. Historical changes in fish assemblage composition following water quality improvement in the mainstem Trinity River of Texas. River Research and Applications 32: 85-99.

Placyk, J.S. 2019. Assessment of gnomic differentiation between the Texas (*Fusconaia askewii*) and triangle (F. lananensis) pigtoe mussels. RFP No. 207c. Texas Comptroller of Public Accounts. Austin, Texas. 8 pp.

Poff, N.L., D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: A paradigm for river conservation and restoration. BioScience 4:769-784.

[RRC]. Texas Railroad Commission. 2021a. Historical crude oil production and well counts. https://www.rrc.texas.gov/oil-and-gas/research-and-statistics/production-data/historical-production-data/crude-oil-production-and-well-counts-since-1935/. Accessed December 2, 2021

[RRC]. Texas Railroad Commission. 2021b. Historical natural gas production and well counts since 1935. https://www.rrc.texas.gov/oil-and-gas/research-and-statistics/production-data/historical-production-data/natural-gas-production-and-well-counts-since-1935/. Accessed December 2, 2021.

[RRC]. Texas Railroad Commission. 2021c. Pipeline safety overview. https://www.rrc.texas.gov/media/bnkh2wgb/pipelinesafetypage.pdf. Accessed December 2, 2021.

Randklev, C.R. 2011. The Ecology and Paleobiogeography of Freshwater Mussels (Family: Unionidae) from Selected River Basins in Texas. Doctor of Philosophy (Biology), University of North Texas, Denton, Texas, May 2011. 109 pp.

Randklev, C.R., M.S. Johnson, E.T. Tsakiris, S.R. Oetker, K.J. Roe, S. McMurray, C.R. Robertson, J. Groce, and N. Wilkins. 2011. First account of a living population of False Spike, *Quadrula mitchelli* (Bivalvia: Unionidae), in the Guadalupe River, Texas. Ellipsaria 13:17- 19.

Randklev, C.R., H. Wang, A. Snelgrove, K. Snow, J. Groce, E. Tsakiris, B. Sowards, W.E. Grant, N. Wilkins, N. Ford, M. Williams, J. Banta, and L. Williams. 2013a. Developing predictive models for the occurrence of rare and threatened mussel species in Texas - Phase I - Final Report. Report on file with Texas Parks and Wildlife Department, Austin, Texas. 86 pp.

Randklev, C.R., J. Skorupski, B. Lundeen, and E.T. Tsarkiris. 2013b. New distributional cecords for four rare species of freshwater mussels (Family: Unionidae) in Southwestern Louisiana. The Southwestern Naturalist, 58(2): 268-273.

Randklev, C.R., M. Cordova, J.Groce, E. Tsarkiris, and B. Sowards. 2014b. Freshwater mussel (Family: Unionidae) of the lower Sabine River between U.S. Hwy 190 and Orange, Texas. Report on file with Texas Parks and Wildlife Department, Austin, Texas.

Randklev, C.R., N. Ford, S. Wolverton, J.H. Kennedy, C. Robertson, K. Mayes, and D. Ford. 2015. The influence of stream discontinuity and life history strategy on mussel community structure: a case study from the Sabine River, Texas. Hydrobiologia 770:173-191.

Randklev, C.R., M. Hart, J. Morton, J. Dudding, and K. Inoue. 2017a. Freshwater mussel (Family: Unionidae) data collection in the middle Trinity River. Final Report to Texas Parks and Wildlife Department. 22 pp.

Randklev, C.R., K. Inoue, M. Hart, and A. Pieri. 2017b. Assessing the Conservation Status of Native Freshwater Mussels (Family: Unionidae) in the Trinity River basin. Final Report to Texas Parks and Wildlife Department. Grant number TX E-164-R. 55 pp.

Randklev, C.R. 2018. Email correspondence regarding Louisiana pigtoe and Texas heelsplitter distribution and database files. March 15, 2018. 2 pp.

Randklev, C.R. 2019a. Email correspondence regarding Texas heelsplitter ecology in reservoirs. May 3, 2019. 8 pp.

Randklev, C.R. 2019b. Email correspondence regarding genetic confirmation of Louisiana pigtoe type specimen. August 1, 2019. 4 pp.

Randklev, C.R. 2019c. Email correspondence regarding genetic confirmation of Louisiana pigtoe in East Fork San Jacinto River, Texas, and Calcasieu River, Louisiana. October 15, 2019.

Redford, K.H., G. Amoto, J. Baillie, P. Beldomenico, E.L. Bennett, N. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G. M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to successfully conserve a (vertebrate) species? Bioscience 61:39–48.

Schwalb, A.N., T.J. Morris, N.E. Mandrak, and K. Cottenie. 2013. Distribution of unionid freshwater mussels depends on the distribution of host fishes on a regional scale. Diversity and Distributions 19: 446-454.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Veechi, H. Huang, N. Harnik, A. Leetmaa, N. Lau, C. Li, J. Velez, and N.Naik. 2007. Model projections of an imminent transition to a more arid climate in Southwestern North America. Science 316: 1181-1184.

Simpson, C.T. 1900. Synopsis of the naiades, or pearly fresh-water mussels. Proceedings of the United States National Museum 22: 501-1044. Available at https://biodiversitylibrary.org/page/32021246

Simpson, C.T. 1914. A descriptive catalogue of the naiades, or pearly fresh-water mussels. Parts I-III: 1540 pp. Smithsonian Institution. Available at https://biodiversitylibrary.org/page/12173861

Smith, D.G. 1985. Recent range expansion of the freshwater mussel *Anodonta implicata* and its relationship to clupeid fish restoration in the Connecticut River system. Freshwater Invertebrate Biology 4:105-108.

Smith, D.R., N.A. Allen, C.P. McGowan, J.A. Szymanski, S.R. Oetker, and H.M. Bell. 2018. Development of a species status assessment process for decisions under the U.S. Endangered Species Act. Journal of Fish and Wildlife Management 9:1-19.

Sparks, B.L. and D.L. Strayer. 1998. Effects of low dissolved oxygen on juvenile *Elliptio complanata* (Bivalvia: Unionidae). Journal of the North American Benthological Society 17:129-134.

Spooner, D.E. and C.C. Vaughn. 2007. Mussels of the Mountain Fork River, Arkansas and Oklahoma. Publications of the Oklahoma Biological Survey. 2nd Series. Vol 8:14-18.

Stern, E.M. 1976. The Freshwater Mussels (Unionidae) of the Lake Maurepas - Pontchartrain -Borgne Drainage System, Louisiana and Mississippi. LSU Historical Dissertations and Theses. 2944. 221 pp. https://digitalcommons.lsu.edu/gradschool_disstheses/2944

Strayer, D. L. 1999. Use of flow refuges by unionid mussels in rivers. Journal of the North American Benthological Society 18:468-476.

Strayer, D.L., J.A. Downing, W.R. Haag, T.L. King, J.B. Layzer, T.J. Newton, and S.J. Nichols. 2004. Changing perspectives on pearly mussels, North America's most imperiled animals. Bioscience 54:429-439.

Strayer, D.L. and H.M. Malcom. 2012. Causes of recruitment failure in freshwater mussel populations in southeastern New York. Ecological Applications 22: 1780-1790.

Strayer, D.L. and D. R. Smith. 2003. A guide to sampling freshwater mussel populations. American Fisheries Society, Monograph No. 8. 110 pp.

Strecker, J.K. 1931. Naiades or pearly fresh-water mussels of Texas. Baylor University Museum Special Bulletin Number Two. 71 pp.

Taylor, R.G., B. Scanlon, P. Doll, M. Rodell, R. van Beek, Y. Wada, L. Longuevergne, M. Leblanc, J.S. Famiglietti, M. Edmunds, L. Konikow, T.R. Green, J. Chen, M. Taniguchi, M. Bierkens, A. MacDonald, Y. Fan, R.M. Maxwell, Y. Yechieli, J. Gurdak, D.M. Allen, M. Shamsudduha, K. Hiscock, P. Yeh, and H. Treidel. 2013. Groundwater and climate change. Nature Climate Change 3:322-329.

[TCEQ]. Texas Commission on Environmental Quality. 2014. Cypress Creek Basin Summary Report. Written by Water Monitoring Solutions Inc. Published by the Northeast Texas Municipal Water District, 83 pp.

[TCEQ]. Texas Commission on Environmental Quality. 2014a. Texas Integrated Report. Index of Water Quality Impairments. 130 pp.

[TCEQ]. Texas Commission on Environmental Quality. 2014b. Texas Integrated Report. Water Bodies with Concerns for Use Attainment and Screening Levels. 199 pp.

[TCEQ]. Texas Commission on Environmental Quality. 2018a. 2016 Texas Integrated Report for the Clean Water Act Sections 305(b) and 303(D). Texas Commission on Environmental Quality. Austin, Texas.

[TCEQ]. Texas Commission on Environmental Quality. 2018b. Texas Commission on Environmental Quality. Water Datasets. Industrial and Municipal Wastewater Outfalls. https://www.tceq.texas.gov/gis/download-tceq-gis-data#waterdatasets Accessed on June 25, 2018.

[TDSHS]. Texas Department of State Health Services. 2018. Texas fish consumption advisory viewer. https://dshscpd.maps.arcgis.com/apps/webappviewer/index.html?id=0d443c1940ba4240923a46dfa08f0289 Accessed on October 18, 2018.

Terando L.A., K. Dow, K. Hiers, K.E. Kunkel, A. Lascurain, D. Marcy, M. Osland, and P. Schramm, 2018: Southeast. Pp. 743-808 in Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart, eds. Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II. U.S. Global Change Research Program, Washington, DC. doi: 10.7930/NCA4.2018.CH19

Terui, A., Y. Miyazaki, A. Yoshioka, and S.S. Matsuzaki. 2015. A cryptic Allee effect: spatial contexts mask and existing fitness – density relationship. Royal Society Open Science 2: 150034. http://dx.doi.org/10.1098/rsos.150034

Thorp, J.P. and A.P. Covich (editors). 2010. Ecology and Classification of North American Freshwater Invertebrates. Second Edition. Academic Press. San Diego, California. 1073 pp.

Timmons, J., J.C. Cathey, N. Dictson, and M. MacFarland. 2011. Feral hogs and water quality in Plum Creek. Texas AgriLife Extension Service, Texas A&M University System. 2 pp.

[TPWD]. Texas Parks and Wildlife Department. 2009. 15 freshwater mussels placed on state threatened list. Available at https://tpwd.texas.gov/newsmedia/releases/?req=20091105c. Accessed on May 30, 2018.

[TPWD]. Texas Parks and Wildlife Department. 2018. Giant salvinia. Available at https://tpwd.texas.gov/huntwild/wild/species/exotic/salvinia.phtml. Accessed on October 10, 2018.

[TPWD]. Texas Parks and Wildlife Department. 2019. The zebra mussel threat. Available at https://tpwd.texas.gov/huntwild/wild/species/exotic/zebramusselmap.phtml. Accessed on October 15, 2019.

[TRA]. Trinity River Authority. 2018. Trinity River Basin Highlights Report: Texas Clean Rivers Program. Trinity River Authority of Texas. Arlington, Texas. 195 pp.

Turgeon, D.D., A.E. Bogan, E.V. Coan, W.K. Emerson, W.G. Lyons, W.L. Pratt, C.F.E. Roper, A. Scheltema, E.G. Thompson, and J.D. Williams. 1988. Common and scientific names of aquatic invertebrates from the United States and Canada: mollusks. American Fisheries Society Special Publication, 16.

Turgeon, D.D., J.F. Quinn, Jr., A.E. Bogan, E.V. Coan, F.G. Hochberg, W.G. Lyons, P.M. Mikkelsen, R.J. Neves, C.F.E. Roper, G. Rosenberg, B. Roth, A. Scheltema, F.G. Thompson, M. Vecchione, and J.D. Williams. 1998. Common and scientific names of aquatic invertebrates from the United States and Canada: mollusks, 2nd edition. American Fisheries Society Special Publication 26, Bethesda, Maryland. 277 pp.

[TWDB]. Texas Water Development Board. 2015. 2016 Region C Water Plan Volume I Main Report. December 2015, 982 pp.

[USDA]. U.S. Department of Agriculture. 2019. USDA Animal and Plant Health Inspection Service – Wildlife Services Feral Hog Damage Webpage. Available at https://www.aphis.usda.gov/aphis/ourfocus/wildlifedamage/programs/nwrc/research-areas/sa_feral_swine/ct_project_feral_swine_ungulate

[USFWS]. U.S. Fish and Wildlife Service. 2006. Region 2. Environmental Contaminants Program. Recommended water quality for federally listed species in Texas. 117 pp.

[USFWS]. U.S. Fish and Wildlife Service. 2009a. Partial 90-Day Finding on a Petition to List 475 Species in the Southwestern United States as Threatened or Endangered With Critical Habitat; Proposed Rule. Federal Register Vol. 74, No. 240. December 16, 2009, p. 66889.

[USFWS]. U.S. Fish and Wildlife Service. 2009b. 90-Day Finding on Petitions To List Nine Species of Mussels From Texas as Threatened or Endangered With Critical Habitat. Federal Register Vol. 74, No. 239. December 15, 2009, p. 66265.

[USFWS]. U.S. Fish and Wildlife Service. 2011. Endangered and threatened wildlife and plants; 12-month finding on a petition to list Texas Fatmucket, Golden Orb, Smooth Pimpleback, Texas Pimpleback, and Texas Fawnsfoot as Threatened or Endangered. 76FR62166. 48 pp.

[USFWS]. U.S. Fish and Wildlife Service. 2014. Final Report: Status and distribution of freshwater mussels (Bivalvia: Unionoida) inhabiting Little River from the Arkansas – Oklahoma State line to Millwood Lake, Arkansas. U.S. Fish and Wildlife Service, Arkansas Ecological Services Field Office, Conway, Arkansas. 72032, p. 29.

[USFWS]. U.S. Fish and Wildlife Service. 2015. Status and distribution of freshwater mussels (Bivalvia: Unionoida) inhabiting the Saline River within Felsenthal National Wildlife Refuge. U.S. Fish and Wildlife Service, Arkansas Ecological Services Field Office, Conway, Arkansas. 72032, p. 5.

[USFWS]. U.S. Fish and Wildlife Service. 2016. USFWS species status assessment framework: an integrated analytical framework for conservation. Version 3.4, dated August 2016.

[USFWS]. U.S. Fish and Wildlife Service. 2017. Final Report: Population Structure of Selected Freshwater Mussel (Bivalvia: Unionoida) Beds in the Little River, Pond Creek National Wildlife Refuge – Phase I. U.S. Department of the Interior – Fish and Wildlife Service, Arkansas Ecological Services Field Office, Conway, Arkasas. 72032. p.8.

[USFWS]. U.S. Fish and Wildlife Service. 2018. Biologists field notes from mussel surveys in upper Guadalupe River basin, Texas. 4 pp.

[USGCRP]. U.S. Global Change Research Program. 2017. Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock, eds.]. U.S. Global Change Research Program, Washington, DC, USA, 470 pp., doi: 10.7930/J0J964J6.

[USGS]. U.S. Geological Survey. 1998. Water quality in the Trinity River basin, Texas, 1992-95. Circular 1171. 44 pp.

[USGS]. U.S. Geological Survey. 2008. Summary of annual mean and annual harmonic mean statistics of daily mean streamflow at 620 U.S. Geological Survey streamflow-gaging stations in Texas through water year 2007. 1287 pp.

[USGS]. U.S. Geological Survey. 2014. National Hydrography Dataset. Available at https://www.usgs.gov/core-science-systems/ngp/national-hydrography/about-national-hydrography-products

[USGS]. U.S. Geological Survey. 2019a. NAS – Nonindigenous Aquatic Species. *Corbicula fluminea* (Asian clam). Available at https://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=92. Accessed on October 16, 2019.

[USGS]. U.S. Geological Survey. 2019b. NAS – Nonindigenous Aquatic Species. Floating water hyacinth. Available at https://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=1130. Accessed on October 16, 2019.

[USGS]. U.S. Geological Survey. 2019c. NAS – Nonindigenous Aquatic Species. *Hydrilla verticillata* (hydrilla). Available at https://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=6. Accessed on October 16, 2019.

[USGS]. U.S. Geological Survey. 2019d. NAS – Nonindigenous Aquatic Species. *Salvinia molesta*. Available at https://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=298. Accessed on October 16, 2019.

[USGS]. U.S. Geological Survey. 2019e. NAS – Nonindigenous Aquatic Species. Zebra mussel. Available at https://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=5. Accessed October 16, 2019.

van Vuuren, D.P., J. Edmonds, M. Kainuma, R. Keywan, A. Thomson, K. Hibbard, G. Hurtt, T. Kram, V. Krey, J. Lamarque, T. Masui, M. Meinshausen, N. Nakicenovic, S. Smith, and S. Rose. 2011. The representative concentration pathways: an overview. Climatic Change 109: 5-31.

Vannote, R.L., and G.W. Minshall. 1982. Fluvial processes and local lithology controlling abundance, structure, and composition of mussel beds. Proceedings of the National Academy of Sciences 79:4103‑4107.

Vaughan, C.M. 2017. The role of tributaries in structuring mussel communities. Biology Theses. Paper 40. University of Texas Tyler, Tyler, Texas. Available at http://hdl.handle.net/10950/557

Vaughn, C.C. 2012. Life history traits and abundance can predict local colonization and extinction rates of freshwater mussels. Freshwater Biology 57: 982-992.

Vaughn, C.C. and C.M. Taylor. 1999. Impoundments and the decline of freshwater mussels: a case study of an extinction gradient. Conservation Biology 13:912-920.

Vaughn, C.C., S.J. Nichols, and D.E. Spooner. 2008. Community and food web ecology of freshwater mussels. Journal of the North American Benthological Society 27(2): 409-423.

Vidrine, M.F. 1985. Fresh-water mussels (Unionacea) of Louisiana: a zoogeographical checklist of post-1890 records. The Louisiana Environmental Professional 2(1): 50-59.

Vidrine, M.F. 1993. The historical distribution of freshwater mussels in Louisiana. Gail Q. Vidrine Collectables. Eunice, LA. 225 pp.

Walters, A.D. and N.B. Ford. 2013. Impact of drought on predation of a state-threatened mussel, *Potamilus amphichaenus*. The Southwestern Naturalist 58:479-481.

Wang, N., C.G. Ingersoll, I.E. Greer, D.K. Hardesty, C.D. Ivey, J. Kunz, W.G. Brumbaugh, F.J. Dwyer, A. Roberts, T. Augspurger, C.J. Kane, R.J. Neves, and M.C. Barnhart. 2007. Chronic toxicity of copper and ammonia to juvenile freshwater mussels (Unionidae). Environmental Toxicology and Chemistry 26: 2048-2056.

Wang N.,C.G. Ingersoll, C.D. Ivey, D.K. Hardesty, T.W. May, T. Augspurger, A.D. Roberts, E. Van Genderen, and M.C. Barnhart. 2010. Sensitivity of early life stages of freshwater mussels (Unionidae) to acute and chronic toxicity of lead, cadmium, and zinc in water. Environmental Toxicology and Chemistry 29: 2053-2063. DOI:10.1002/etc.250

Wang, N., C.D. Ivey, C.G. Ingersoll, W.G. Brumbaugh, D. Alvarez, E.J. Hammer, C.R. Bauer, T. Augspurger, S. Raimondo, and M.C. Barnhart. 2017. Acute sensitivity of a broad range of freshwater mussels to chemicals with different modes of toxic action. Environmental Toxicology and Chemistry 36: 786-796.

Wang, N., C.D. Ivey, R.A. Dorman, C.G. Ingersoll, J. Stevens, E.J. Hammer, C.R. Bauer, and D.R. Mount. 2018. Acute toxicity of sodium chloride and potassium chloride to a unionid mussel (*Lampsilis siliquoidea*) in water exposures. Environmental Toxicology and Chemistry 37: 3041-3049.

[WMS]. Water Monitoring Solutions, Inc. 2018. 2018 Cypress Creek Basin Highlights Report: a summary of water quality in the Cypress Creek Basin in 2017. Prepared for Northeast Texas Municipal Water District. 53 pp.

Watters, G.T. 1996. Small dams as barriers to freshwater mussels (Bivalvia: Unionoida) and their hosts. Biological Conservation 75: 79-85.

Watters, G.T. 2000. Freshwater mussels and water quality: a review of the effects of hydrologic and instream habitat alterations. Proceedings of the First Freshwater Mollusk Conservation Society Symposium, 1999: 261-274.

Wehner, M.F., J.R. Arnold, T. Knutson, K.E. Kunkel, and A.N. LeGrande, 2017: Droughts, floods, and wildfires. Pp. 231-256 in Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock, eds.: Climate Science Special Report: Fourth National Climate Assessment, Volume I. U.S. Global Change Research Program, Washington, DC. doi: 10.7930/J0CJ8BNN

Whisenant, A. 2019. Email correspondence regarding BA Steinhagen mussel assessment. July 31, 2019. 8 pp.

Williams, J.D., A.E. Bogan, R.S. Butler, K.S. Cummings, J.T. Garner, J.L. Harris, N.A. Johnson, and G.T. Watters. 2017a. A revised list of the freshwater mussels (Mollusca: Bivalvia: Unionida) of the United States and Canada. Freshwater Mollusk Biology and Conservation 20:33-58.

Williams, L., N. Ford, M. Williams, and S. Rumbelow. 2017b. East Texas threatened mussels. Draft final report submitted to Texas Comptroller's Office. University of Texas - Tyler, Tyler, Texas. 43 pp.

Willner, S., A. Levermann, F. Zhao, and K. Frieler. 2018. Adaptation required to preserve future high-end river flood risk at present levels. Science Advances 4:eaao1914. 8 pp.

Wohl, E. 2015. Legacy effects on sediments in river corridors. Earth-Science Reviews 2015: 30-53.

Wolaver, B.D., C.E. Cook, D.L. Sunding, S.F. Hamilton, B.R. Scanlon, M.H. Young, X. Xu, and R.C. Reedy. 2014. Potential economic impacts of environmental flows following a possible listing of endangered Texas freshwater mussels. Journal of the American Water Resources Association 1-21. DOI: 10.1111/jawr.12171

Wright, B.H. 1896. New American Unionidae. Nautilus 9:133-135.

Wright, B.H. 1898. A new Unio from Texas. The Nautilus 12:93. Available at https://biodiversitylibrary.org/page/1746720

Wuebbles, D., G. Meehl, K. Hayhoe, T.R. Karl, K. Kunkel, B. Santer, M. Wehner, B. Colle, E.M. Fischer, R. Fu, A. Goodman, E. Janssen, V. Kharin, H. Lee, W. Li, L.N. Long, S.C. Olsen, Z. Pan, A. Seth, J. Sheffield, and L. Sun. 2013. CMIP5 climate 1 model analyses: climate extremes in the United States. Bulletin of the American Meteorological Society 95:571-583.

Yeager, M.M., D.S. Cherry, and R.J. Neves. 1994. Feeding and burrowing behaviors of juvenile rainbow mussels, *Villosa iris* (Bivalvia: Unionidae). Journal of the North American Benthological Society 13:217-222.

Appendix B. Cause and Effects Analysis for Louisiana Pigtoe and Texas Heelsplitter

Template for Cause and Effects Evaluation

THEME: ?			
[ESA Factor(s): ?]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	*What is the ultimate source of the actions causing the stressor? Ie, Urban Development, Oil and Gas Development, Agriculture*	See next page for confidences to apply at each step.	Literature Citations, with page numbers , for each step. Use superscript to delineate which statement goes with which citation. These can be repeated per theme, but not within a theme.
- Activity(ies)	*What is actually happening on the ground as a result of the action? Be specific here.*		
STRESSOR(S)	*What are the changes in environmental conditions on the ground that may be affecting the species? For example, removal of nesting habitat, increased temperature, loss of flow*		
- Affected Resource(s)	*What are the resources that are needed by the species that are being affected by this stressor? Or is it a direct effect on individuals?*		
- Exposure of Stressor(s)	*Overlap in time and space. When and where does the stressor overlap with the resource need of the species (life history and habitat needs)? This is not the place to describe where geographically it is occuring, but where in terms of habitat.*		
- Immediacy of Stressor(s)	*What's the timing and frequency of the stressors? Are the stressors happening in the past, present, and/or future?*		
Changes in Resource(s)	*Specifically, how has(is) the resource changed(ing)?*		
Response to Stressors: - INDIVIDUALS	*What are the effects on individuals of the species to the stressor? (May be by life stage)*		
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: - POPULATIONS [RESILIENCY]	*What are the effects on population characteristics (lower reproductive rates, reduced population growth rate, changes in distribution, etc)?*		
- GEOGRAPHIC SCOPE	*What is the geographic extent of the stressor relative to the range of the species/populations? In other words, this stressor effects what proportion of the rangewide populations?*		
- MAGNITUDE	*How large of an effect do you expect it to have on the populations?*		
SUMMARY	*What is the bottom line- is this stressor important to carry forward in your analysis, or is it only having local effects, or no effects?*		

This table of Confidence Terminology explains how we characterized our confidence levels in the cause and effects tables on the following pages.

Confidence Terminology	**Explanation**
Highly Confident	We are more than **90% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by documented accounts or research and/or strongly consistent with accepted conservation biology principles.
Moderately Confident	We are **70 to 90% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by some available information and/or consistent with accepted conservation biology principles.
Somewhat Confident	We are **50 to 70% sure** that this relationship or assumption accurately reflects the reality in the wild as supported by some available information and/or consistent with accepted conservation biology principles.
Low Confidence	We are less than **50% sure** that this relationship or assumption accurately reflects the reality in the wild, as there is little or no supporting available information and/or uncertainty consistency with accepted conservation biology principles. Indicates areas of high uncertainty.

Theme: Changes to water quality			
[ESA Factor(s): A, E]	**Analysis**	**Confidence / Uncertainty**	**Supporting Information**
SOURCE(S)	Population growth, human activity, and changing land uses are the drivers. Examples include Urban Development, Oil and Gas Development, Agriculture, confined animal feeding operations, etc.[1] Attoyac bacteria sources: on-site sewage facilities, wildlife, cattle, dogs, feral hogs, poultry litter, hunting camps, horses, and wastewater treatment facilities[2].	Highly confident	[1]Ford et al. 2014, p. 9. [2]Gregory et al. 2014, p. xii.
- Activity(ies)	Lost ecosystem functionality as forests and grasslands are denuded or converted for other uses. Increases in water demand for agriculture and human consumption results in increased groundwater pumping, reservoir construction, altered hydrology, and lower water quality from point and nonpoint sources. Pulp and paper mill effluent may contribute to absence of freshwater mussels near the mouth of Anacoco Bayou[1]. Oil extraction, WWTP effluent, and surrounding agriculture impact E TX rivers[2].	Highly confident	[1]Randklev et al. 2013b, p. 272. [2]Williams et al. 2017b, p. 17.
STRESSOR(S)	Heavy shell erosion observed in waters with pH = 5.6[1]. Erosion, lower streambank stability, and lower water quality, which includes a variety of potentially harmful constituents, such as changes to basic water chemistry (e.g., increase in temperature (which increases toxicity of many pollutants), increase in total dissolved solids/salinity (as measured by Conductivity), elevated ammonia and nitrogen, and low dissolved oxygen), persistent, bioaccumulative, and toxic substances such as pesticides and trace metals., and hormonally active compounds (i.e., emerging contaminants). Tanker truck and other transportation related spills can adversely effect water quality[3].	Highly confident	[1]Burlakova et al. 2012, p. 6. [2]Augspurger et al. 2003, p. 2569. [3]Jones et al. 2001.
- Affected Resource(s)	Watershed-level effects can occur, including loss of riparian habitat, increase in invasive species, lower biodiversity, altered stream functionality (changes to chemical, physical, and biological processes).	Moderately confident	
- Exposure of Stressor(s)	Contaminants from point and nonpoint sources, including hazardous spills, may affect water quality with magnitude varying by volume of discharge, dilution capacity of receiving waters, duration of exposure, life stage of mussel exposed, and whether stressor acts in isolation or simultaneously with other stressors that may compound the effects. Contaminants in water may be short-term acute exposures resulting in immediate mortality, or sub-lethal long-term exposures. Persistent, bioaccumulative and toxic (PBT) compounds may accumulate in sediments, resulting in sediment toxicity. Contaminants may also exert toxicity on host fishes and interfere with life cycle requirements of mussels. Sediment pore-water concentrations of NH_3 typically exceed that of the surface water[1].	Highly confident	[1]Augspurger et al. 2003, p. 2574.
- Immediacy of Stressor(s)	Varies by stream segment depending on point and nonpoint sources in the watershed, hydrology (e.g., frequency of low flow conditions), etc. This has happened in the past, is currently happening, and will continue to happen in the future. Although efforts under the CWA have generally improved water quality conditions in the U.S. compared to the mid-20th century post-industrial era, human population growth along with increasing demand for limited water resources, as well as increasing demand for wastewater disposal, continues to deteriorate remaining water resources.	Highly confident	

Changes in Resource(s)	Contaminants in water and sediment may inhibit mussel survival, growth and reproduction, or that of their host fishes.	Highly confident	
Response to Stressors: **- INDIVIDUALS**	May be sub-lethal, such as inhibiting growth or reproduction, or cause mortality of individuals. DNA damage occurs in the mussel *Unio pictorum* , when found downstream of paper mills and oil refineries[1]. Heavy metals may inhibit glochidial attachment[2]. Juveniles more susceptible to anthropogenic disturbances [3].	Highly confident	[1]D. Ford 2013, p. 4. [2]Arm et al. 2014, p. 114. [3]Ford et al. 2018, p. 14.
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: **- POPULATIONS** **[RESILIENCY]**	Will vary by nature and magnitude of local stressors, but capable of causing population declines, lowering resiliency, or even extirpation. Low levels of salinity can have dramatic effect on reproduction, physiology, and survival in *Elliptio complanata* [1].	Highly confident	[1]Blakeslee et al. 2013, p. 2853.
- GEOGRAPHIC SCOPE	Little River: freshwater mussel declines in Little R have been attributed to impoundments and degraded WQ from point source effluents, these impacts likely affect host fish thereby limiting recruitment[1]. Pulp and paper mill effluent may contribute to absence of freshwater mussels near the mouth of Anacoco Bayou[2]. Portions of the Calcasieu R impacted by paper mill wastes and sand mining[3]. Big Cypress CR is listed 303d for elevated Hg in fish tissues, low pH, and low DO. pH impairment may be removed by the state due to the standard being met since 2014; Little Cyprees Bayou listed for low DPO and elevated bacteria[4]. Large portions of the Trinity and Neches Rivers have legacy contamination, including PCBs and Dioxins (polychlorinated dibenzofurans and dibenzo-p-dioxins (PCDFs/PCDDs) in the Trinity, and mercury and dioxins in the Neches [5,7] Several lakes along the Sabine River have mercury contamination, including Hills Lake, Clear Lake, and Toledo Bend Reservoir (Panola County), as do Big Cypress Creek and Caddo Lake[6].	Highly confident	[1]Davidson et al. 2014, p. 1. [2]Randklev et al. 2013b, p. 272. [3]Vidrine 1993. [4]WMS 2018, pp. 16, 37. [5]TDSHS 2018. [6]TCEQ 2018a,b. [7]Perkin and Bonner 2016.
- MAGNITUDE	Attoyac Bayou: fecal coliforms often exceeded standards in the late 1990s and elevated ammonia levels were rountinley observed in 2008[1].	Highly confident	[1]Gregory et al. 2014, p. xi.
SUMMARY	Carry stressor forward. Not localized or isolated.	Highly confident	

THEME: Changes to hydrology (altered flow regime)			
[ESA Factor(s): A, E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Urban development[1], reservoir operation, agriculture (diversion, ground water extraction, etc.) and climate change[2] (flood/scour from very large rainfall events).	Highly confident	[1]Ford et al. 2014, p. 9. [2]Archambault et al. 2013, p. 230, 247.
- Activity(ies)	Hydroloelectric dam operations[1], out-of-basin water transfers. Climate change is likely to result in more extreme flooding and droughts and lead to changes in surface water, soil moisture, and groundwater[2].	Highly confident	[1]Davidson 2017, p. 3; N. Ford 2013, p. 3; Randklev et al. 2013b, p. 272. [2]Taylor et al. 2013, entire.
STRESSOR(S)	Altered flow regimes (more frequent peak flows, increased scouring in channel, loss of water due to pumping and out-of-basin transfers), inundation of habitat upstream of dams, decrease in water temperature down stream of hydroelectric dams[1].	Highly confident	[1]Randklev et al. 2013b, p. 272.
- Affected Resource(s)	Water temperature, stability of stream sediments, stability of stream banks, water availability, inundation of stream habitat.	Moderately confident	
- Exposure of Stressor(s)	Inundation occurs upstream of dam and seasonally with changing water levels, temperature effects of hydroelectric operations occur downstream of dam, altered flows due to dam operations primarily occurs downstream of dam, altered flows due to climate change, altered flows due to pumping and out-of-basin transfers[1].	Highly confident	[1]N.Ford 2013, p. 3; Ford et al. 2016, p. 47.
- Immediacy of Stressor(s)	Impacts from hydroelectric dams can be expected to continue. Water basin transfers are likely to increase in the future. Climate change effects are expected to intensify into the future.	Moderately confident	
Changes in Resource(s)	Unstable banks and substrates, reduction in water temperature downstream of hydroelectric dams, fluctuating water levels of impounded areas. Changes in flow rates and volume due to impoundments can cause scouring and deposition impacting mussels[1]. Overgrazing since mid-1800's caused loss of vegetative cover and soils, which allows runoff from precipitation to increase contributing to scouring in streams; also, changes in rainfall patterns to fewer light and moderate showers and longer periods of drought with heavy, damaging floods contribute to scouring impacts[2]. After inundation, flows are altered which can lead to increased sedimentation, organic material deposition, decreased oxygen levels due to lack of flow and increased oxygen demand due to decomposition, increase in water depth, a possible lack of suitable nutrients available to mussels that may impact reproduction[3].	Moderately confident	[1]Howells 1997, p. 32. [2]Howells 2010a, p. 9. [3]Neck and Howells 1995, p. 14.

Response to Stressors: **- INDIVIDUALS**	High shear stress dislodges small, lightweight juveniles from the substrate without displacing the heavier adults[1]. Oxbows and tribs provide refugia from main channel high flows (BA Steinhagen releases)[2]. Excysted juveniles dispersal distance influenced by the magnitude of velocity and velocity gradients[3]. Individuals deposited downstream will likely die. Those smothered with deposited sediment will die.	Highly confident	[1]Bakken 2013, p. 5. [2]N. Ford 2013, p. 10. [3]Daraio et al. 2012, p. 601.
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses?* *And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: **- POPULATIONS** **[RESILIENCY]**	Diversity and abundance are negatively impacted by hydropower generation in the lower Sabine River due to altered flow, temperature, and sediment regimes; sinuosity and connectivity with the floodplain may lessen these impacts[1].	Highly confident	[1]Randklev et al. 2014, pp. 9-10.
- GEOGRAPHIC SCOPE	Entire extent of range. 20km downstream of Pine Creek Dam and Broken Bow Dam[1]. In TX, negative correlation between human population density and the proportion of rare species in the watershed[2]. Flow variability likely accounts for 14% of the variability in mussel community composition[3]. Substrate scouring occurred in the uppermost Sabine R mussel sanctuary due to highwater releases from LK Tawakoni, beds only found in mid and lower sanctuaries[4]. Impoundments constructed in the early 1900s on East FK, Elm FK, West FK, and Clear FK Trinity River may have acutely impacted mussel distribution in the DFW area, compounded by other anthropogenic impacts[5]. Discharge below Toledo Bend Dam is high pulsed during periods of power generation[6]. CR- Anthropogenic hydrologic alteration is prevelant throughout the entire range of Louisiana pigtoe and Texas heelsplitter. These systems are impacted by mainstem reservoirs, tributary reservoirs, and surfacewater and groundwater extraction. The magnitude of the impacts of these flow alterations varies by type, with impacts being localized (small weir) or impacting many river miles (large hydropower reservoir).	Highly confident	[1]Davidson 2017, p. 10. [2]Burlakova et al. 2011b, p. 403. [3]Dascher et al. 2017, p. 3. [4]Ford et al. 2009, p. 290-291. [5]Randklev 2011, pp. 36-39. [6]Randklev et al. 2011, p. 3.
- MAGNITUDE	The overall range, and distribution of Louisiana Pigtoe and Texas Heelsplitters have been significantly altered due to hydrological alterations, including large hydropower operations. These effects of current and historical stressors will persist[1].	Highly confident	[1]Haag 2012, p. 328-330.
SUMMARY	Affects all populations. Carry stressor forward, combine with Hydrology (low flow). These reservoirs act as large scale barriers that isolate populations, prevent host fish movements, and preclude genetic exchange.	Highly confident	

THEME: Changes to hydrology (inundation, low flow conditions)			
[ESA Factor(s): A, E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Dams, drought, pumping/groundwater extraction, potential climate change	Highly confident	
- Activity(ies)	Municipal and agricultural water demands - Reservoirs throughout the range of both species provided municipal water supply. Southern Neches River water extraction for rice and crawfish farming. Flood Control - Reservoirs in Neches Basin (other?) limit number and severity of pulse flows. Climate Change - May result in periods of extreme drought thus reducing surface flows. In one East Texas reservoir [likely BA Steinhagen] that was brought down by 2 m (~6.5 ft) every second year, stranded individuals (Texas Heelsplitter) were observed burrowing into sand and mud substrates or following the declining water line[1].	Highly confident	[1]Howells 2010b, p. 3.
STRESSOR(S)	Loss of flow due to drought reduces the amount of available habitat as a result of narrowing stream bed[1]. Loss of flow results in habitat degradation from lack of pulse flows, increase in fine sediment[38], DO reduction[156], increased water temperatures[7], increased contaminant exposure[4], ammonia[9], stranding of individuals[2], increased exposure to predation[1], and reduction of nutrients into system.	Highly confident	[1]Golladay et al. 2004, p. 501, 503; Haag and Warren 2008, p. 1172-1173. [2]Howells 2010b, p. 3. [3]Box and Mossa 1999, p. 100. [4]Augspurger et al. 2007, p. 2025. [5]Sparks and Strayer 1998, pp. 132–133. [6]Johnson 2001. [7]Pandolfo et al. 2010. [8]Kondolf 1997, pp. 535, 548.
- Affected Resource(s)	Adequate water quality, wide stream bed, substrate enhancement, cover from predation	Highly confident	
- Exposure of Stressor(s)	Low flows result in reduction in anchoring habitat for adults and juveniles, documented predator access to adults and juveniles cover from predation[1]. Low flows are important for reproduction (egg fertilization, host fish/mussel interaction, juvenile anchoring, glochidia niche).	Highly confident	[1]Thorp and Covich 2010.
- Immediacy of Stressor(s	Reservoirs in these stream systems for decades. Managed water releases from dams presently result in extended periods of low flow. Likely to continue into the future without release strategies. Effects of climate change are only expected to increase into the future as droughts become more frequent and air temperatures increase, resulting in more surface water extraction and additional water demands arise.	Moderately confident	

Changes in Resource(s)	Low flows have reduced available habitat to narrow stream beds, stranded individuals, increased exposure to predation. Additionally may lower fitness or cause mortality due to reduced water quality.	Highly confident	
Response to Stressors: - INDIVIDUALS	Byssus production in juvenile *Lampsilis* was more impacted by low flow (drought) regime with 93-99% reduction when compared to the watered regime[1]. Glochidia survival affected by increased water temp[2]. Mortality. Sub-lethal effects[3]	Highly confident	[1]Archambault et al. 2013, p. 236, 244. [2]Pandolfo et al. 2010, pg. 961 - 963. [3]Gagnon et al. 2004, p. 675.
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: - POPULATIONS [RESILIENCY]	Populations may be reduced or eliminated as a result of habitat loss, reduction in breeding age adults from predation, lowered fitness or mortality of all lifestages due to water quality/contaminants. Thermal stress associated with low water levels -> observed declines in abundance and species richness[1].	Moderately confident	[1]Galbraith et al. 2010, p. 1180.
- GEOGRAPHIC SCOPE	Toledo Bend tributaries/Sabine R low flow/drought in 2010-2011, many freshwater mussel populations in Sabine National Forest were dewatered[1]. B.A. Steinhagen Reservoir drawdown decimated the largest known *P. amphichaenus* population in 2006 through 2007[2]. This reoccured in 2019. LA-Calcasieu, Vermillion, Mermentau, and lower Sabine: Increased water extraction during low rainfall periods due to agricultural practices[3].	Moderately confident	[1]Arnold et al. 2013, p. 24. [2]Howells 2010b, p. 11, 16. [3]Kelso et al. 2011, p. 14.
- MAGNITUDE	High E TX rivers are large enough and rainfall is more consistant minimizing impacts due to inadequate flow resulting from a lack of precipitation[1]. Projected TX population growth from 2010 to 2060 is 80% (25 to 46 million), water demand increase of ~20%, water supply decrease up to 10% due to groundwater depletion[2].	Somewhat confident	[1]Williams et al. 2017b, p. 17. [2]Wolaver et al. 2014, p. 1081.
SUMMARY	As low flows are mostly the result of dams and have altered the natural flow regimes range wide for both and can be exacerbated by drought. Affects all populations. Carry stressor forward, combine with Hydrology (flow changes).	Highly confident	

THEME: Changes to substrate (sedimentation)			
[ESA Factor(s): A, E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Tributary and streambed scouring, streambank erosion, land development and resulting erosion of uplands brought by runoff	Highly confident	
- Activity(ies)	Land use changes[1], urbanization, hydrological modifications	Highly confident	[1]Arbuckle and Downing 2002, p. 311; Arm et al. 2014, p. 45.
STRESSOR(S)	Filling in of substrate, smothering, toxicity from contaminants bound to substrate particles[1].	Highly confident	[1]D. Ford 2013, p. 2., Allen and Vaughn 2010, p. 383.
- Affected Resource(s)	Direct smothering of individuals, changing suitability of anchoring habitat, reducing feeding of juveniles[1]. Also coarse gravel, cobble moving through system changing habitat. Contaminants bound to substrate particles compromising metabolic processes.	Highly confident	[1]D. Ford 2013, p. 2.
- Exposure of Stressor(s)	Juvenile and adults living in substrate, potential loss of host fish use of habitat[1].	Highly confident	[1]D.Ford 2013, p. 2.
- Immediacy of Stressor(s)	Historical, current, future. After high flow events, as water velocity decreases, particles drop to substrate.	Highly confident	
Changes in Resource(s)	Summary statement- fine and coarse sediment moving through system, changing habitat for existing pops and preventing recruitment. Freshwater mussels require a stable environment due to limited mobility and age of sexual maturation[1]. Observed localized siltation on mussel beds due to riparian clearing[2].		[1]Randklev et al. 2014, p. 9. [2]Galbraith et al. 2010, p. 1181.
Response to Stressors: - INDIVIDUALS	Clogged gills, reduced fitness and growth rates, mortality, recruitment failure, changed host fish interactions.	Highly confident	[1]Sparks and Strayer 1998, p. 129. [2]Box and Mossa 1999, pp. 99, 100.

POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: - POPULATIONS [RESILIENCY]	Substrate changes can result in population level response- these changes are not highly localized. Can affect recruitment, population growth rates, etc. Poor substrate quality can lead to low resiliency[1].		[1]Allen and Vaughn 2010. p. 390.
- GEOGRAPHIC SCOPE	Large mussel bed downstream of BA Steinhagen covered by shifting sands due to altered flows by dam operations[3]. Large percentage of the mid sanctuary (Sabine R) had severve erosion and numerous bankfalls[4]. S. Sulphur R was realigned and channelized, N. Sulphur channelized in 1920, sedimentation occurring throughout Sulphur R drainage[5]. Sandy soils in east TX are subject to any distrubance of natural cover resulting in extensive erosion and increased deposition in streams[6]. Sabine R below Toledo Bend Reservoir - prevalent substrate is sand[7]. Diversity and abundance are negatively impacted by hydropower generation in the lower Sabine R. due to altered flow, temperature, and sediment regimes; sinuosity and connectivity with the floodplain may lessen these impacts[8]. Lower parts of Calcasieu R and Sabine R heavily impacted; "increased sedimentation resulting from erosion in adjacent riparian and upland habitats is a common characteristic of virtually all streams in southern Louisiana."[9]. Portions of the Calcasieu R impacted by paper mill wastes and sand mining[10]. The mainstem of the Trinity, Neches, and Sabine rivers are on the 303(d) list for contaminants. There is potential for these contaminants to impact survival, growth, and reproduction in mussel communities in these systems[11]. In addition to substrate contamination, substrate scouring from increased high flow events in the Trinity River is impacting bank stability and sediment along bank habitats where TH occurs[12]. In addition, mainstem reservoirs in the Sabine River have caused downstream declines in mussel richness and abundance; one factor for these declines could be changes in sediment dynamics[13][14]	**Moderately Confident**	[3]N. Ford 2013, pp. 9-10. [4]Ford et al. 2009, p. 282. [5]Heffentrager 2013, p. 4-5. [6]Howells 1997, p. 31. [7]Karatayev and Burlakova 2008, p. 24. [8]Randklev et al. 2014, pp. 9-10. [9]Kelso et al. 2011, pp. 11-12. [10]Vidrine 1993. [11] TCEQ 2018a. [12]Randklev et al. 2017b, p. 5. [13]Randklev et al. 2015, p. 16. [14]Ford et al. 2009, p. 290.
- MAGNITUDE	Sediment accumulation is a pervasive problem throughout the range of Louisiana Pigtoe and Texas Heelsplitters.	Highly confident	[1]N. Ford 2013, p. 10.
SUMMARY	Sediment accumulation in the substrates occupied by Louisiana Pigtoe and Texas Heelsplitters has reduced habitat availability for both species historically and is expected to continue. Conversely, high flows (e.g. flooding) have scoured mussel habitat and resulted in bank collapse. These stressors will be carried forward in our analysis of future conditions of the species.	Moderately Confident	

THEME: Invasive species			
[ESA Factor(s): C]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Zebra Mussels, Giant Salvinia, Asian Clam Direct competition for resources with Louisiana Pigtoe and Texas Heelsplitter from invasive Zebra Mussels. Habitat alterations from non-native aquatic plants (i.e. Giant Salvinia, Hydrilla, etc…)	Somewhat confident	Howells 2010a, p. 13 Howells 2010b, pp. 14-15 Kaller et al. 2007, pp. 173-174
- Activity(ies)	Zebra Mussels are present in the Trinity and Red River basins in Texas (other river basins in the other states as well) and there is potential for them to continue to spread to other river basins, or further expansion within basins they are currently present. Aquatic invasive plant species are prevelant throughout the range of Louisiana Pigtoe and Texas Heelsplitter.	Somewhat confident	Howells 2010a, p. 13 Howells 2010b, pp. 14-15 Kaller et al. 2007, pp. 173-174
STRESSOR(S)	Hydrilla and Giant Salvinia can become too dense for mussels to use lake habitats and alter water quality.	Somewhat confident	
- Affected Resource(s)	Dissolved oxygen reduced due to blocked sunlight and decomposition of plant matter[1]. Asian Clam die-offs can cause water column ammonia to increase to levels that could impact native mussels[2][3].	Somewhat confident	[1]TPWD 2018. [2]Cherry et al. 2005, p. 378; [3]Cooper et al. 2005, p. 392.
- Exposure of Stressor(s)	Zebra mussels and exotic macrophytes prefer lacustrine, backwater, and very low flow areas in east Texas. Asian clam is most sucessful in flowing water[1].	Somewhat confident	[1]Howells 2014, p. 125.
- Immediacy of Stressor(s)	Timing and frequency of invasive threats existing today are likley to increase in severity over time due to climate change impacts.	Somewhat confident	
Changes in Resource(s)	Resource competition, degradation of habitat, increased predation.	Somewhat confident	
Response to Stressors: **- INDIVIDUALS**	Reduced food quality due to zebra mussels being more efficient at sorting food particles[1].	Somewhat confident	[1]Baker and Levinton 2003, p. 103.
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: **- POPULATIONS** **[RESILIENCY]**	What are the effects on population characteristics (lower reproductive rates, reduced population growth rate, changes in distribution, etc)?	Somewhat confident	

- GEOGRAPHIC SCOPE	Physical evidence of Zebra Mussels not found in ETX rivers[1]. Mill CR, LA: Given that hogs spend considerable time near aquatic resources[2] and appear to contribute E. coli into streams, we believe that it is logical that the previously measured high fecal coliform counts in the Mill Creek watershed[36] were probably the result of the large numbers of feral and free-ranging hogs rather than deer, turkeys, beavers, horses, or other potential sources. The DNA data potentially implicate feral hogs as the primary source of fecal coliforms that were negatively associated with freshwater mussels and important nutrient processing insects in the Mill Creek watershed[5]. Feral hogs appear to decrease freshwater mussel (members of the family Unionidae commonly known as pearly mussels) diversity and abundance by creating organic enrichment and changes in microbial community composition[5]. Feral hogs may compound existing perturbations leading to further declines or localized extirpation[6]. Invasive aquatic species are prevalant throughout the range of Louisiana Pigtoe and Texas Heelsplitter. Reservoir habitats currently appear to be disproportionately affected by these aquatic invasice species so they may impact Texas Heelsplitter populations disproportionately. Feral hogs destabilize banks, leading to further erosion and sloughing during high flows[7].	Somewhat confident	[1]Ford et al. 2016, p. 47. [2]Mersinger and Silvy 2007, p. 165. [3]Kaller and Kelso 2003. [5]Kaller and Kelso 2006. [6]Kaller et al. 2007, pp. 173-174. [7]Timmons et al. 2011, entire.
- MAGNITUDE	Invasives are present throughout the range, though more severe in southern portions.	Somewhat confident	
SUMMARY	Invasives compete directly with mussels for space and food resources, can impact water quality, degrade habitat.	Somewhat confident	

THEME: Direct mortality			
[ESA Factor(s): A, C, E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	For Louisiana pigtoe, 1) human collection for fish bait[1], 2) human collection for scientific purposes, 3) dewatering/ dessication, 4) WQ/acute & chronic toxicity. Texas heelsplitter is difficult to find and rarely collected in the field so direct mortality by collection is not known to be a stressor for the species; however, dewatering/desiccation and WQ/toxicity are likely stressors. Mammalian predation.	Highly Confident	[1]Orsak, personal observation, Little River OK, May 2018.
- Activity(ies)	1-4) direct mortality. Decresasing stream flows result in increasing predation by terrestrial predators due to increased access, reducing populations. Collection and sampling of both species at known sites can impact population sizes.	Highly Confident	
STRESSOR(S)	**Collection**. Increased temperature and loss of flow may contribute to conditions that allow collection; increased concern regarding the status of mussels has led to increased interest in research, increased funding for studies and increased collection for science, although impacts are thought to be minor. Collection for fish bait is believed to be localized but may impact affected populations heavily in areas popular for recreational sports. **Predation** on freshwater mussels is a natural ecological interaction. Raccoons, river otters, snapping turtles, and fish are known to prey upon east Texas mussels. Under natural conditions, the level of predation occurring within populations is not likely to be a significant risk to that population. However, during periods of low flow, terrestrial predators have increased access to portions of the river that are generally too deep under normal flow conditions. Muskrats and raccoons are known to prey upon live mussels, as evidenced by freshly fragmented valves scattered along vegetated riverbank margins.	Somewhat confident	
- Affected Resource(s)	Direct on Individuals but currently disease and predation do not appear to be problematic to *P. riddellii* [1] or *P. amphichaenus* [2].	Somewhat confident	[1]Howells 2010a, p. 12. [2]Howells 2010b, p. 13.
- Exposure of Stressor(s)	**Collection.** Scientific collection is not thought to be widespread; the extent to which collection for fish bait occurs across the range is unknown. **Hydrology.** Dewatering/desiccation will vary by season, watershed, and climatic conditions. **Water quality** and risk of acute or chronic toxicity will vary by location and watershed, including land uses and proximity of mussel beds to sources of point and nonpoint pollution. Age of exposure will also affect toxicity, with early life stages being most vulnerable. **Predation.** As stream flows decline, access by terrestrial predators increases, increasing predation rates by raccoons[1] and muskrats[2]. Adults are more susceptible to predation and collection than juveniles, as they are larger and easier to find.	Somewhat confident	[1]Walters and Ford 2013, p. 479. [2]Golladay et al 2004, p. 503.

- Immediacy of Stressor(s)	Mortality of Texas heelsplitter due to predation have been observed during low flow periods. Raccoons have preyed on individual Texas heelsplitters stranded by low waters or deposited in shallow water or on bars following flooding or low water periods[1]. As drought and low flow are predicted to occur more often and for longer periods due to climate change, populations are expected to experience additional predation pressure in the future.	Somewhat confident	[1]Walters and Ford 2013, p. 479.
Changes in Resource(s)	Depredated and collected mussels removed from breeding population, thus reducing current and potential future number of individuals.	Highly confident	
Response to Stressors: - INDIVIDUALS	Removal from population and loss of breeding potential	Highly Confident	
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: - POPULATIONS [RESILIENCY]	Loss of individuals of both species from combined effect of collection, dewatering/desiccation, predation, water quality will result in populations possessing less resiliency and increasing risk of extirpation. Future models predict decreasing water volumes/stream flows thus exacerbating current trends.	Highly Confident	
- GEOGRAPHIC SCOPE	Mammalian predation at ponds on Camp Maxey was as high as 19%, exacerbated by fluctuating water levels[1]. Sabine R upstream of Toledo Bend: 58 of 79 recently deceased *P. amphichaenus* exhibited signs of predation, all were <100mm in length, raccoon suspected[2]. While the threat of direct mortality extends throughout the entire range of *P. riddellii* and *P. amphichaenus*, these specific stressors are limited to specific areas.	Moderately confident	[1]Burlakova and Karatayev 2007, p. 291. [2]Walters and Ford 2013, pp. 479-480.
- MAGNITUDE	Predation/collection is an exacerbating factor on populations already under pressure from various other stressors. Could potentially impact some populations reducing resiliency.	Somewhat confident	
SUMMARY	Mortality from dewatering events and resulting dessication are expected to continue for some populations of Texas Heelsplitter in reservoirs and Louisiana Pigtoe in agricultural and/or municipal canal systems. Reduced stream flows in the future would expose both species to increases in predation. These effects will be carried forward in our analysis of effects to Louisiana Pigtoe and Texas Heelsplitters into the future. Collection for Louisiana Pigtoe is localized and could effect populations and is thus carried forward. Collection for Texas Heelsplitter is not expected to have an effect due to its rarity, thus it is not carried forward.	Moderately confident	

THEME: Fragmentation			
[ESA Factor(s): A, E]	**Analysis**	Confidence / Uncertainty	Supporting Information
SOURCE(S)	Impoundments, transportation structures, dewatered stream segments.	Highly Confident	
- Activity(ies)	Dam construction, flood control, low-water crossings, reduced flow resulting in barrier to movement	Highly Confident	
STRESSOR(S)	These activities result in deep impounded waters reducing available streambed habitat[2] as well as function as barriers to host fish movement/disperal upstream and potentially downstream of mussel populations thereby isolating populations[12]. Impoundments can significantly decrease genetic variability in mussel populations overtime, rate of decrease influenced by life span[3]. Isolated populations are susceptible to genetic drift (change of gene frequencies in a population over time) and inbreeding depression which may cause death, reduced fertility, reduced fitness and morphological chromosomal abnormalities.	Highly Confident	[1]Watters 1996. p. 83. [2]Newton et al. 2008, p.430. [3]Abernathy et al. 2013, p.25; Fuller and Doyle 2018, p. 1445-1446..
- Affected Resource(s)	Dam construction fragments the range of *P. riddellii* , leaving remaining habitats and populations isolated by the structures as well as by extensive areas of deep uninhabitable, impounded waters. Dams impound river habitats throughout almost the entire range of the species, and these impoundments have left isolated patches of remnant habitat between impounded reaches. While *P. amphichaenus* inhabitats reservoirs as well as streams, historically the species only occurred in streams and sloughs as lakes/reservoirs did not occur naturally within its range[1].	Highly Confident	[1]Howells 2014, p. 69.
- Exposure of Stressor(s)	Impounded Water - permanent stream bed habitat loss to adults and juvenile *P. riddellii* . Barriers - permanently precludes movement of adults, juveniles, glochidia and host fish, thereby isolating populations.	Highly Confident	
- Immediacy of Stressor(s)	Reservoirs (impounded water plus barrier) have historically and currently acted as stressors upon these species. Existing reservoirs will continue to act as stressors into the future and proposed new reservoirs could exacerbate current conditions. Impacts from these stressors occur in the recent past, present, and expected to continue into the future.	Highly Confident	
Changes in Resource(s)	As existing populations are isolated from one another, genetic exchange between populations has been eliminated and any populations that may be extirpated through stochastic events will not be naturally recolonized.	Moderately Confident	
Response to Stressors: - INDIVIDUALS	Habitat fragmentation acts on the population level. Individuals are unaffected.	Highly Confident	
POPULATION & SPECIES RESPONSES	*[Following analysis will determine how do individual effects translate to population and species-level responses? And what is the magnitude of this stressor in terms of species viability?]*		
Effects of Stressors: - POPULATIONS [RESILIENCY]	Reduced range/distribution due to stream bed loss, lack of gene flow between fragmented populations with the potential for genetic drift and/or inbreeding depression.	Highly Confident	

- GEOGRAPHIC SCOPE	Watersheds throughout the entire range of both species have been fragmented by large dams, reservoirs and smaller barriers[1].	Highly Confident	[1]Randklev et al. 2016.
- MAGNITUDE	Population fragmentation due to barriers to fish movement has occurred for both mussel species historically. Currently and into the future, this fragmentation reduces the ability of all populations to rebound from stochastic events.	Highly Confident	
SUMMARY	Fragmentation severes gene exchange through a river system, prevents recolonization upstream of barriers, reduces available habitat, and increases the number of isolated populations. Additional barriers associated with water development are proposed within the range of both species. Carry forward in analysis.	Highly Confident	

Appendix C. Discussion of future scenario model forecasts, evaluation criteria and future condition tables for Louisiana Pigtoe and Texas Heelsplitter

Appendix C – Discussion of future scenario model forecasts, evaluation criteria and future condition tables for Louisiana Pigtoe and Texas Heelsplitter

Texas Heelsplitter

The range of the Texas Heelsplitter is currently represented by five focal areas within three river basins: the Sabine River/Toledo Bend population in the Sabine River basin; the Neches River/B.A. Steinhagen reservoir and Lower Neches River populations in the Neches River basin; and Grapevine Lake and the Trinity River/Lake Livingston in the Trinity River basin.

Sabine Basin

Sabine River/Toledo Bend Reservoir Focal Area – Current Condition

The functionally extirpated Sabine River/Toledo Bend focal area (Table C.2.14) is expected to remain extirpated in the next 50 years in all future scenarios. Two segments within the focal area are on the 303(d) list of impaired waterbodies for bacteria. A new poultry processing plant has been permitted to release wastewater in the upper portion of the focal area downstream of Lake Tawakoni. Wastewater releases are permitted at 2.18 million gallons per day with an ammonia limit of 3.94 mg/L, which is beyond the threshold for freshwater mussel tolerances. Water quality degradation is expected as result, despite wastewater dilution from mixing with stream flow. The construction of Lake Tawakoni and Toledo Bend Reservoir has impacted natural hydrologic conditions and dam releases causing substrate scouring eliminating mussel habitat downstream until sheer stress dissipates. An additional off-channel reservoir in the middle of the focal area and a water diversion project are proposed to meet future water demand. When constructed, water quality and hydrologic conditions would further degenerate from current conditions. Bank erosion is prevalent throughout the focal area, resulting in elevated inputs of sediment impacting suitable substrates for mussel beds.

Sabine River/Toledo Bend Reservoir Focal Area - Moderate Increase in Stressors

In the **Moderate 10-year scenario (**Table C.2.2), the focal area is projected to endure a moderate decline in water quality due to degradation resulting from a general increase in point and non-point source discharges, including significant wastewater effluent flows from a new poultry processing plant into a portion of the river with documented mussel beds. This degradation in water quality is expected to negatively influence overall mussel survival and reproductive success, potentially affecting both mussel and host fish movement, and subsequently causing fragmentation of suitable habitat. In some cases, water quality degradation may result in increased direct mortality of Texas Heelsplitter, contributing to a moderate decline in this focal area. Changes to hydrology, substrate and invasive species are expected to maintain their current condition of moderate decline. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, substrate, fragmentation, and direct mortality are expected to continue in tandem with population growth and associated impacts (e.g., habitat loss, increased demand for water supply, and increased generation of wastewater). Declining conditions of water quality, fragmentation, and direct mortality would be exacerbated by the effects of climate change. The moderate decline in hydrology is expected, in part, from future predicted reductions in flow, as represented by reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate (Lafontaine et al. 2019, entire). Subsequently, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. The threat posed by invasive species is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, substrate, fragmentation, and direct mortality will continue due to the threats described above. Hydrology is expected to severely decline due to climate change, including significant reductions in 7-day minimum and summer minimum base flows, as well as the construction of an off-channel reservoir in the middle Sabine River basin; these changes to hydrology and flow will further degrade water quality. Threats from invasive species are expected to maintain current condition. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected severe decline in population resiliency (Table C.2.7).

Sabine River/Toledo Bend Reservoir Focal Area - Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8), we anticipate moderate declines in water quality in this focal area due to degradation in general, and specifically resulting from discharges of effluent from a new poultry processing plant. This degradation in water quality is expected to negatively influence overall mussel survival and reproductive success, potentially affecting both mussels and host fish movement, and subsequently causing fragmentation of suitable habitat. In some cases, water quality degradation will result in increased direct mortality of Texas Heelsplitter, contributing to a moderate decline in this focal area. Changes to hydrology, substrate and invasive species are expected to maintain their current condition. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, substrate, fragmentation, and direct mortality continue. Conditions of water quality, fragmentation and direct mortality would be exacerbated by the same stressors described above. A moderate decline in substrate condition is expected as sediments accumulate on mussel beds from a lack of adequate cleansing flows. This change in substrate condition is correlated with an expected severe decline in hydrological condition from reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate in addition to an off-channel reservoir constructed in the middle of the Sabine River basin; these changes to hydrology and flow will further degrade water quality. Invasive species condition is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected severe decline in population resiliency (Table C.2.11).

In the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality and hydrology are anticipated resulting from increasing demands for waters supply and increasing point and non-point source pollution. Changes to flow include an estimated 30% reduction in minimum base flows as well as the construction of an off-channel reservoir in the middle Sabine River basin. Moderate declines in substrate, fragmentation, and direct mortality are anticipated from the same sources described in the Severe 25-year scenario. Invasive species condition is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled Texas Heelsplitter habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected severe decline in population resiliency (Table C.2.13).

Neches Basin

Neches River/B.A. Steinhagen Focal Area – Current Condition

The Neches River/B.A. Steinhagen focal area currently has a low probability of persistence. Tributaries and segments of the focal area are on the 303(d) impaired water bodies list for dioxin and mercury in edible tissue, bacteria, and depressed dissolved oxygen. Numerous segments had concerns for nutrients, particularly ammonia and total phosphorus; however, decreasing trends for these parameters were often observed. Stream flows are influenced by Lake Palestine in the upper portion of the focal area and B.A. Steinhagen in the southern portion of the focal area. Drawdowns of B.A. Steinhagen resulted in direct mortality of Texas Heelsplitter in 2006 through 2007, and again in 2019.

Neches River/B.A. Steinhagen Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario (**Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, substrate, fragmentation, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Sediment accumulation on mussel beds is projected to increase from a lack of adequate cleansing flows. The proposed Rockland Reservoir on the main channel of the Neches River, which would function as a fish passage barrier, is anticipated to be operational at this time-step. Direct mortality is expected to increase due to water quality degradation, reductions in water volume, and habitat loss from reservoir construction. A severe decline in hydrology is attributed to three proposed water delivery projects within the focal area combined with an overall reduction in stream flows. Lake Columbia is an off-channel reservoir proposed in the upper portion of the focal area, a run-of river water diversion is proposed for the middle of the focal area, and Rockland Reservoir is proposed near the downstream end of the focal area. Additionally, reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate are expected. The invasive species factor is expected to maintain current condition. The projected moderate and severe decline in habitat and population factors (i.e., water quality and quantity, fish host availability, reproduction/recruitment, occupied habitat, and abundance) is expected to result in a severe decline in population resiliency (Table C.2.5). Low population condition is anticipated during this time-step.

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, substrate, fragmentation, and direct mortality are expected to continue as the threats discussed in the Moderate 25-year scenario are realized and exacerbated by further reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate. Severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors continue, as well as declines to habitat factors, contributing to a projected severe decline in population resiliency (Table C.2.7); however, not to the point of extirpation. Low population condition is expected during this time-step.

Neches River/B.A. Steinhagen Focal Area – Severe Increase in Stressors

Change from the current low population condition is not expected as no changes to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area is projected to maintain its current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, substrate, fragmentation, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural steam flows with some increases to municipal wastewater effluent return flows). Sediment accumulation on mussel beds is projected to increase from a lack of adequate cleansing flows. The proposed Rockland Reservoir on the main channel of the Neches River, which would function as a fish passage barrier, is anticipated to be operational at this time-step. Direct mortality is expected to increase due to water quality degradation, reductions in water volume, and habitat loss from reservoir construction. A severe decline in hydrology is attributed to three proposed water delivery projects within the focal area combined with an overall reduction in stream flows. Lake Columbia is an off-channel reservoir proposed in the upper portion of the focal area, a run-of river water diversion is proposed for the middle of the focal area, and Rockland Reservoir is proposed near the downstream end of the focal area. Additionally, reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate are expected. The invasive species factor is expected to maintain current condition. The projected moderate and severe decline in habitat and population factors (i.e., water quality and quantity, fish host availability, reproduction/recruitment, occupied habitat, and abundance) is expected to result in a severe decline in population resiliency (Table C.2.11). Low population condition is anticipated to continue during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), moderate declines in substrate, fragmentation, and direct mortality are expected to continue as the threats discussed in the Severe 25-year scenario are realized and exacerbated by further changes to hydrology, including reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate. Both water quality and quantity undergo a severe decline as summer minimum base flows are projected to decrease by 30% from present levels (Lafontaine et al. 2019, entire), in addition to the other water volume reductions considered in the Severe 25-year scenario. Severe declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population factors, as well as declines to habitat factors, contribute to a continuing severe decline in population resiliency (Table C.2.13). Extirpation is expected during this time-step.

Lower Neches River Focal Area – Current Condition

The Lower Neches River focal area currently has a low current condition/probability of persistence. Tributaries and segments of the focal area are on the 303(d) impaired water bodies list for dioxin and mercury in edible tissue. Numerous segments had concerns for bacteria, impaired habitat, and nutrients, particularly ammonia and total phosphorus; however, decreasing trends for these parameters were often observed. Stream flows are influenced by B.A. Steinhagen Reservoir in the upper portion of the focal area. Substrates below the reservoir are subjected to sheer stress from water releases, causing shifting sediments to impact mussel beds. No impoundments are proposed within the focal area.

Lower Neches River Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3) and low population condition.

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Hydrologic impacts related to climate change, including a reduction in 7-day minimum flows and summer minimum base flows, are expected. Direct mortality is expected to increase as a result of these and other changes to habitat factors. Substrate, fragmentation and invasive species continue to maintain current condition. The moderate declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulted in a projected moderate decline in population resiliency (Table C.2.5). Low population condition is anticipated during this time-step.

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and direct mortality continue due to the same sources described in the Moderate 25-year scenario. Hydrologic alterations driven by climate change will experience a severe decline due to further reductions in 7-day minimum flows and summer minimum base flows. Substrate, fragmentation and invasive species continue to maintain current condition. The moderate declines in water quality and direct mortality combined with the severe decline in hydrology habitat factors, along with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulted in a projected moderate decline in population resiliency (Table C.2.7). Extirpation is expected during this time-step.

Lower Neches River Focal Area – Severe Increase in Stressors

Change from the current low population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency and low population condition (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, hydrology, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Hydrologic impacts related to climate change, including reductions in 7-day minimum and summer minimum base flows, are expected. Direct mortality is expected to increase as a result of these and other changes to habitat factors. Substrate, fragmentation and invasive species continue to maintain current condition. The moderate declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance), resulted in a projected moderate decline in population resiliency (Table C.2.11). Low population condition is anticipated during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), both water quality and hydrology undergo severe decline as ongoing water quality degradation is exacerbated by a greater than 30% reduction in 7-day minimum flows and summer minimum base flows from present-day levels. Direct mortality is expected to continue in moderate decline as a result. Substrate, fragmentation and invasive species continue to maintain current condition. The focal area is projected to experience severe declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) (Table C.2.13). Since two of the three habitat factors are in severe decline, the focal area is expected to experience a severe decline in population resiliency resulting in extirpation during this time-step.

Trinity Basin

Grapevine Lake Focal Area – Current Condition

The Texas Heelsplitter population in the Grapevine Lake focal area is currently considered functionally extirpated and is expected to remain so over the next 50 years in all future scenarios. Lake Grapevine functions as a local water supply source and receives municipal wastewater is discharges. The focal area is on the 303(d) impaired water bodies list for pH. The aquatic invasive zebra mussel has been documented in the lake.

Grapevine Lake Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in population factors (i.e., host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, direct mortality, and invasive species are anticipated. Water quality degradation is expected as Grapevine Lake is in a highly urbanized area and thus will be subjected to general increasing point and non-point source pollution with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels. These water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Lake elevation is expected to fluctuate due to reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate in addition to increasing water demand. As water levels fluctuate from hydrologic impacts, direct mortality is expected to increase (indicated by a decreasing condition value for direct mortality (Table C.2.5)) from stranding and predation as well as from fluctuations in water quality and other habitat factors. Zebra mussels are anticipated to infest Lake Grapevine, resulting in increased competition for space and nutrients. Substrate and fragmentation continue to maintain current condition. The moderate declines in water quality and hydrology habitat factors coupled with those for direct mortality and invasive species project to moderate declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population factors resulted in a projected moderate decline in population resiliency (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, direct mortality and invasive species continue due to the same sources described in the Moderate 25-year scenario, while hydrology undergoes a severe decline. Hydrologic alterations driven by climate change will experience a severe decline due to further reductions in 7-day minimum flows and summer minimum base flows. Substrate and fragmentation continue to maintain current condition. The moderate declines in water quality and the severe decline in hydrology habitat factors combined with the moderate decline in direct mortality and invasive species project to moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population) resulting in a projected moderate decline in population resiliency (Table C.2.7).

Grapevine Lake – Severe Increase in Stressors

Change from extirpated population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, hydrology, direct mortality, and invasive species are anticipated. Water quality degradation is expected as Grapevine Lake is in a highly urbanized area and would be subjected to general increasing point and non-point source pollution with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Lake elevation is expected to fluctuate due to reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate in addition to increasing water demand. As water levels fluctuate, direct mortality is expected to increase from stranding and predation as well as from fluctuations in water quality and other habitat factors. Zebra mussels are anticipated to infest Lake Grapevine, resulting in increased competition for space and nutrients. Substrate and fragmentation continue to maintain current condition. The moderate declines in water quality and hydrology habitat factors coupled with moderate declines for direct mortality and invasive species projected to moderate declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population factors resulting in a projected moderate decline in population resiliency (Table C.2.11).

In the **Severe 50-year scenario** (Table C.2.12), moderate declines in water quality, direct mortality and invasive species are expected to continue, while hydrology undergoes a severe decline. Declines in water quality, direct mortality and invasive species continue due to the same sources described in the Severe 25-year scenario. Hydrology is anticipated to experience a severe decline due to further reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate. Substrate and fragmentation continue to maintain current condition. The moderate decline in water quality and the severe decline in hydrology habitat factors coupled with the moderate decline in direct mortality and invasive species project to moderate declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population factors resulting in a projected moderate decline in population resiliency (Table C.2.13).

Trinity River/Lake Livingston – Current Condition

The Trinity River/Lake Livingston focal area currently has a low current condition/probability of persistence. Point sources are significant in the upper Trinity (large daily volumes of treated municipal wastewater discharged), with contaminants typical of effluent dominated waters near urban centers and some distance downstream, including elevated nutrients (e.g., nitrogen, phosphorus), dissolved solids, disinfection by-products, total organic carbon, haloacetic acid, and trihalomethane (TWDB 2015 p. 1.45, 1.46); contaminants of emerging concern like pharmaceuticals, fragrances, and musks are also present. Non-point source pollution typical of urban and rural areas also impacts water quality. Legacy contamination including dioxins, PCBs, furans, and chlordane have affected large areas of the upper Trinity with fish consumption advisories/bans in place. Fluctuations in dissolved oxygen occur; low dissolved oxygen is typically not a problem but can drop to levels stressful for fish (2 - 3 mg/L) in some segments during low flows and warm weather; elevated nutrients may cause algal blooms and fish kills due to phytotoxins or large diurnal fluctuations in dissolved oxygen (TRA 2018, p. 47). Urban run-off and non-point sources may contribute a variety of trace metals (e.g. lead), pesticides, and other pollutants that can harm aquatic life, some of which accumulate in fish and other biota (TWDB 2015, p. 1.50). Reservoir development, groundwater drawdown, and return flows of treated wastewater have greatly altered natural flow patterns in the focal area. Portions of the focal area are on the 303(d) impaired water bodies list for nutrients, bacteria, ammonia and chlorine. Elevated base flows from wastewater returns have resulted in increased shear stress, bank instability, and scouring to bedrock in areas (Clark and Mangham 2019, p.13).

Trinity River/Lake Livingston – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated and the focal area would maintain current population resiliency (Table C.2.3) and maintain low population condition.

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, substrate and direct mortality are anticipated. Water quality degradation is expected as the Trinity River is impacted by increasing urbanization and a 1.5 times increase in water demand; thus, the focal area will be subjected to general increasing point and non-point source pollution with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels. These water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Erratic hydrologic conditions are expected due to reductions in 7-day minimum flows and summer minimum base flows combined with periods of flooding attributed to urban run-off from increases in impervious surfaces and increasingly intense storm events attributed to climate change. These intense high flows are expected to scour the stream bed removing suitable mussel substrate habitat. Direct mortality is expected to increase (indicated by a decreasing condition value for direct mortality (Table C.2.5)) due to the negative changes in water quality, hydrology and substrate. The moderate declines in water quality, hydrology, and substrate habitat factors as well as direct mortality project to moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency. Despite the increase in stressors, the population maintains low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), severe declines in water quality, hydrology, and direct mortality condition are projected as ongoing water quality degradation is exacerbated by increasing water demands and a greater than 30% reduction in 7-day minimum flows and summer minimum base flows from present-day levels. Substrate is expected to continue in moderate decline as those threats described in the Moderate 25-year scenario continue. Fragmentation and invasive species continue to maintain current condition. The severe declines in direct mortality (indicated by a decreasing condition value for direct mortality (Table C.2.7)), as well as in the water quality and hydrology habitat factors projected to severe decline for all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance). A severe decline in resiliency is anticipated, ultimately resulting in extirpation within the time-step of this scenario (Table C.2.7).

Trinity River/Lake Livingston – Severe Increase in Stressors

Change from the current low population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated and the focal area would maintain current population resiliency and maintain low population condition (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, hydrology, substrate and direct mortality are expected from the same stressors described in the Moderate 25-year scenario, but higher in magnitude. The moderate declines in water quality, hydrology, and substrate habitat factors, as well as direct mortality (indicated by a decreasing condition value for direct mortality (Table C.2.4)), project to moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) factors resulting in a projected moderate decline in population resiliency (Table C.2.11). Despite the increase in stressors, the population maintains low condition.

In the **Severe 50-year scenario** (Table C.2.12), a severe decline in water quality, hydrology, and direct mortality is anticipated from the same stressors described in the Moderate 50-year scenario, but with greater magnitude. Substrate is expected to continue in moderate decline as those threats described in the Severe 25-year scenario continue. Fragmentation and invasive species continue to maintain current condition. The severe declines in direct mortality (indicated by a decreasing condition value for direct mortality (Table C.2.6), as well as in the water quality and hydrology habitat factors, projected to severe decline for all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance)(Table C.2.13). A severe decline in resiliency is anticipated, ultimately resulting in extirpation within the time-step of this scenario.

Louisiana Pigtoe

The range of the Louisiana Pigtoe is currently represented by 13 focal areas within six river basins: Little River/Rolling Fork, Cossatot River, Saline River, Lower Little River and Big Cypress Bayou in the Red River basin; Upper Calcasieu River in the Calcasieu-Mermentau River basin; Pearl River in the Pearl River basin; Sabine River and Bayou Anacoco in the Sabine River basin; Angelina River, Neches River and Lower Neches River in the Neches River basin; and East Fork San Jacinto River in the San Jacinto River basin.

Red River Basin

Little River/Rolling Fork – Current Condition

The Little River/Rolling Fork focal area currently has a moderate population condition/ probability of persistence. Tributaries and portions of the focal area are listed as impaired on the 303(d) list for mercury, zinc, lead, silver, pH, dissolved oxygen, and turbidity. Six wastewater permits discharge into the Little River for a combined total of 4.7 million gallons per day.

Little River/Rolling Fork – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in hydrology and substrate are anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (13 – 25%) and summer minimum base flows (7 – 14%) while the durations of high flow events increase (up to 16%) (Lafontaine et al. 2019, entire). The increasing duration of high flows are expected to cause scouring of the stream bed, removing suitable mussel substrate habitat and sediment deposition in mussel beds. Water quality, fragmentation, direct mortality, and invasive species continue to maintain current condition. The moderate declines in hydrology and substrate habitat factors project to moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency shifting the population into low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, substrate, fragmentation, and direct mortality are expected as hydrology undergoes severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degrades due to increasing concentrations of pollutants attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water. Periodic fragmentation would occur as a result of streambed drying and direct mortality is expected from desiccation and exposure to predators. With the moderate decline in water quality, substrate, fragmentation, and direct mortality condition coupled with the severe decline in hydrology, severe declines in population factors host fish availability, reproduction/recruitment, occupied habitat, and abundance are projected, resulting in a severe decline in population resiliency. Low population condition is anticipated during this time-step (Table C.2.7).

Little River/Rolling Fork – Severe Increase in Stressors

Change from the current moderate population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in hydrology and substrate are anticipated. Changes in hydrologic conditions attributed to climate change are expected with a reduction in 7-day minimum flows between 30 – 40% and summer minimum base flows 26 – 33%, while durations of high flow events increase up to 16% (Lafontaine et al. 2019, entire). The effects to substrate are anticipated to manifest somewhere between the Moderate 25 and 50 year scenarios described above. Water quality, fragmentation, direct mortality, and invasive species continue to maintain current condition. The moderate declines in hydrology and substrate habitat factors project to moderate declines in all population factors resulting in a projected moderate decline in population resiliency shifting the population into low condition (Table C.2.11).

In the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality and hydrology are anticipated from the same stressors described in the Moderate 50-year scenario, but greater in magnitude. Water quality degrades due to increasing concentrations of pollutants attributed to a 30 – 40% drop in 7-day minimum flows, 26 – 33% drop in summer minimum base flows (Lafontaine et al. 2019, entire) and increasing water demand. Flashiness (intense flow of short duration) in the stream system is expected to increase the occurrence of harmful shear stresses and sediment deposition, thus a continuation of moderate decline in substrate. With the drop in minimum base flows, fragmentation would intensify beyond the level described in the Moderate 50-year scenario. Direct mortality moves to moderate decline as desiccation and exposure to predation is expected to increase from streambed narrowing and drying. Invasive species maintains current condition. Severe declines in water quality and hydrology coupled with moderate declines in substrate, fragmentation, and direct mortality project to severe declines in all population factors (Table C.2.13). It is projected that the upper reaches of the focal area will experience a severe decline while more stable conditions in the lower portion will persist avoiding extirpation, thus maintaining the population in low condition.

Cossatot River – Current Condition

The Cossatot population currently has a high population condition/probability of persistence. No 303(d) impairments are listed for this focal area, but mercury in fish tissue is beyond EPA recommended consumption level. More than 60 wastewater permitted facilities, mostly pig farms, but also sand/gravel mining are in the focal area. Gillham Lake, upstream of the focal area, alters natural stream flows.

Cossatot River – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (16 – 21%) and summer minimum base flows (21 – 23%) while a -5% decrease in flashiness is expected (Lafontaine et al. 2019, entire). Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency. The hydrologic impacts were not deemed significant enough to downgrade the population to moderate; therefore, it remains in high condition during this time-step (Table C.2.5).

During the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and substrate are expected as hydrology undergoes a severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Sand and gravel operations in the watershed are expected to contribute sediment into the system affecting water quality. Run-off from concentrated animal feeding operations, in this instance hog farms, is expected as well. Water quality degrades due to these inputs and the decline in 7-day minimum flows and summer minimum base flows. The combination of decreased flashiness described in the Moderate 25-year scenario and expected sediment deposition would impact substrate as cleansing flows become less frequent. Fragmentation, direct mortality, and invasive species continue to maintain current condition. With the moderate decline in water quality and substrate coupled with the severe decline in hydrology, moderate declines in all population factors are projected to continue. A moderate decline in population resiliency is projected as a result with the population downgraded to moderate condition during this time-step (Table C.2.7).

Cossatot River – Severe Increase in Stressors

Change from the current high population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (35 – 42%) and summer minimum base flows (26 – 30%) while a reduction in flashiness is expected (Lafontaine et al. 2019, entire). Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.11). The hydrologic impacts were not deemed significant enough at this time-step to downgrade the population to moderate; therefore, it remains in high condition.

During the **Severe 50-year scenario** (Table C.2.12), a severe decline in hydrology is expected, triggering effects to other habitat factors. Water quality and substrate habitat factors degrade to severe decline and moderate decline respectively. The reductions in 7-day minimum flows and summer minimum base flows intensify, approaching the upper range discussed in the Severe 25-year scenario. The same affects to water quality (increasing concentration on pollutants) and substrate (sediment accumulation on mussel beds) described in the Moderate 50-year scenario occur. Fragmentation, direct mortality, and invasive species continue to maintain current condition. With the severe decline in water quality and hydrology coupled with the moderate decline in substrate, severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors are projected (Table C.2.13). A severe decline in population resiliency is projected as a result with the population downgraded to low condition during this time-step.

Saline River (Little) – Current Condition

The Saline River focal area is currently in moderate condition and is expected to decline to low condition for the next 50 years throughout all future scenarios. Portions of the focal area are not in attainment for dissolved oxygen. Natural flow conditions have been altered by Dierk's Lake in the upstream portion of the focal area. Although erratic flow is uncommon, prolonged high water is common for flood control.

Saline River (Little) – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (18 – 25%) and summer minimum base flows (16 – 19%) (Lafontaine et al. 2019, entire). Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency. The hydrologic impacts were not deemed significant enough to downgrade the population to low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and substrate are expected as the hydrology undergoes severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water. With the moderate decline in water quality and substrate coupled with the severe decline in hydrology, moderate declines in population factors host fish availability, reproduction/recruitment, occupied habitat, and abundance are projected, resulting in a moderate decline in population resiliency. The hydrologic impacts were not deemed significant enough to downgrade the population low condition to extirpated (Table C.2.7).

Saline River (Little) – Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8)**,** no changes from the current condition are expected. Therefore, no change in habitat or population factors is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (33 – 40%) and summer minimum base flows (28 – 33%) while a reduction in flashiness is expected (Lafontaine et al. 2019, entire). Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.11). The hydrologic impacts were deemed significant enough at this time-step to downgrade the population to low condition.

During the **Severe 50-year scenario** (Table C.2.12), a severe decline in hydrology is expected, triggering effects to other habitat factors. Severe decline in water quality is anticipated, as substrate and fragmentation habitat factors undergo moderate decline. The reductions in 7-day minimum flows and summer minimum base flows intensify, approaching the upper range discussed in the Severe 25-year scenario. The affects to water quality (increasing concentration on pollutants) and substrate (sediment accumulation on mussel beds) described in the Moderate 50-year scenario occur with more intensity. Decreased flashiness described in the Moderate 25-year scenario would affect substrate as cleansing flows become less frequent. With the drop in summer minimum base flows, fragmentation is expected due to episodic stream bed drying. Direct mortality and invasive species continue to maintain current condition. With the severe decline in water quality and hydrology coupled with the moderate decline in substrate and fragmentation, severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors are projected (Table C.2.13). A severe decline in population resiliency is projected, but impacts were not deemed significant enough at this time-step to downgrade the population to extirpated; therefore, it remains in low condition.

Lower Little River Focal Area – Current Condition

The Louisiana Pigtoe population in the Lower Little River focal area is currently considered functionally extirpated and is expected to remain so over the next 50 years in all future scenarios. A portion of the focal area is on the 303(d) impairment list for temperature.

Lower Little River Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors is anticipated and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (19%) and summer minimum base flows (12%) (Lafontaine et al. 2019, entire). Releases from Millwood Lake dam are expected to buffer losses from minimum base flows described above. Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.5).

During the **Moderate 50-year scenario** (Table C.2.6), a moderate decline in hydrology is persists. Changes in hydrologic conditions described in the Moderate 25-year scenario intensify (Lafontaine et al. 2019, entire), and releases from Millwood Lake dam continue to buffer losses from minimum base flows described above. Water quality, substrate, fragmentation, direct mortality, and invasive species maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.7).

Lower Little River – Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8)**,** no changes from the current condition are expected. Therefore, no change in habitat or population factors is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (39%) and summer minimum base flows (29%) (Lafontaine et al. 2019, entire). Releases from Millwood Lake dam are expected to buffer losses from minimum base flows described above. Water quality, substrate, fragmentation, direct mortality, and invasive species maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.11).

During the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality and hydrology are anticipated from the same stressors described in the Severe 25-year scenario, but greater in magnitude. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reductions in 7-day minimum flows and summer minimum base flows are projected to decrease near or above 30%. Releases from Millwood Lake dam supplements some loss from minimum base flows described above. Substrate, fragmentation, direct mortality, and invasive species maintain current condition. Due to the severe decline in the water quality and hydrology habitat factors, severe declines were projected for all population factors which in turn projected a severe decline in population resiliency (Table C.2.13).

Big Cypress Bayou – Current Condition
The Big Cypress Bayou focal area currently has a moderate population condition/probability of persistence. A portion of the focal area (Texas River Segment 0402) was identified on the Texas §303(d) List as having elevated mercury in fish tissue, low pH, and depressed dissolved oxygen in 1998, 2000, and 2010, respectively. The impairments remained on the 2014 Texas §303(d) List. However, pH samples collected since 2014 show that the standard is being met and Texas River Segment 0402 was removed from the 2016 §303(d) List. Another portion (Texas River Segment 0409) was identified as impaired for low levels of dissolved oxygen in 2000 and for elevated bacteria (*E. coli*) levels in 2006. The 2014 and 2016 Texas §303(d) Lists confirmed the impairment. Data collected since 2014 indicate elevated bacteria and low dissolved oxygen levels are still present. Multiple wastewater treatment plants discharge effluent into the focal area. Voluntary instream flows for Cypress Basin are in place, but the strategies to meet future water needs of regional water plans and the State Water Plan are not to be limited by these voluntary goals for instream flows. On-channel Lake of the Pines and Lake Bob Sandlin upstream of focal area have altered natural stream flow conditions. A proposed reservoir could affect flows in the focal area (Little Cypress Reservoir), but the North East Texas Regional Water Planning Group does not recommend the designation of the potential reservoir site as a unique reservoir site. The invasive/exotic aquatic plant Giant salvinia is established in this watershed.

Big Cypress Bayou – Moderate Increase in Stressors
Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (20 – 23%) and summer minimum base flows (16 – 29%) (Lafontaine et al. 2019, entire). Water quality, substrate, fragmentation, direct mortality, and invasive species maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency. The hydrologic impacts were not deemed significant enough to downgrade the population to low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and direct mortality are expected as the hydrology undergoes severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degrades due to anthropogenic alterations affecting total maximum daily loads, conductivity and other pollutants attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Channel narrowing or complete loss of flowing water due to the reduction in summer minimum base flows would cause desiccation and increased exposure to predation. With the moderate decline in water quality and direct mortality coupled with the severe decline in hydrology, moderate declines in population factors host fish availability, reproduction/recruitment, occupied habitat, and abundance are projected, resulting in a moderate decline in population resiliency. Therefore, the population is downgraded to low condition during this time-step (Table C.2.7).

Big Cypress Bayou – Severe Increase in Stressors
In the **Severe 10-year scenario** (Table C.2.8**),** no changes from the current condition are expected. Therefore, no change in habitat or population factors is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), a moderate decline in hydrology is anticipated. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (27 – 35%) and summer minimum base flows (30 – 40%) (Lafontaine et al. 2019, entire) as well as increasing water demand. Water quality, substrate, fragmentation, direct mortality, and invasive species continue to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.11). The hydrologic impacts were not deemed significant enough at this time-step to downgrade the population; therefore, it remains in moderate condition.

During the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality and hydrology are anticipated from the same stressors described in the Severe 25-year scenario, but greater in magnitude. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reductions in 7-day minimum flows and summer minimum base flows are projected to decrease $\leq$ 30% increasing probability of desiccation and exposure to predation. Substrate, fragmentation, and invasive species maintain current condition. Due to the severe decline in the water quality and hydrology habitat factors, moderate declines were projected for all population factors. Because two of the three habitat factors are in severe decline, the focal area is expected to experience a severe decline in population resiliency resulting in low condition during this time-step (Table C.2.13).

Calcasieu-Mermentau River Basin

Upper Calcasieu River – Current Condition
The Upper Calcasieu River focal area currently has a low current condition/probability of persistence. It is listed as impaired on the 303(d) list for pH and fecal coliform. Sources of point and non-point pollution include municipal wastewater discharges, paper mill effluent, and sand/gravel mining. Calcasieu River within the focal area is designated under Louisiana's Natural and Scenic River System. These waterways are protected by a permit process, and there are certain prohibitions against channelization, impoundment construction, and channel realignment. Continued population growth at the historical rate will likely increase demand for high-quality water supplies for both public supply and industrial uses. Increased water extraction during low rainfall periods to supply local agricultural practices is anticipated.

Upper Calcasieu River Focal Area – Moderate Increase in Stressors
Change from the current condition is expected for hydrology and fragmentation during the **Moderate 10-year scenario** (Table C.2.2). Removal of a low-head dam in the upper portion of the focal area is planned and is anticipated to improve hydrologic conditions and remove a fish passage barrier, thus decreasing fragmentation in the system. All other threats maintain current condition. With the dam removal, moderate improvements to hydrology and habitat structure/substrate habitat factors are expected as stream flows return to more natural conditions. Moderate improvements to all population factors are anticipated as a result and a moderate improvement in population resiliency is projected. The population maintains its low condition despite the improving habitat and population factors as they merely buffer effects from other threats to habitat factors described above (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology is anticipated while the moderate improvement to fragmentation continues. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows of 23% and summer minimum base flows of 37% (Lafontaine et al. 2019, entire) while demands for surface and groundwater continue their current trend. Positive biological and hydrological responses from reduced fragmentation are expected. Water quality, substrate, direct mortality, and invasive species maintain current condition. With the moderate decline in the hydrology and moderate improvement in fragmentation habitat factors, all population factors are projected to maintain current condition. The focal area would maintain current population resiliency, with the population remaining in low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and substrate are expected as the hydrology undergoes severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water as well as sediment accumulation on mussel beds from a lack of adequate cleansing flows. With the moderate decline in water quality and substrate coupled with the severe decline in hydrology, moderate declines in all population factors are projected, resulting in a moderate decline in population resiliency. Extirpation of the population is projected during this time-step (Table C.2.7).

Upper Calcasieu River – Severe Increase in Stressors

Change from the current condition is expected for hydrology and fragmentation during the **Severe 10-year scenario** (Table C.2.8). Moderate improvements to both habitat factors are expected from the reduced threats described in the Moderate 10-year scenario. The population maintains low condition despite improving habitat and population factors as they merely buffer effects from the other threats to habitat factors described in the Moderate Increase in Stressors section (Table C.2.9).

During the **Severe 25-year scenario** (Table C.2.10), a severe decline in hydrology is expected, triggering moderate declines in water quality and direct mortality. Moderate improvements to fragmentation are expected as threats are reduced as stream flows return to more natural conditions after dam removal. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows of 35% and summer minimum base flows of 52% (Lafontaine et al. 2019, entire) while demands for surface and groundwater continue their current trend. Water quality degrades due to increasing concentrations of pollutants attributed to the decline in summer minimum base flows. Reduction in summer minimum base flows would subject mussel beds to drying events from channel narrowing or dewatering increasing exposure to predation and desiccation. Positive biological and hydrological responses from reduced fragmentation are expected as stream flows return to more natural conditions. Based on these threats, moderate decline is projected for the water quality habitat factor; severe decline is projected for the hydrology habitat factor; and moderate improvement is projected for the habitat structure/substrate habitat factor. Declines in all population factors are projected as a result, leading to a moderate decline in population resiliency with the population remaining in low condition (Table C.2.11).

In the **Severe 50-year scenario** (Table C.2.12), a severe decline in hydrology is expected as the impacts discussed in the Severe 25-year scenario intensify, causing cascading effects to other habitat factors. A severe decline in water quality and moderate declines in substrate and fragmentation are triggered as a result. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water causing fragmentation. As a result, severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors are projected, causing a severe decline in population resiliency (Table C.2.13). Extirpation of the population is projected during this time-step.

Pearl River Basin

Pearl River Focal Area – Current Condition

The Pearl River focal area currently has a low current condition/probability of persistence. The main channel and/or numerous tributaries are on the 303(d) list of impaired waterbodies for various causes including biological impairment, sulfate, pH, dissolved oxygen, and turbidity. Other past and current stressors to water quality include point and non-point source pollution from urban areas and chemical releases from a paper mill near Bogalusa, Louisiana in 2011 causing a substantial fish kill. The Ross R. Barnett Reservoir, construction completed in 1963, influences the current hydrologic condition of the focal area. An additional reservoir on the main channel of the Pearl River below the Ross R. Barnett Reservoir is proposed for flood control.

Pearl River Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario (**Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated with the focal area maintaining current population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, substrate, fragmentation, and direct mortality are anticipated while invasive species maintain current conditions. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (15 – 19%) and summer minimum base flows (12 – 19%) (Lafontaine et al. 2019, entire). In addition, hydrologic conditions would be negatively affected by the construction of a flood control reservoir proposed for the upper portion of the focal area during this time-step. Water quality degradation due to increasing wastewater returns and concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows. With the reduction in base flows, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. The flood control reservoir would function as a fish passage barrier, causing the loss of approximately 20 miles of occupied habitat. Direct mortality is expected to increase due to habitat loss and hydrologic alteration from reservoir construction. As a result of these threats, moderate decline is expected for the water quality, hydrology, and habitat structure/substrate habitat factors as well as all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat and abundance). The focal area would undergo a moderate decline in population resiliency, with the population remaining in low condition (Table C.2.5).

During the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, hydrologic conditions, substrate, fragmentation and direct mortality are expected to continue. Threats from the same sources described in the Moderate 25-year scenario continue, but with increasing intensity. Invasive species maintain current condition. As a result of these threats, moderate decline is expected for the water quality, hydrology, and habitat structure/substrate habitat factors as well as all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat and abundance). The focal area would undergo a moderate decline in population resiliency, with the population remaining in low condition (Table C.2.7).

Pearl River – Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8), no changes from the current condition are expected. Therefore, no change in habitat or population factors is anticipated, and the focal area would maintain current population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, hydrology, substrate, fragmentation, and direct mortality are anticipated while invasive species maintain current conditions. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (20 – 22%) and summer minimum base flows (14 – 22%) (Lafontaine et al. 2019, entire). With the modeled reductions in base flows as well as the construction of the flood control reservoir, the same threats described in the Moderate 25-year scenario would occur, but with greater intensity. As a result of these threats, moderate decline is expected for the water quality, hydrology, and habitat structure/substrate habitat factors as well as all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat and abundance). The focal area would undergo a moderate decline in population resiliency, with the population remaining in low condition (Table C.2.11).

During the **Severe 50-year Scenario** (Table C.2.12), a severe decline in hydrology is expected as the impacts discussed in the Severe 25-year scenario intensify, causing cascading effects to other habitat factors. With the modeled reductions in base flows as well as the construction of the flood control reservoir, the same threats in the Severe 25-year scenario would occur, but with greater intensity. A moderate decline in water quality, substrate, fragmentation and direct mortality continue as a result. A severe decline in the hydrology habitat factor is projected while water quality and habitat quality/substrate are projected to undergo a moderate decline. Severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors are projected, causing a severe decline in population resiliency (Table C.2.13). Extirpation of the population is projected during this time-step.

Sabine River Basin

Sabine River Focal Area – Current Condition

The currently extirpated Sabine River focal area (Table C.2.14) is expected to remain extirpated in the next 50 years in all future scenarios. Two segments within the focal area are on the 303(d) list of impaired waterbodies for bacteria. A new poultry processing plant has been permitted to release wastewater in the upper portion of the focal area downstream of Lake Tawakoni. Wastewater releases are permitted at 2.18 million gallons per day with an ammonia limit of 3.94 mg/L, which is beyond the threshold for freshwater mussel tolerances. Water quality degradation is expected as result, despite wastewater dilution from mixing with stream flow. The construction of Lake Tawakoni and Toledo Bend Reservoir impacted natural hydrologic conditions and dam releases cause substrate scouring eliminating mussel habitat downstream until sheer stress dissipates. An additional off-channel reservoir in the middle of the focal area and a water diversion project are proposed to meet future water demand. When constructed, water quality and hydrologic conditions would further degenerate from current conditions. Bank erosion is prevalent throughout the focal area, resulting in elevated inputs of sediment impacting suitable substrates for mussel beds.

Sabine River Focal Area- Moderate Increase in Stressors

In the **Moderate 10-year scenario (**Table C.2.2), the focal area is projected to endure a moderate decline in water quality due to degradation resulting from a general increase in point and non-point source discharges, including significant wastewater effluent flows from a new poultry processing plant into a portion of the river with documented mussel beds. This degradation in water quality is expected to negatively influence overall mussel survival and reproductive success, potentially affecting both mussel and host fish movement, and subsequently causing fragmentation of suitable habitat. In some cases, water quality degradation may result in increased direct mortality of Louisiana Pigtoe, contributing to a moderate decline in this focal area. Hydrology, substrate and invasive species are expected to maintain their current condition of moderate decline. The changes to threat conditions described above negatively affected modelled Louisiana Pigtoe water quality and habitat structure/substrate habitat factors and all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, substrate, fragmentation, and direct mortality are expected to continue in tandem with population growth and associated impacts (e.g., habitat loss, increased demand for water supply, and increased generation of wastewater). Declining conditions of water quality, fragmentation, and direct mortality would be exacerbated by the effects of climate change. The moderate decline in hydrology is expected, in part, from future predicted reductions in flow, as represented by reductions in 7-day minimum flows (1 – 30%) and summer minimum base flows (10 – 29%) arising from a changing climate (Lafontaine et al. 2019, entire). Subsequently, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. The threat posed by invasive species is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled habitat and population factors (moderate declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, substrate, fragmentation, and direct mortality will continue due to the threats described above. Hydrology is expected to severely decline due to climate change, including significant reductions in 7-day minimum and summer minimum base flows, as well as the construction of an off-channel reservoir in the middle Sabine River basin; these changes to hydrology and flow will further degrade water quality. Threats from invasive species are expected to maintain current condition. The changes to threat conditions described above negatively affected modelled habitat with a severe decline in hydrology and moderate declines in water quality and habitat structure/substrate. Population factors of host fish availability, reproduction/recruitment, occupied habitat, and abundance undergo severe decline, resulting in a projected severe decline in population resiliency (Table C.2.7).

Sabine River Focal Area - Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8), we anticipate moderate declines in water quality, fragmentation and direct mortality based on the same threats assessed in the Moderate 10-year scenario. Changes to hydrology, substrate and invasive species are expected to maintain their current condition. The changes to threat conditions described above negatively affected modelled Louisiana Pigtoe water quality and habitat structure/substrate habitat factors and all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, substrate, fragmentation, and direct mortality continue. Conditions of water quality, fragmentation and direct mortality would be exacerbated by the same stressors described above. A moderate decline in substrate condition is expected as sediments accumulate on mussel beds from a lack of adequate cleansing flows. This change in substrate condition is correlated with an expected severe decline in hydrological condition from reductions in 7-day minimum flows (20 – 22%) and summer minimum base flows (14 – 22%) (Lafontaine et al. 2019, entire) arising from a changing climate in addition to an off-channel reservoir constructed in the middle of the Sabine River basin; these changes to hydrology and flow will further degrade water quality. Invasive species condition is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled habitat factors with a severe decline in hydrology and moderate declines in water quality and habitat structure/substrate. Population factors of host fish availability, reproduction/recruitment, occupied habitat, and abundance undergo severe decline, resulting in a projected severe decline in population resiliency (Table C.2.11).

In the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality and hydrology are anticipated resulting from increasing demands for waters supply and increasing point and non-point source pollution. Changes to flow include an estimated 30% reduction in minimum base flows as well as the construction of an off-channel reservoir in the middle Sabine River basin. Moderate declines in substrate, fragmentation, and direct mortality are anticipated from the same sources described in the Severe 25-year scenario. Invasive species condition is expected to maintain current condition. The changes to threat conditions described above negatively affected modelled Louisiana Pigtoe habitat and population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected severe decline in population resiliency (Table C.2.13).

Bayou Anacoco Focal Area – Current Condition

The Bayou Anacoco focal area has a moderate current condition/probability of persistence. It is currently on the 303(d) impaired water bodies list for total dissolved solids and fecal coliform. Municipal and Industrial wastewater discharges into Bayou Anacoco include Boise Packing and Newsprint-Deridder Paper Mill (39 million gallons per day) and City of Leesville Wastewater Treatment Facility (2.1 million gallons per day). Lake Vernon and Anacoco Lake are upstream of the focal area. The two impoundments and wastewater discharges have altered natural hydrologic and water quality conditions throughout the focal area.

Bayou Anacoco Focal Area – Moderate Increase in Stressors

In the **Moderate 10-year scenario (**Table C.2.2), the focal area is projected to endure a moderate decline in hydrology due to reduced stream flows from dam repairs and filling of Vernon Lake. Subsequently, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. Threats to water quality, fragmentation, direct mortality and invasive species are expected to maintain their current condition. The changes to threat conditions described above resulted in moderate decline in hydrology and habitat structure/substrate habitat factors. All population factors are projected to undergo moderate decline as a result, and a projected moderate decline in population resiliency is expected. The population is downgraded to low condition during this time-step (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), a moderate decline in hydrology is expected to continue due to a modelled 35% reduction in 7-day minimum flows and 30% reduction in summer minimum base flows (Lafontaine et al. 2019, entire). All other threat categories are expected to maintain current condition. Due to the moderate decline in the hydrology habitat factor, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency. The hydrologic impacts were not deemed significant enough to downgrade the population, and the system is expected to recover from Vernon Lake dam repairs/filling resulting in a projected upgrade of the population to moderate condition (Table C.2.5).

During the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality and substrate are expected as hydrology undergoes severe decline as the impacts discussed in the Moderate 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degradation due to increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water as well as sediment accumulation on mussel beds from a lack of adequate cleansing flows. With the moderate decline in water quality and substrate coupled with the severe decline in hydrology, moderate declines in all population factors are projected, resulting in a moderate decline in population resiliency. The population is downgraded to low condition during this time-step (Table C.2.7).

Bayou Anacoco Focal Area- Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8), we anticipate moderate declines in hydrology and substrate based on the same threats assessed in the Moderate 10-year scenario. Changes to water quality, fragmentation, direct mortality and invasive species are expected to maintain their current condition. The changes to threat conditions described above negatively affected modelled Louisiana Pigtoe hydrology and habitat structure/substrate habitat factors and all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulting in a projected moderate decline in population resiliency. The population is downgraded to low condition during this time-step (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), a severe decline in hydrology is expected to continue due to a modelled 41% reduction in 7-day minimum flows and 36% reduction in summer minimum base flows (Lafontaine et al. 2019, entire). A moderate decline in water quality is expected due to threats from increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels attributed to paper mill and municipal wastewater effluent, the decline in 7-day minimum flows, summer minimum base flows, and increasing water demand. These water quality impacts are expected to increase threats in direct mortality. Substrate, fragmentation, and invasive species are expected to maintain current condition. Due to the severe decline in the hydrology and moderate decline in water quality habitat factors, moderate declines were projected for all population factors which in turn projected a moderate decline in population resiliency (Table C.2.11). The decline in population resiliency was not deemed significant enough to downgrade the population, and the system is expected to recover from Vernon Lake dam repairs/filling resulting in a projected upgrade of the population to moderate condition.

During the **Severe 50-year scenario** (Table C.2.12), a severe decline in water quality and moderate decline in substrate is expected as hydrology undergoes severe decline as the impacts discussed in the Severe 25-year scenario intensify, causing cascading effects to other habitat factors. Water quality degradation described in the Severe 25-year scenario intensify. Reduction in summer minimum base flows would subject substrates to more frequent and profound drying events from channel narrowing or complete loss of flowing water as well as sediment accumulation on mussel beds from a lack of adequate cleansing flows. With the moderate decline substrate coupled with the severe decline in hydrology and water quality, severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors are projected (Table C.2.13). A severe decline in population resiliency is projected, and extirpation is expected during this time-step.

Neches River Basin

Angelina River Focal Area – Current Condition

The Angelina River focal area currently has a low population condition/probability of persistence. Segments of the focal area are on the 303(d) impaired water bodies list for bacteria. Fecal coliform often exceeded standards in the late 1990s, and elevated ammonia levels were routinely observed in 2008. No impoundments are on Angelina River upstream or within the focal area. Two reservoirs, Lake Columbia and Lake Ponta, are proposed in on a major tributary to the focal area. Both would be constructed on Mud Creek in the upper watershed of the focal area, altering hydrology and substrates.

Angelina River Focal Area – Moderate Increase in Stressors

In the **Moderate 10-year scenario (**Table C.2.2), the focal area is projected to endure a moderate decline in substrate and direct mortality from threats associated with underwater seismic testing. Seismic tests involve explosive charges placed in "shot" holes. Drilling shot holes into the substrate and subsequent explosions are expected to result in direct mortality of individuals and degraded substrate habitat. Threats to water quality, hydrology, fragmentation, and invasive species are expected to maintain their current condition. The changes to threat conditions described above resulted in moderate decline in the habitat structure/substrate habitat factor. All population factors are projected to undergo moderate decline as a result and a projected moderate decline in population resiliency is expected. The population maintains low condition during this time-step (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), increasing stressors result in severe declines in hydrology and substrate and moderate declines in water quality, fragmentation, and direct mortality. Declining water quality condition attributed to the moderate decline in hydrology is expected, in part, from future predicted reductions in flow, as represented by reductions in 7-day minimum flows (25 – 32%) and summer minimum base flows (28 – 29%) arising from a changing climate (Lafontaine et al. 2019, entire). Stream flow reductions from reservoir development in the upper watershed of the focal area are expected as well. These reductions in stream flow are expected to cause temporary fragmentation due to dry periods. Subsequently, a severe decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. Direct mortality from seismic testing is anticipated. Threats from invasive species are expected to maintain current condition. Due to the increase in stressors, severe decline in the hydrology and habitat structure/substrate habitat factors and a moderate decline in the water quality habitat factors is projected. As a result, severe declines were projected for all population factors which in turn projected a severe decline in population resiliency. The population continues to maintain low condition during this time-step (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), severe declines in water quality, hydrology, substrate and fragmentation are expected while direct mortality continues in moderate decline. Hydrology threats intensify, causing cascading threats to the other habitat factors described in the Moderate 25-year scenario. Due to the increase in stressors, severe declines in all habitat factors and population factors are projected, which in turn projected a severe decline in population resiliency. Extirpation is anticipated during this time-step (Table C.2.7).

Angelina River Focal Area – Severe Increase in Stressors

In the **Severe 10-year scenario** (Table C.2.8), we anticipate moderate declines in substrate and direct mortality based on the same threats assessed in the Moderate 10-year scenario. Water quality, hydrology, fragmentation and invasive species are expected to maintain their current condition. The changes to threat conditions described above negatively affected modelled Louisiana Pigtoe habitat structure/substrate habitat factor and all population factors, resulting in a projected moderate decline in population resiliency. The population maintains low condition during this time-step (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), increasing stressors result in severe declines in hydrology and substrate and moderate declines in water quality, fragmentation, and direct mortality. Declining conditions of water quality attributed to the moderate decline in hydrology is expected, in part, from future predicted reductions in flow, as represented by reductions in 7-day minimum flows (34 – 35%) and summer minimum base flows (42%) arising from a changing climate (Lafontaine et al. 2019, entire). Stream flow reductions from reservoir development in the upper watershed of the focal area are expected as well. These reductions in stream flow are expected to cause temporary fragmentation due to dry periods. Subsequently, a severe decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. Direct mortality from seismic testing is anticipated. Threats from invasive species are expected to maintain current condition. Due to the increase in stressors, severe decline in the hydrology and habitat structure/substrate habitat factors and a moderate decline in the water quality habitat factors is projected. As a result, severe declines were projected for all population factors which in turn projected a severe decline in population resiliency (Table C.2.11). The population continues to maintain low condition during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality, hydrology, substrate and fragmentation are expected while direct mortality continues in moderate decline. Hydrology threats intensify, exacerbating threats to the other habitat factors described in the Severe 25-year scenario. Due to the increase in stressors, severe declines in all habitat factors and population factors are projected, which in turn projected a severe decline in population resiliency (Table C.2.13). Extirpation is anticipated during this time-step.

Neches River Focal Area – Current Condition

The Neches River focal area currently has a high population condition/probability of persistence. Tributaries and segments of the focal area are on the 303(d) impaired water bodies list for dioxin and mercury in edible tissue, bacteria, and depressed dissolved oxygen. Numerous segments had concerns for nutrients, particularly ammonia and total phosphorus; however, decreasing trends for these parameters were often observed. Stream flows are influenced by Lake Palestine in the upper portion of the focal area and B.A. Steinhagen Reservoir in the southern portion of the focal area.

Neches River Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency. The population continues to maintain high condition during this time-step (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, substrate, fragmentation, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Sediment accumulation on mussel beds is projected to increase from a lack of adequate cleansing flows. The proposed Rockland Reservoir on the main channel of the Neches River, which would function as a fish passage barrier, is anticipated to be operational at this time-step. Direct mortality is expected to increase due to water quality degradation, reductions in water volume, and habitat loss from reservoir construction. A severe decline in hydrology is attributed to three proposed water delivery projects within the focal area combined with an overall reduction in stream flows. Lake Columbia is an off-channel reservoir proposed in the upper portion of the focal area, a run-of river water diversion is proposed for the middle of the focal area, and Rockland Reservoir is proposed near the downstream end of the focal area. Additionally, reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate are expected. The invasive species factor is expected to maintain current condition. The projected moderate and severe decline in habitat and population factors (i.e., water quality and quantity, fish host availability, reproduction/recruitment, occupied habitat, and abundance) is expected to result in a severe decline in population resiliency. The population is downgraded to low condition during this time-step (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate declines in water quality, substrate, fragmentation, and direct mortality are expected to continue as the threats discussed in the Moderate 25-year scenario are realized and exacerbated by further reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate. Severe declines in host fish availability, reproduction/recruitment, occupied habitat, and abundance population factors continue, as well as declines to habitat factors, contributing to a projected severe decline in population resiliency. The population continues to maintain low condition during this time-step (Table C.2.7).

Neches River Focal Area – Severe Increase in Stressors

Change from the current moderate population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Thus, no change in population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated and the focal area is projected to maintain its current population resiliency. The population continues to maintain high condition during this time-step (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, substrate, fragmentation, and direct mortality are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural steam flows with some increases to municipal wastewater effluent return flows). Sediment accumulation on mussel beds is projected to increase from a lack of adequate cleansing flows. The proposed Rockland Reservoir on the main channel of the Neches River, which would function as a fish passage barrier, is anticipated to be operational at this time-step. Direct mortality is expected to increase due to water quality degradation, reductions in water volume, and habitat loss from reservoir construction. A severe decline in hydrology is attributed to three proposed water delivery projects within the focal area combined with an overall reduction in stream flows. Lake Columbia is an off-channel reservoir proposed in the upper portion of the focal area, a run-of river water diversion is proposed for the middle of the focal area, and Rockland Reservoir is proposed near the downstream end of the focal area. Additionally, reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate are expected. The invasive species factor is expected to maintain current condition. The projected moderate and severe decline in habitat and population factors (i.e., water quality and quantity, fish host availability, reproduction/recruitment, occupied habitat, and abundance) is expected to result in a severe decline in population resiliency (Table C.2.11). The population is downgraded to low condition during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), moderate declines in substrate, fragmentation, and direct mortality are expected to continue as the threats discussed in the Severe 25-year scenario are realized and exacerbated by further changes to hydrology, including reductions in 7-day minimum flows and summer minimum base flows arising from a changing climate. Both water quality and quantity undergo a severe decline as summer minimum base flows are projected to decrease by 30% from present levels (Lafontaine et al. 2019, entire), in addition to the other water volume reductions considered in the Severe 25-year scenario. Severe declines in host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance population factors, as well as declines to habitat factors, contribute to a continuing severe decline in population resiliency (Table C.2.13); however, not to the point of extirpation. The population continues to maintain low condition during this time-step.

Lower Neches River Focal Area – Current Condition

The Lower Neches River focal area currently has a low population condition/probability of persistence. See the information in the Neches River focal area for current water quality information. Stream flows are influenced by B.A. Steinhagen in the upper portion of the focal area.

Lower Neches River Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current low population resiliency (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality and hydrology are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Hydrologic impacts related to climate change, including a reduction in 7-day minimum flows (21 – 25%) and summer minimum base flows (24 – 32%) (Lafontaine et al. 2019, entire), are expected. Substrate, fragmentation, direct mortality, and invasive species maintain current condition. The moderate declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulted in a projected moderate decline in population resiliency. The population continues to maintain low condition during this time-step (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), moderate decline in water quality continues due to the same sources described in the Moderate 25-year scenario. Hydrologic alterations driven by climate change will experience further reductions in 7-day minimum flows and summer minimum base flows. Substrate, fragmentation, direct mortality, and invasive species maintain current condition. The moderate decline in water quality combined with the severe decline in hydrology habitat factors, along with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) resulted in a projected moderate decline in population resiliency. The population continues to maintain low condition during this time-step (Table C.2.7).

Lower Neches River Focal Area – Severe Increase in Stressors

Change from the current low population condition is not expected as no change to habitat factors occur during the **Severe 10-year scenario** (Table C.2.8). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency. The population maintains low condition during this time-step (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality and hydrology are anticipated. Water quality degradation is expected from a general increase in point and non-point source pollution, with increasing concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels; these water quality impacts will be exacerbated by changes to hydrology (i.e., general decrease in natural stream flows with some increases to municipal wastewater effluent return flows). Hydrologic impacts related to climate change, including reductions in 7-day minimum (30 – 36%) and summer minimum base flows (32 – 41%) (Lafontaine et al. 2019, entire), are expected. Substrate, fragmentation, direct mortality, and invasive species maintain current condition. The moderate declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance), resulted in a projected moderate decline in population resiliency (Table C.2.11). The population maintains low condition during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), both water quality and hydrology undergo severe decline as ongoing water quality degradation is exacerbated by a greater than 30% reduction in 7-day minimum flows and summer minimum base flows from present-day levels (Lafontaine et al. 2019, entire). Substrate, fragmentation, direct mortality, and invasive species maintain current condition. The focal area is projected to experience severe declines in water quality and hydrology habitat factors, coupled with moderate declines in population factors (host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance). Since two of the three habitat factors are in severe decline, the focal area is expected to experience a severe decline in population resiliency. The population is anticipated to become extirpated during this time-step (Table C.2.13).

San Jacinto River Basin

East Fork San Jacinto River Focal Area – Current Condition

The East Fork San Jacinto focal area currently has a low population condition/probability of persistence. It is on the 303(d) impaired water bodies list for bacteria. No impoundments are on the East Fork San Jacinto upstream or within the focal area. Lake Houston is downstream of the focal area. No new impoundments are proposed within or upstream of the focal area. Sand mining, in particular, has led to increased nutrient loads in the San Jacinto River which can result in an increase in cyanobacteria levels (Region H water plan pg 1-23).

East Fork San Jacinto Focal Area – Moderate Increase in Stressors

Change from the current condition is not expected for any threats during the **Moderate 10-year scenario** (Table C.2.2). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency. The population maintains low condition during this time-step (Table C.2.3).

In the **Moderate 25-year scenario** (Table C.2.4), moderate declines in water quality, hydrology, substrate, and direct mortality are anticipated while fragmentation and invasive species maintain current condition. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (9%) and summer minimum base flows (30%) (Lafontaine et al. 2019, entire). Water quality degradation due to increasing wastewater returns and concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels are attributed to the decline in 7-day minimum flows, summer minimum base flows and increased water demand. With the reduction in base flows, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. Direct mortality is expected to increase due to the threats above as well as desiccation and increased exposure to predation during dry periods. As a result of these threats, moderate decline is expected for the water quality, hydrology, and habitat structure/substrate habitat factors as well as all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat and abundance). The focal area would undergo a moderate decline in population resiliency, with the population remaining in low condition (Table C.2.5).

In the **Moderate 50-year scenario** (Table C.2.6), severe declines in water quality, hydrology, and substrate are expected while direct mortality continues in moderate decline. Fragmentation and invasive species threats maintain current condition. Hydrology threats described in the Moderate 25-year scenario intensify, causing cascading threats to the other habitat factors described in the Moderate 25-year scenario. Due to the increase in stressors, severe declines in water quality and hydrology and a moderate decline in habitat structure/substrate habitat factors are projected. Severe declines in all population factors are projected as a result, which in turn projected a severe decline in population resiliency. Extirpation is anticipated during this time-step (Table C.2.7).

East Fork San Jacinto Focal Area – Severe Increase in Stressors

Change from the current condition is not expected for any threats during the **Severe 10-year scenario** (Table C.2.8). Therefore, no change in habitat or population factors (i.e., water quality and quantity, host fish availability, reproduction/recruitment, survival, occupied habitat, and abundance) is anticipated, and the focal area would maintain current population resiliency. The population maintains low condition during this time-step (Table C.2.9).

In the **Severe 25-year scenario** (Table C.2.10), moderate declines in water quality, hydrology, substrate, and direct mortality are anticipated while fragmentation and invasive species maintain current condition. Changes in hydrologic conditions attributed to climate change are expected due to reductions in 7-day minimum flows (24%) and summer minimum base flows (36%) (Lafontaine et al. 2019, entire) and increasing water demand. Water quality degradation (due to increasing wastewater returns and concentrations of some pollutants (e.g., ammonia and bacteria) and deleterious effects to basic water chemistry (e.g., dissolved oxygen) that can negatively affect mussels) attributed to the decline in 7-day minimum flows, summer minimum base flows and increasing water demand. With the reduction in base flows, a moderate decline in substrate condition is anticipated as sediments accumulate on mussel beds from a lack of adequate cleansing flows. Direct mortality is expected to increase due to the threats above as well as desiccation and increased exposure to predation during dry periods. As a result of these threats, moderate decline is expected for the water quality, hydrology, and habitat structure/substrate habitat factors as well as all population factors (host fish availability, reproduction/recruitment, survival, occupied habitat and abundance) (Table C.2.11). The focal area would undergo a moderate decline in population resiliency, with extirpation occurring during this time-step.

In the **Severe 50-year scenario** (Table C.2.12), severe declines in water quality, hydrology, substrate, and direct mortality are expected. Fragmentation and invasive species threats maintain current condition. Hydrology threats described in the Severe 25-year scenario intensify, causing cascading threats to the other habitat factors described in the Severe 25-year scenario. Due to the increase in stressors, severe declines in water quality and hydrology and a moderate decline in habitat structure/substrate habitat factors are projected. Severe declines in all population factors are projected as a result, which in turn projected a severe decline in population resiliency (Table C.2.13). Extirpation is anticipated during this time-step.

C.1 Future scenario evaluation criteria for Louisiana Pigtoe and Texas Heelsplitter

Table C.1.1: Louisiana Pigtoe threat matrix definitions used to determine population resiliency model input values. ND indicates not defined.

Habitat Parameters	Significant Conservation/Research	Moderate Improvement	Maintain Current Condition	Moderate Decline	Severe Decline
Condition Value	2	1	0	-1	2
Water Quality Changes	ND	WQ is good or excellent. Point and non-point sources of contaminants within watershed are low. No known contaminant concerns (e.g., dissolved oxygen sufficient, no thermal extremes documented). If available, total dissolved solids (TDS) or other indicators of anthropogenic alteration are stable or decreasing.	WQ is moderately impacted. Point and non-point sources of contaminants within watershed are present at moderate levels. TDS or other indicators of anthropogenic alteration are stable or slightly increasing.	WQ is highly impacted. Point and non-point sources of contaminants within watershed are at high levels. TDS or other indicators of anthropogenic alteration are increasing.	WQ is limiting for aquatic life. Point and non-point sources of contaminants within watershed are at levels that preclude mussel or host fish survival.
Hydrology Changes	ND	Hydrology remains unaltered from natural conditions; fully meets requirements of mussels. No impacts to flow components (subsistence, base, high flow pulses, overbanking) from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activities. Flowing water is present year-round with no recorded zero-flow days, even during droughts.	Hydrology moderately impacted. One or more flow components (subsistence, base, high flow pulses, overbanking) impacted from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activity. Biological and geomorphic functions mostly intact. Extremely high, low, or erratic flows are infrequent.	Hydrology highly impacted. One or more flow components (subsistence, base, high flow pulses, overbanking) severely altered from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activity. Biological and geomorphic functions highly impacted. Extremely high, low, or erratic flows are routine; zero flow days occur. PRMS model estimates less than 20% reduction in flows are considered moderate.	Dry stream bed / zero flow days occur frequently, hydrology severely altered; frequency of high flows and shear stress is sufficient to scour substrate and dislocate mussels; substrates are unstable, resulting in unsuitable habitat for mussels. PRMS model estimates greater than 20% reduction in flows and/or changes to hydrology severe enough to impact survival.
Substrate Changes	ND	Riffle and run habitat common. Substrates are stable. Gravel and cobble substrate sufficient to provide anchoring habitat. Low levels of sedimentation on substrate.	Riffle and run habitat uncommon. Substrates are moderately stable. Gravel and cobble substrate sufficient to provide anchoring habitat with some mobilization of particles and light sedimentation on substrate.	Riffle and run habitat rare. Substrates are highly unstable; habitat eroded, or being buried by mobilized sediments from upstream.	No suitable habitat present.
Fragmentation	ND	No impoundments/ barriers limiting mobility of host fish.	New or existing impoundments/ barriers moderately reducing mobility of host fish and impacting dispersal range of glochidia.	New or existing impoundments/ barriers severely reducing mobility of host fish and impacting dispersal range of glochidia.	New or existing impoundments/ barriers has limited mobility of host fish and impacted dispersal of glochidia at level causing extirpation/extinction.
Direct Mortality	ND	Predation, collection, or other actions resulting in direct mortality are not impacting populations.	Predation, collection, or other actions resulting in direct mortality are moderately impacting populations.	Predation, collection, or other actions resulting in direct mortality are severely impacting populations.	Predation, collection, or other actions resulting in mortality have caused extirpation/extinction
Invasive Species	ND	No invasive species present.	Invasive species moderately impacting populations.	Invasive species highly impacting populations.	Invasive species limiting to mussels or host fish. Invasive species present and severely impacting populations.

Table C.1.2: Texas Heelsplitter threat matrix definitions used to determine population resiliency model input values. ND indicates not defined.

Habitat Parameters	Significant Improvement	Moderate Improvement	Maintain Current Condition	Moderate Decline	Severe Decline
Condition Value	2	1	0	-1	2
Water Quality Changes	ND	WQ is good or excellent. Point and non-point sources of contaminants within watershed are low. No known contaminant concerns (e.g., dissolved oxygen sufficient, no thermal extremes documented). If available, total dissolved solids (TDS) or other indicators of anthropogenic alteration are stable or decreasing.	WQ is moderately impacted. Point and non-point sources of contaminants within watershed are present at moderate levels. TDS or other indicators of anthropogenic alteration are stable or slightly increasing.	WQ is highly impacted. Point and non-point sources of contaminants within watershed are at high levels. TDS or other indicators of anthropogenic alteration are increasing.	WQ is limiting for aquatic life. Point and non-point sources of contaminants within watershed are at levels that preclude mussel or host fish survival.
Hydrology Changes	ND	Hydrology remains unaltered from natural conditions; fully meets requirements of mussels. No impacts to flow components (subsistence, base, high flow pulses, overbanking) from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activities. Flowing water is present year-round with no recorded zero-flow days, even during droughts.	Hydrology moderately impacted. One or more flow components (subsistence, base, high flow pulses, overbanking) impacted from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activities. Biological and geomorphic functions mostly intact. Occupied reservoirs maintain stable water levels or experience moderate fluctuations. Extremely high, low, or erratic flows are infrequent.	Hydrology highly impacted. One or more flow components (subsistence, base, high flow pulses, overbanking) severely altered from impoundments, reservoirs, diversions, groundwater extraction, or other anthropogenic activities. Biological and/or geomorphic functions highly impacted. Frequency and magnitude of water fluctuations in occupied reservoirs is high. Extremely high, low, or erratic flows are routine; zero flow days occur. PRMS model estimates less than 20% reduction in flows are considered moderate.	Extremely high, low, and/or erratic flows are frequent, resulting in unsuitable habitat for mussels. Large magnitude reservoir drawdowns occur frequently. PRMS model estimates greater than 20% reduction in flows are considered significant and/or changes to hydrology severe enough to impact survival.
Substrate Changes	ND	Pool and backwater habitats common. Stable mud, sand, and silt substrates sufficient to provide anchoring habitat. Low levels of sedimentation on substrate.	Pool and backwater habitats uncommon. Mud, sand, and silt substrates moderately stable, providing anchoring habitat with some mobilization of particles and light sedimentation on substrate.	Pool and backwater habitat rare; substrates highly unstable, habitat eroded, or being buried by mobilized sediments from upstream.	No suitable habitat present.
Fragmentation	ND	No impoundments/ barriers limiting mobility of host fish.	New or existing impoundments/ barriers moderately reducing mobility of host fish and impacting dispersal range of glochidia.	New or existing impoundments/ barriers severely reducing mobility of host fish and impacting dispersal range of glochidia.	New or existing impoundments/ barriers has limited mobility of host fish and impacted dispersal of glochidia at level causing extirpation/extinction.
Direct Mortality	ND	Predation, collection, or other actions resulting in direct mortality are not impacting populations.	Predation, collection, or other actions resulting in direct mortality are moderately impacting populations.	Predation, collection, or other actions resulting in direct mortality are severely impacting populations.	Predation, collection, or other actions resulting in direct mortality have caused extirpation/extinction
Invasive Species	ND	No invasive species present.	Invasive species moderately impacting populations.	Invasive species highly impacting populations.	Invasive species limiting to mussels or host fish. Invasive species present and severely impacting populations.

C.2 Future condition tables by scenario and time step for Louisiana Pigtoe and Texas Heelsplitter

Table C.2.1: Population resiliency model input and output definitions for all scenarios and time steps

Forecasted Change in State	Input Value	Output	Change to Population Resiliency
Significant improvement	2	44 ≥ Resiliency > 22	Significant improvement in population resiliency
Moderate improvement	1	22 ≥ Resiliency > 0	Moderate improvement in population resiliency
Maintain Current Condition	0	Resiliency= 0	Maintain current population resiliency
Moderate Decline	-1	0 > Resiliency ≥ (-22)	Moderate decline in population resiliency
Severe Decline	-2	(22) > Resiliency > (44)	Severe decline in population resiliency

Resiliency = (-45) indicates two of the three Habitat Factors are severely declining, therefore Resiliency = severe decline.

Table C.2.2: Scenario 1 – moderate increase in stressors, 10 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality Changes	Hydrology Changes	Substrate Changes	Fragmentation	Direct Mortality	Invasive Species
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	-1	-1	-1	-1	0
	Neches	Neches R/BA Steinhagen	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	-1	0
	Trinity	Grapevine LK	-1	-1	0	0	-1	-1
		Trinity R/Livingston	-1	-1	-1	0	-1	0
Louisiana Pigtoe	Red	Little R/Rolling FK	0	-1	-1	0	0	0
		Cossatot R	0	-1	0	0	0	0
		Saline R (Little)	0	-1	0	0	0	0
		Lower Little R	0	-1	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	-1	0	0	0	0
	Calcasieu	Upper Calcasieu R	0	-1	0	1	0	0
	Pearl	Pearl R	-1	-1	-1	-1	-1	0
	Sabine	Sabine R	-1	-1	-1	-1	-1	0
		Bayou Anacoco	0	-1	0	0	0	0
	Neches	Angelina R	-1	-2	-2	-1	-1	0
		Neches R	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	0	0
	San Jacinto	E FK San Jacinto R	-1	-1	-1	0	-1	0

Table C.2.3: Scenario 1 – moderate increase in stressors, 10 year time step stressors evaluation model output

			Habitat Factors			Population Factors				
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality	Hydrology	Habitat Structure/ Substrate	Host Fish Availability	Reproduction/ Recruitment	Survival	Occupied Habitat	Abundance	Change to Resiliency
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	0	-1	-2	-4	-3	-2	-9	-11
	Neches	Neches R/BA Steinhagen	0	0	0	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0	0	0	0
	Trinity	Grapevine LK	0	0	0	0	0	0	0	0	0
		Trinity R/Livingston	0	0	0	0	0	0	0	0	0
Louisiana Pigtoe	Red	Little R/Rolling FK	0	0	0	0	0	0	0	0	0
		Cossatot R	0	0	0	0	0	0	0	0	0
		Saline R (Little)	0	0	0	0	0	0	0	0	0
		Lower Little R	0	0	0	0	0	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	0	0	0	0	0	0	0	0
	Calcasieu	Upper Calcasieu R	0	1	1	2	4	2	2	8	10
	Pearl	Pearl R	0	0	0	0	0	0	0	0	0
	Sabine	Sabine R	-1	0	-1	-2	-4	-3	-2	-9	-11
		Bayou Anacoco	0	-1	-1	-2	-4	-2	-2	-8	-10
	Neches	Angelina R	0	0	-1	-1	-2	-2	-1	-5	-6
		Neches R	0	0	0	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0	0	0	0
	San Jacinto	E FK San Jacinto R	0	0	0	0	0	0	0	0	0

Table C.2.4: Scenario 1 – moderate increase in stressors, 25 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality Changes	Hydrology Changes	Substrate Changes	Fragmentation	Direct Mortality	Invasive Species
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	-1	-1	-1	-1	0
	Neches	Neches R/BA Steinhagen	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	-1	0
	Trinity	Grapevine LK	-1	-1	0	0	-1	-1
		Trinity R/Livingston	-1	-1	-1	0	-1	0
Louisiana Pigtoe	Red	Little R/Rolling FK	0	-1	-1	0	0	0
		Cossatot R	0	-1	0	0	0	0
		Saline R (Little)	0	-1	0	0	0	0
		Lower Little R	0	-1	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	-1	0	0	0	0
	Calcasieu	Upper Calcasieu R	0	-1	0	1	0	0
	Pearl	Pearl R	-1	-1	-1	-1	-1	0
	Sabine	Sabine R	-1	-1	-1	-1	-1	0
		Bayou Anacoco	0	-1	0	0	0	0
	Neches	Angelina R	-1	-2	-2	-1	-1	0
		Neches R	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	0	0
	San Jacinto	E FK San Jacinto R	-1	-1	-1	0	-1	0

Table C.2.5: Scenario 1 – moderate increase in stressors, 25 year time step stressors evaluation model output

			Habitat Factors			Population Factors					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality	Hydrology	Habitat Structure/ Substrate	Host Fish Availability	Reproduction/ Recruitment	Survival	Occupied Habitat	Abundance	Change to Resiliency
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	-1	-2	-4	-8	-5	-4	-17	-21
	Neches	Neches R/BA Steinhagen	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-1	0	-2	-4	-3	-2	-9	-11
	Trinity	Grapevine LK	-1	-1	0	-2	-4	-4	-2	-10	-12
		Trinity R/Livingston	-1	-1	-1	-3	-6	-4	-3	-13	-16
Louisiana Pigtoe	Red	Little R/Rolling FK	0	-1	-1	-2	-4	-2	-2	-8	-10
		Cossatot R	0	-1	0	-1	-2	-1	-1	-4	-5
		Saline R (Little)	0	-1	0	-1	-2	-1	-1	-4	-5
		Lower Little R	0	-1	0	-1	-2	-1	-1	-4	-5
	Big Cypress	Big Cypress Bayou	0	-1	0	-1	-2	-1	-1	-4	-5
	Calcasieu	Upper Calcasieu R	0	-1	1	0	0	0	0	0	0
	Pearl	Pearl R	-1	-1	-2	-4	-8	-5	-4	-17	-21
	Sabine	Sabine R	-1	-1	-2	-4	-8	-5	-4	-17	-21
		Bayou Anacoco	0	-1	0	-1	-2	-1	-1	-4	-5
	Neches	Angelina R	-1	-2	-3	-6	-12	-7	-6	-25	-31
		Neches R	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-1	0	-2	-4	-2	-2	-8	-10
	San Jacinto	E FK San Jacinto R	-1	-1	-1	-3	-6	-4	-3	-13	-16

Table C.2.6: Scenario 1 – moderate increase in stressors, 50 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality Changes	Hydrology Changes	Substrate Changes	Fragmentation	Direct Mortality	Invasive Species
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	-2	-1	-1	-1	0
	Neches	Neches R/BA Steinhagen	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-2	0	0	-1	0
	Trinity	Grapevine LK	-1	-2	0	0	-1	-1
		Trinity R/Livingston	-2	-2	-1	0	-2	0
Louisiana Pigtoe	Red	Little R/Rolling FK	-1	-2	-1	-1	-1	0
		Cossatot R	-1	-2	-1	0	0	0
		Saline R (Little)	-1	-2	-1	0	0	0
		Lower Little R	0	-1	0	0	0	0
	Big Cypress	Big Cypress Bayou	-1	-2	0	0	-1	0
	Calcasieu	Upper Calcasieu R	-1	-2	-1	0	0	0
	Pearl	Pearl R	-1	-1	-1	-1	-1	0
	Sabine	Sabine R	-1	-2	-1	-1	-1	0
		Bayou Anacoco	-1	-2	-1	0	0	0
	Neches	Angelina R	-2	-2	-2	-2	-1	0
		Neches R	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-2	0	0	0	0
	San Jacinto	E FK San Jacinto R	-2	-2	-2	0	-1	0

Table C.2.7: Scenario 1 – moderate increase in stressors, 50 year time step stressors evaluation model output

			Habitat Factors			Population Factors					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality	Hydrology	Habitat Structure/ Substrate	Host Fish Availability	Reproduction/ Recruitment	Survival	Occupied Habitat	Abundance	Change to Resiliency
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	-2	-2	-5	-10	-6	-5	-21	-26
	Neches	Neches R/BA Steinhagen	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-2	0	-3	-6	-4	-3	-13	-16
	Trinity	Grapevine LK	-1	-2	0	-3	-6	-5	-3	-14	-17
		Trinity R/Livingston	-2	-2	-1	-5	-10	-7	-5	-22	-45
Louisiana Pigtoe	Red	Little R/Rolling FK	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Cossatot R	-1	-2	-1	-4	-8	-4	-4	-16	-20
		Saline R (Little)	-1	-2	-1	-4	-8	-4	-4	-16	-20
		Lower Little R	0	-1	0	-1	-2	-1	-1	-4	-5
	Big Cypress	Big Cypress Bayou	-1	-2	0	-3	-6	-4	-3	-13	-16
	Calcasieu	Upper Calcasieu R	-1	-2	-1	-4	-8	-4	-4	-16	-20
	Pearl	Pearl R	-1	-1	-2	-4	-8	-5	-4	-17	-21
	Sabine	Sabine R	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Bayou Anacoco	-1	-2	-1	-4	-8	-4	-4	-16	-20
	Neches	Angelina R	-2	-2	-4	-8	-16	-9	-8	-33	-45
		Neches R	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-2	0	-3	-6	-3	-3	-12	-15
	San Jacinto	E FK San Jacinto R	-2	-2	-2	-6	-12	-7	-6	-25	-45

Table C.2.8: Scenario 2 – severe increase in stressors, 10 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality Changes	Hydrology Changes	Substrate Changes	Fragmentation	Direct Mortality	Invasive Species
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	0	0	-1	-1	0
	Neches	Neches R/BA Steinhagen	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0
	Trinity	Grapevine LK	0	0	0	0	0	0
		Trinity R/Livingston	0	0	0	0	0	0
Louisiana Pigtoe	Red	Little R/Rolling FK	0	0	0	0	0	0
		Cossatot R	0	0	0	0	0	0
		Saline R (Little)	0	0	0	0	0	0
		Lower Little R	0	0	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	0	0	0	0	0
	Calcasieu	Upper Calcasieu R	0	1	0	1	0	0
	Pearl	Pearl R	0	0	0	0	0	0
	Sabine	Sabine R	-1	0	0	-1	-1	0
		Bayou Anacoco	0	-1	-1	0	0	0
	Neches	Angelina R	0	0	-1	0	-1	0
		Neches R	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0
	San Jacinto	E FK San Jacinto R	0	0	0	0	0	0

Table C.2.9: Scenario 2 – severe increase in stressors, 10 year time step stressors evaluation model output

			Habitat Factors			Population Factors					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality	Hydrology	Habitat Structure/ Substrate	Host Fish Availability	Reproduction/ Recruitment	Survival	Occupied Habitat	Abundance	Change to Resiliency
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-1	0	-1	-2	-4	-3	-2	-9	-11
	Neches	Neches R/BA Steinhagen	0	0	0	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0	0	0	0
	Trinity	Grapevine LK	0	0	0	0	0	0	0	0	0
		Trinity R/Livingston	0	0	0	0	0	0	0	0	0
Louisiana Pigtoe	Red	Little R/Rolling FK	0	0	0	0	0	0	0	0	0
		Cossatot R	0	0	0	0	0	0	0	0	0
		Saline R (Little)	0	0	0	0	0	0	0	0	0
		Lower Little R	0	0	0	0	0	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	0	0	0	0	0	0	0	0
	Calcasieu	Upper Calcasieu R	0	1	1	2	4	2	2	8	10
	Pearl	Pearl R	0	0	0	0	0	0	0	0	0
	Sabine	Sabine R	-1	0	-1	-2	-4	-3	-2	-9	-11
		Bayou Anacoco	0	-1	-1	-2	-4	-2	-2	-8	-10
	Neches	Angelina R	0	0	-1	-1	-2	-2	-1	-5	-6
		Neches R	0	0	0	0	0	0	0	0	0
		Lower Neches R	0	0	0	0	0	0	0	0	0
	San Jacinto	E FK San Jacinto R	0	0	0	0	0	0	0	0	0

Table C.2.10: Scenario 2 – severe increase in stressors, 25 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	**Representation Areas**	**POPULATIONS (Focal Area)**	**Water Quality Changes**	**Hydrology Changes**	**Substrate Changes**	**Fragmentation**	**Direct Mortality**	**Invasive Species**
Texas Heelsplitter	**Sabine**	Sabine R/Toledo Bend	-1	-2	-1	-1	-1	0
	Neches	Neches R/BA Steinhagen	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	-1	0
	Trinity	Grapevine LK	-1	-1	0	0	-1	-1
		Trinity R/Livingston	-1	-1	-1	0	-1	0
Louisiana Pigtoe	**Red**	Little R/Rolling FK	0	-1	-1	0	0	0
		Cossatot R	0	-1	0	0	0	0
		Saline R (Little)	0	-1	0	0	0	0
		Lower Little R	0	-1	0	0	0	0
	Big Cypress	Big Cypress Bayou	0	-1	0	0	0	0
	Calcasieu	Upper Calcasieu R	-1	-2	0	1	-1	0
	Pearl	Pearl R	-1	-1	-1	-1	-1	0
	Sabine	Sabine R	-1	-2	-1	-1	-1	0
		Bayou Anacoco	-1	-2	0	0	-1	0
	Neches	Angelina R	-1	-2	-2	-1	-1	0
		Neches R	-1	-2	-1	-1	-1	0
		Lower Neches R	-1	-1	0	0	0	0
	San Jacinto	E FK San Jacinto R	-1	-1	-1	0	-1	0

Table C.2.11: Scenario 2 – severe increase in stressors, 25 year time step stressors evaluation model output

			Habitat Factors			Population Factors					
SPECIES	**Representation Areas**	**POPULATIONS (Focal Area)**	**Water Quality**	**Hydrology**	**Habitat Structure/ Substrate**	**Host Fish Availability**	**Reproduction/ Recruitment**	**Survival**	**Occupied Habitat**	**Abundance**	**Change to Resiliency**
Texas Heelsplitter	**Sabine**	Sabine R/Toledo Bend	-1	-2	-2	-5	-10	-6	-5	-21	-26
	Neches	Neches R/BA Steinhagen	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-1	0	-2	-4	-3	-2	-9	-11
	Trinity	Grapevine LK	-1	-1	0	-2	-4	-4	-2	-10	-12
		Trinity R/Livingston	-1	-1	-1	-3	-6	-4	-3	-13	-16
Louisiana Pigtoe	**Red**	Little R/Rolling FK	0	-1	-1	-2	-4	-2	-2	-8	-10
		Cossatot R	0	-1	0	-1	-2	-1	-1	-4	-5
		Saline R (Little)	0	-1	0	-1	-2	-1	-1	-4	-5
		Lower Little R	0	-1	0	-1	-2	-1	-1	-4	-5
	Big Cypress	Big Cypress Bayou	0	-1	0	-1	-2	-1	-1	-4	-5
	Calcasieu	Upper Calcasieu R	-1	-2	1	-2	-4	-3	-2	-9	-11
	Pearl	Pearl R	-1	-1	-2	-4	-8	-5	-4	-17	-21
	Sabine	Sabine R	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Bayou Anacoco	-1	-2	0	-3	-6	-4	-3	-13	-16
	Neches	Angelina R	-1	-2	-3	-6	-12	-7	-6	-25	-31
		Neches R	-1	-2	-2	-5	-10	-6	-5	-21	-26
		Lower Neches R	-1	-1	0	-2	-4	-2	-2	-8	-10
	San Jacinto	E FK San Jacinto R	-1	-1	-1	-3	-6	-4	-3	-13	-16

Table C.2.12: Scenario 2 – severe increase in stressors, 50 year time step stressors evaluation model input

ETX FWM Scenario Development			Threats					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality Changes	Hydrology Changes	Substrate Changes	Fragmentation	Direct Mortality	Invasive Species
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-2	-2	-1	-1	-1	0
	Neches	Neches R/BA Steinhagen	-2	-2	-1	-1	-1	0
		Lower Neches R	-2	-2	0	0	-1	0
	Trinity	Grapevine LK	-1	-2	0	0	-1	-1
		Trinity R/Livingston	-2	-2	-1	0	-2	0
Louisiana Pigtoe	Red	Little R/Rolling FK	-2	-2	-1	-1	-1	0
		Cossatot R	-2	-2	-1	0	0	0
		Saline R (Little)	-2	-2	-1	-1	0	0
		Lower Little R	-2	-2	0	0	0	0
	Big Cypress	Big Cypress Bayou	-2	-2	0	0	-1	0
	Calcasieu	Upper Calcasieu R	-2	-2	-1	-1	0	0
	Pearl	Pearl R	-1	-2	-1	-1	-1	0
	Sabine	Sabine R	-2	-2	-1	-1	-1	0
		Bayou Anacoco	-2	-2	-1	0	0	0
	Neches	Angelina R	-2	-2	-2	-2	-1	0
		Neches R	-2	-2	-1	-1	-1	0
		Lower Neches R	-2	-2	0	0	0	0
	San Jacinto	E FK San Jacinto R	-2	-2	-2	0	-2	0

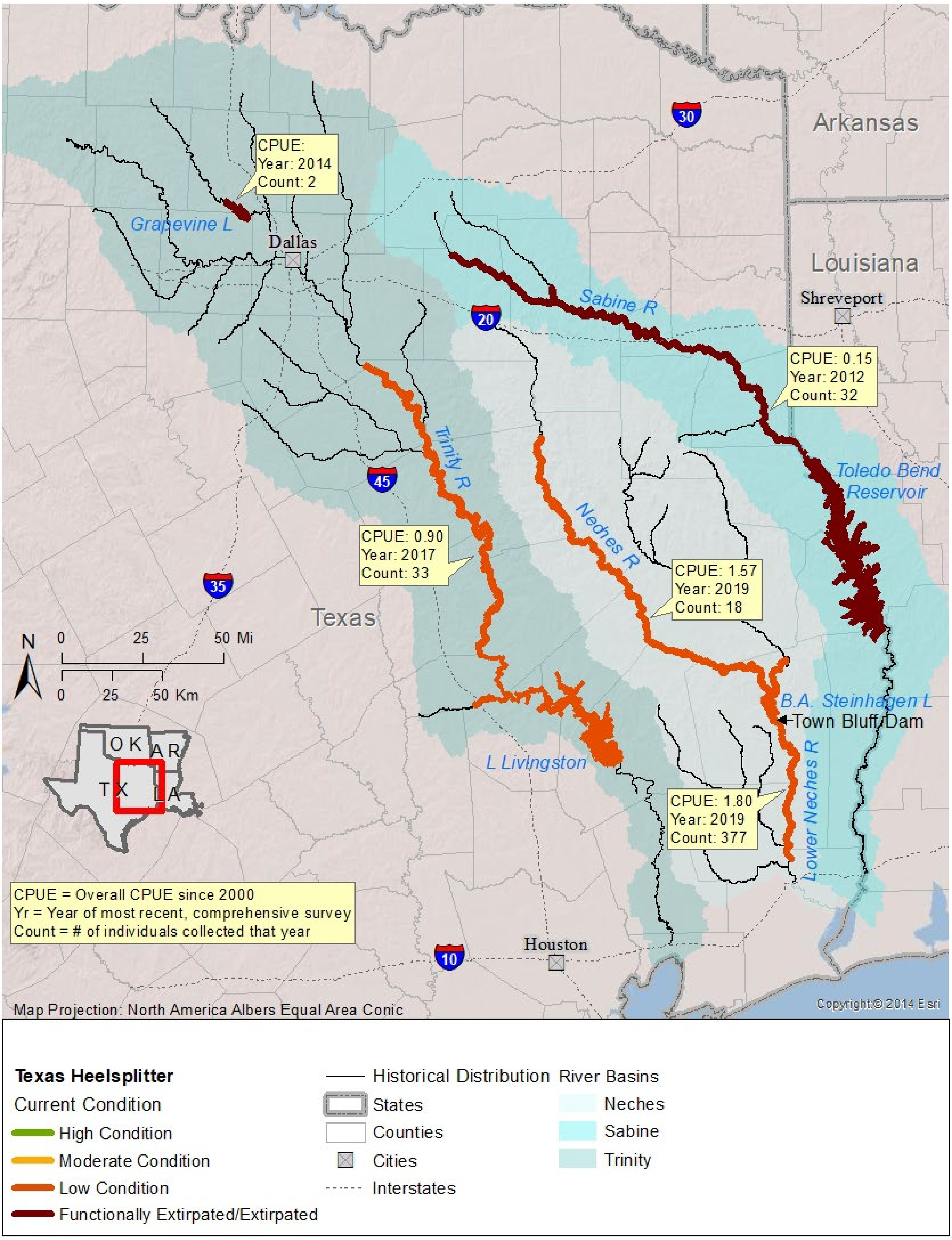
Table C.2.13: Scenario 2 – severe increase in stressors, 50 year time step stressors evaluation model output

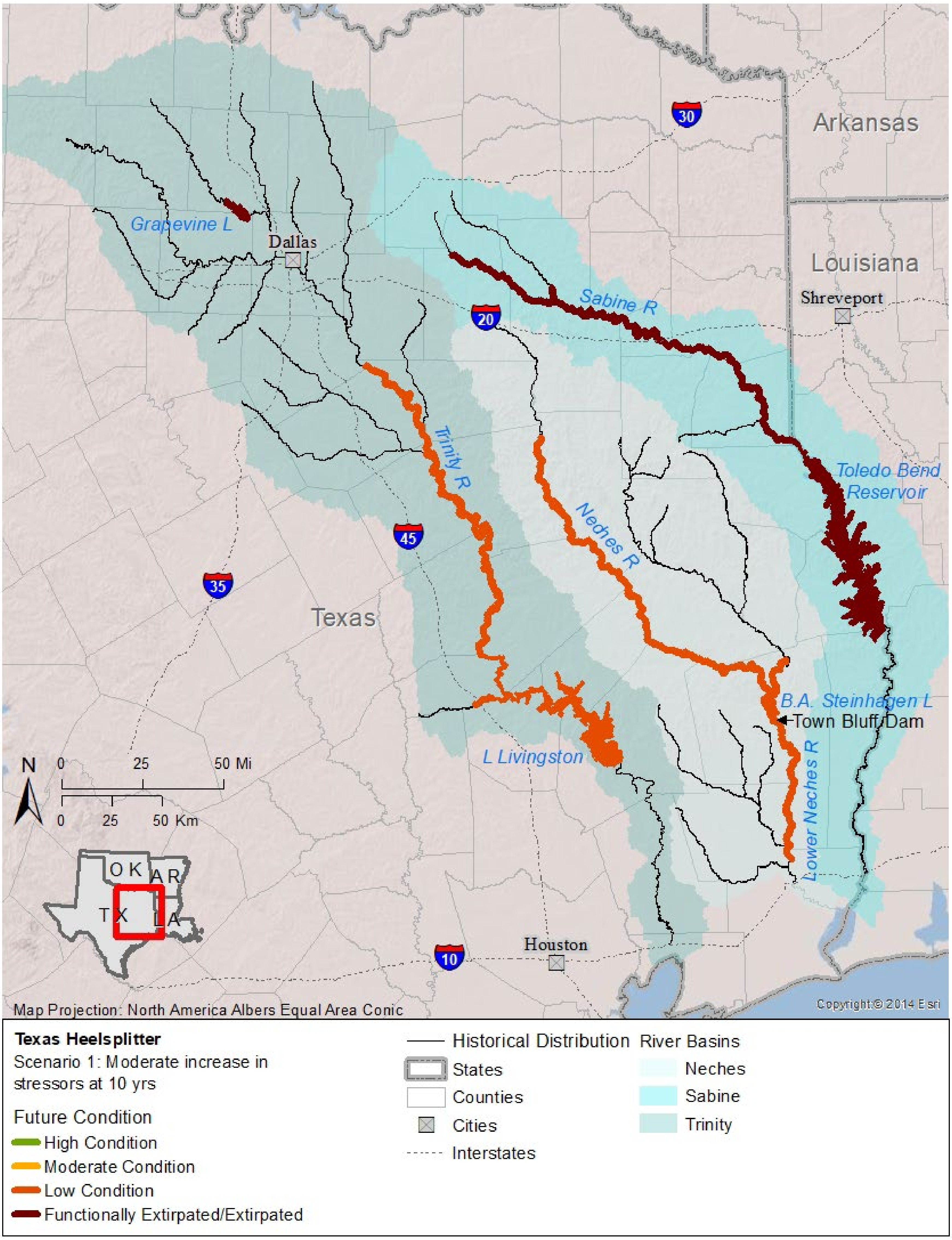
			Habitat Factors			Population Factors					
SPECIES	Representation Areas	POPULATIONS (Focal Area)	Water Quality	Hydrology	Habitat Structure/ Substrate	Host Fish Availability	Reproduction/ Recruitment	Survival	Occupied Habitat	Abundance	Change to Resiliency
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	-2	-2	-2	-6	-12	-7	-6	-25	-45
	Neches	Neches R/BA Steinhagen	-2	-2	-2	-6	-12	-7	-6	-25	-45
		Lower Neches R	-2	-2	0	-4	-8	-5	-4	-17	-45
	Trinity	Grapevine LK	-1	-2	0	-3	-6	-5	-3	-14	-17
		Trinity R/Livingston	-2	-2	-1	-5	-10	-7	-5	-22	-45
Louisiana Pigtoe	Red	Little R/Rolling FK	-2	-2	-2	-6	-12	-7	-6	-25	-45
		Cossatot R	-2	-2	-1	-5	-10	-5	-5	-20	-45
		Saline R (Little)	-2	-2	-2	-6	-12	-6	-6	-24	-45
		Lower Little R	-2	-2	0	-4	-8	-4	-4	-16	-45
	Big Cypress	Big Cypress Bayou	-2	-2	0	-4	-8	-5	-4	-17	-45
	Calcasieu	Upper Calcasieu R	-2	-2	-2	-6	-12	-6	-6	-24	-45
	Pearl	Pearl R	-1	-2	-2	-5	-10	-6	-5	-21	-26
	Sabine	Sabine R	-2	-2	-2	-6	-12	-7	-6	-25	-45
		Bayou Anacoco	-2	-2	-1	-5	-10	-5	-5	-20	-45
	Neches	Angelina R	-2	-2	-4	-8	-16	-9	-8	-33	-45
		Neches R	-2	-2	-2	-6	-12	-7	-6	-25	-45
		Lower Neches R	-2	-2	0	-4	-8	-4	-4	-16	-45
	San Jacinto	E FK San Jacinto R	-2	-2	-2	-6	-12	-8	-6	-26	-45

Table C.2.14: Future population condition for Louisiana Pigtoe and Texas Heelsplitter resiliency under each scenario and time-step

Future Scenarios				Scenario 1: Moderate Increase in Stessors - (RCP 4.5)			Scenario 2: Severe Increase in Stressors - (RCP 8.5)		
SPECIES	**Representation Areas**	**POPULATIONS (Focal Areas)**	**Current Condition**	10-yrs	25-yrs	50-yrs	10-yrs	25-yrs	50-yrs
Texas Heelsplitter	Sabine	Sabine R/Toledo Bend	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
	Neches	Neches R/BA Steinhagen	Low	Low	Low	Low	Low	Low	Extirpated
		Lower Neches R	Low	Low	Low	Extirpated	Low	Low	Extirpated
	Trinity	Grapevine LK	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
		Trinity R/Livingston	low	Low	Low	Extirpated	Low	Low	Extirpated
Louisiana Pigtoe	Red	Little R/Rolling FK	Moderate	Moderate	Low	Low	Moderate	Low	Low
		Cossatot R	High	High	High	Moderate	High	High	Low
		Saline R (Little)	Moderate	Moderate	Moderate	Low	Moderate	Low	Low
		Lower Little R	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
	Big Cypress	Big Cypress Bayou	Moderate	Moderate	Moderate	Low	Moderate	Moderate	Low
	Calcasieu	Upper Calcasieu R	Low	Low	Low	Extirpated	Low	Low	Extirpated
	Pearl	Pearl R	Low	Low	Low	Low	Low	Low	Extirpated
	Sabine	Sabine R	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated	Extirpated
		Bayou Anacoco	Moderate	Low	Moderate	Low	Low	Moderate	Extirpated
	Neches	Angelina R	Low	Low	Low	Extirpated	Low	Low	Extirpated
		Neches R	High	High	Low	Low	High	Low	Low
		Lower Neches R	Low	Low	Low	Low	Low	Low	Extirpated
	San Jacinto	E FK San Jacinto R	Low	Low	Low	Extirpated	Low	Extirpated	Extirpated

Figure C.1 Large-sized Current and Future Population Condition Maps for Texas Heelsplitter

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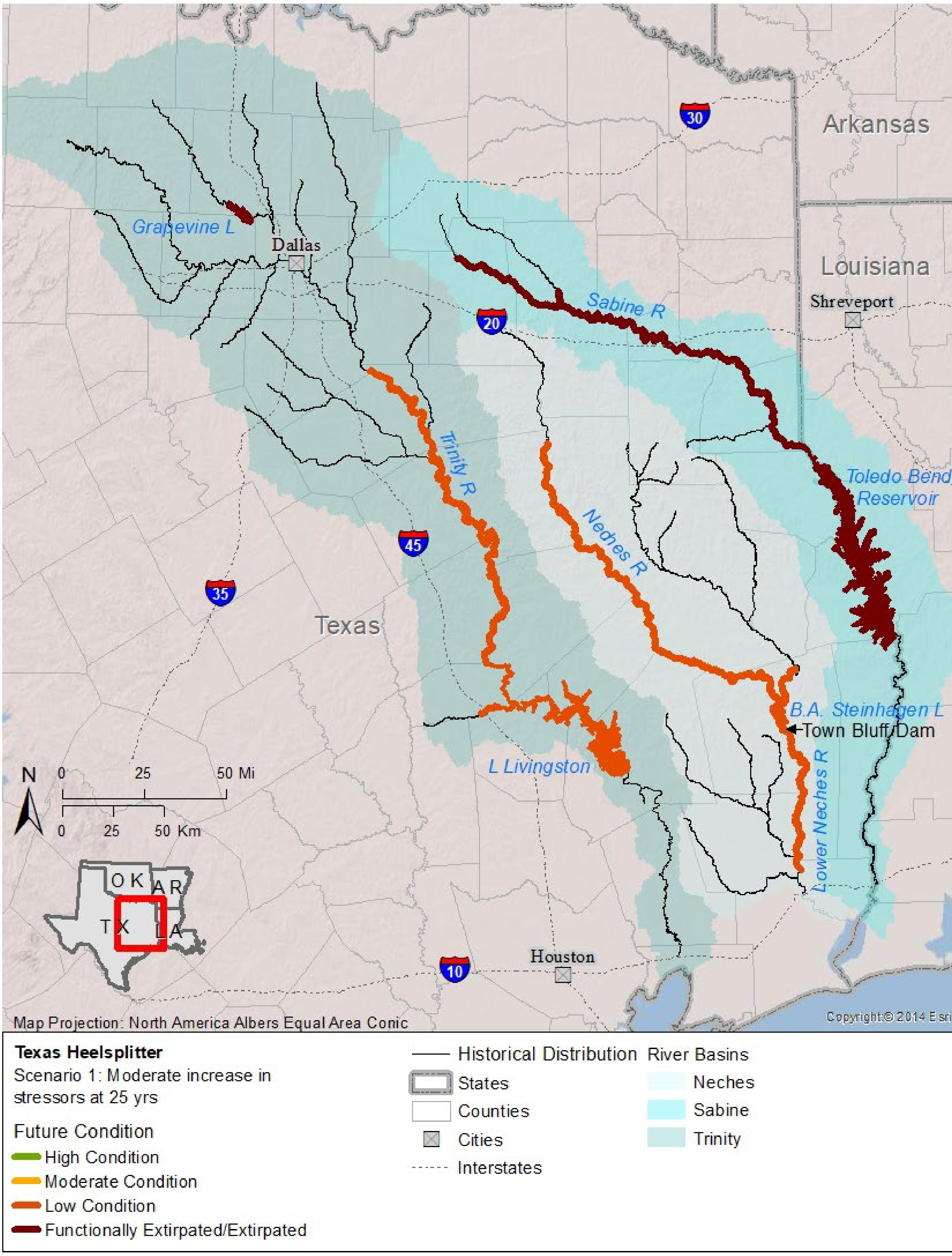
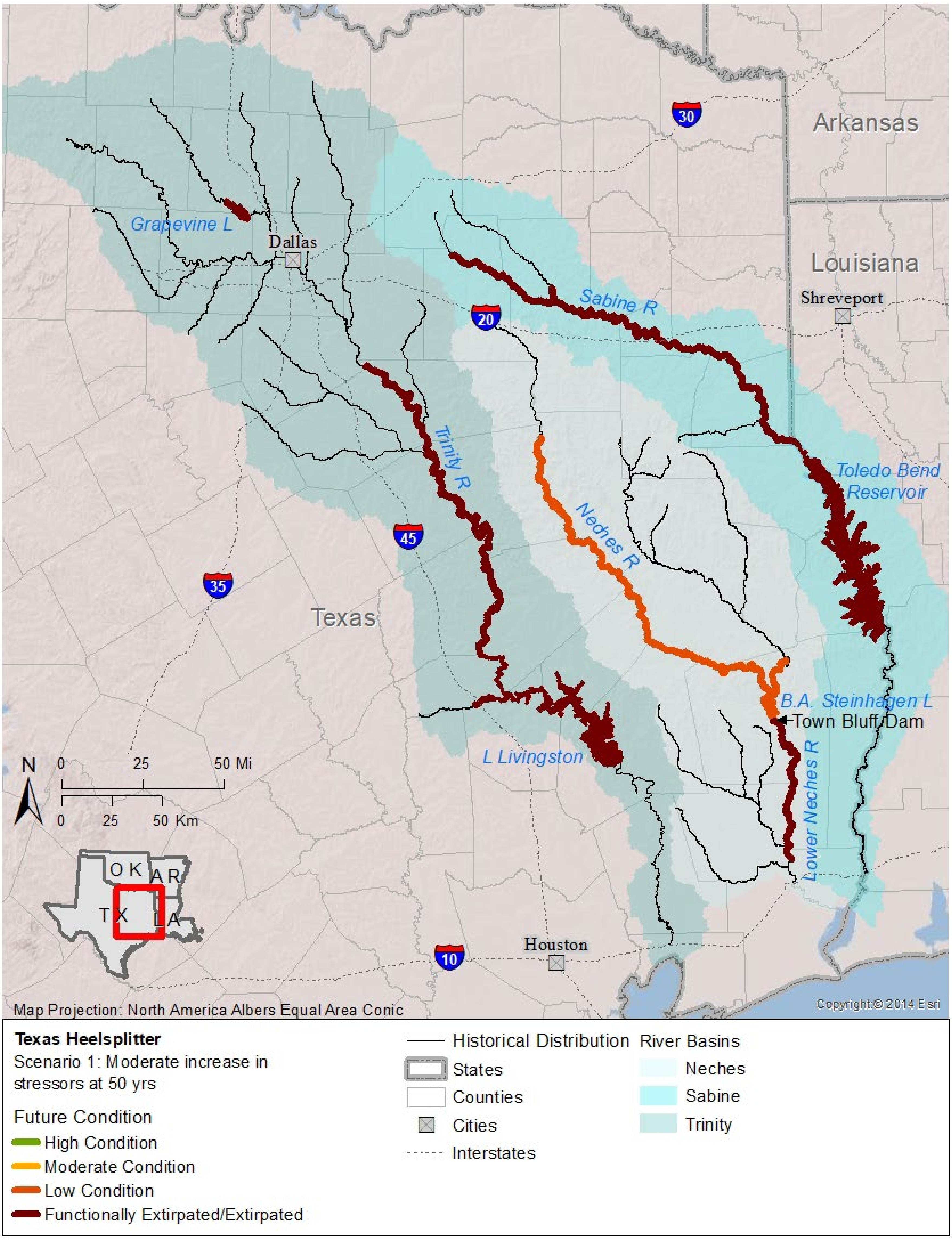
Louisiana
Shreveport
Dallas
Grapevine L
Sabine R
Trinity R
Neches R
Toledo Bend Reservoir
B.A. Steinhagen L
Town Bluff Dam
L Livingston
Lower Neches R
Texas
Houston
30
20
45
35
10
N
0 25 50 Mi
0 25 50 Km
OK
AR
TX
LA
Map Projection: North America Albers Equal Area Conic
Copyright:© 2014 Esri
Texas Heelsplitter
Scenario 1: Moderate increase in stressors at 10 yrs
Future Condition
High Condition
Moderate Condition
Low Condition
Functionally Extirpated/Extirpated
Historical Distribution
States
Counties
Cities
Interstates
River Basins
Neches
Sabine
Trinity



Arkansas
Louisiana
Shreveport
Grapevine L
Dallas
Sabine R
Trinity R
Neches R
Toledo Bend Reservoir
Texas
B.A. Steinhagen L
Town Bluff Dam
L Livingston
Lower Neches R
Houston
30
20
45
35
10
N
0 25 50 Mi
0 25 50 Km
OK
AR
TX
LA
Map Projection: North America Albers Equal Area Conic
Copyright:© 2014 Esri
Texas Heelsplitter
Scenario 1: Moderate increase in stressors at 25 yrs
Future Condition
High Condition
Moderate Condition
Low Condition
Functionally Extirpated/Extirpated
Historical Distribution
States
Counties
Cities
Interstates
River Basins
Neches
Sabine
Trinity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Louisiana
Shreveport
Dallas
Grapevine L
Sabine R
Trinity R
Neches R
Toledo Bend Reservoir
Texas
B.A. Steinhagen L
Town Bluff Dam
L Livingston
Lower Neches R
Houston
30
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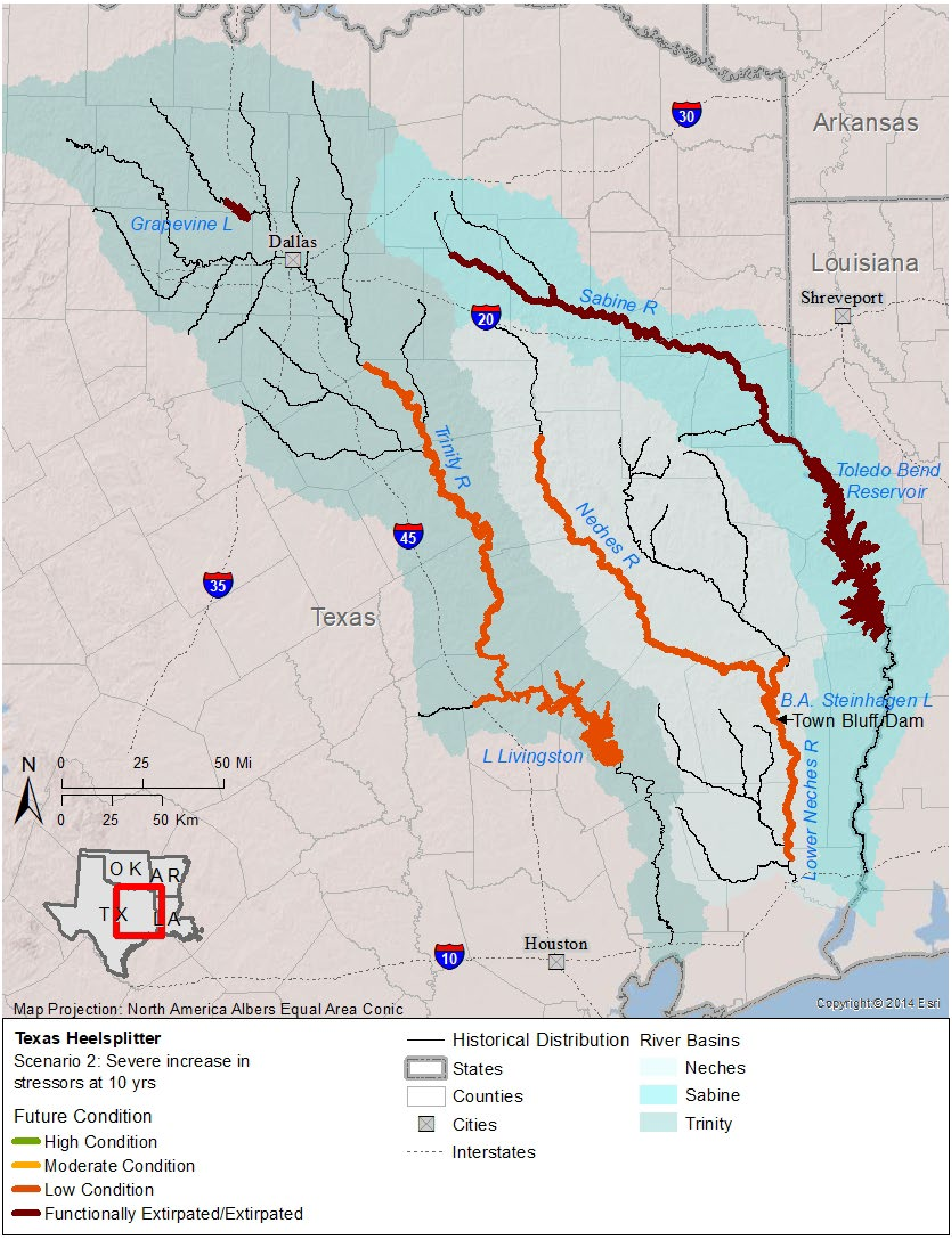
0 25 50 Mi
0 25 50 Km
OK
AR
TX
LA
Map Projection: North America Albers Equal Area Conic
Copyright:© 2014 Esri
Texas Heelsplitter
Scenario 1: Moderate increase in stressors at 50 yrs
Future Condition
High Condition
Moderate Condition
Low Condition
Functionally Extirpated/Extirpated
Historical Distribution
States
Counties
Cities
Interstates
River Basins
Neches
Sabine
Trinity



Arkansas
Louisiana
Shreveport
Grapevine L
Dallas
Sabine R
Trinity R
Neches R
Toledo Bend Reservoir
Texas
B.A. Steinhagen L
Town Bluff Dam
L Livingston
Lower Neches R
Houston
30
20
45
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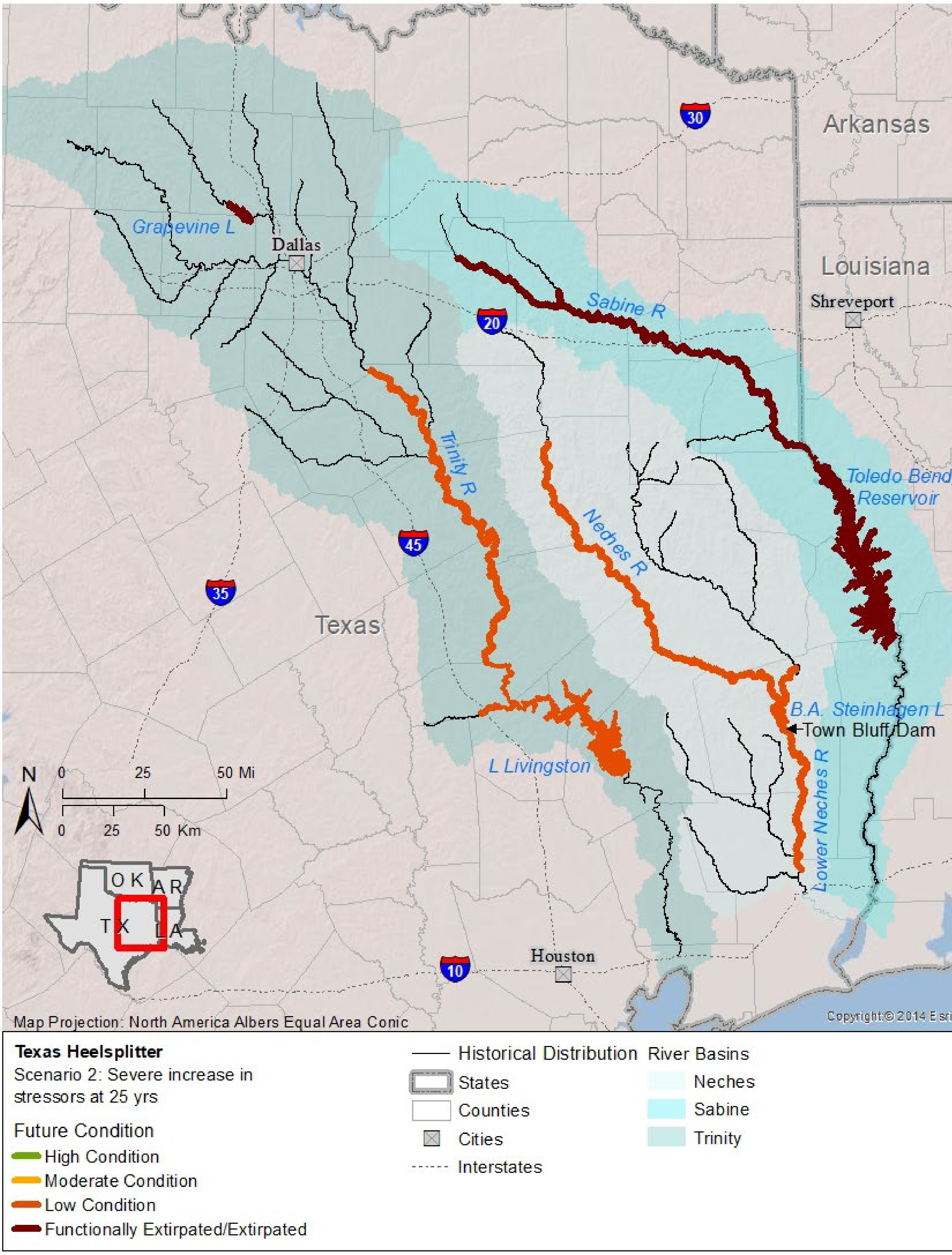
10
N
0 25 50 Mi
0 25 50 Km
OK
AR
TX
LA
Map Projection: North America Albers Equal Area Conic
Copyright:© 2014 Esri
Texas Heelsplitter
Scenario 2: Severe increase in stressors at 10 yrs
Future Condition
High Condition
Moderate Condition
Low Condition
Functionally Extirpated/Extirpated
Historical Distribution
States
Counties
Cities
Interstates
River Basins
Neches
Sabine
Trinity



Arkansas
Louisiana
Shreveport
Dallas
Grapevine L
Sabine R
Trinity R
Neches R
Toledo Bend Reservoir
Texas
B.A. Steinhagen L
Town Bluff Dam
L Livingston
Lower Neches R
Houston
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0 25 50 Km
OK
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TX
LA
Map Projection: North America Albers Equal Area Conic
Copyright:© 2014 Esri
Texas Heelsplitter
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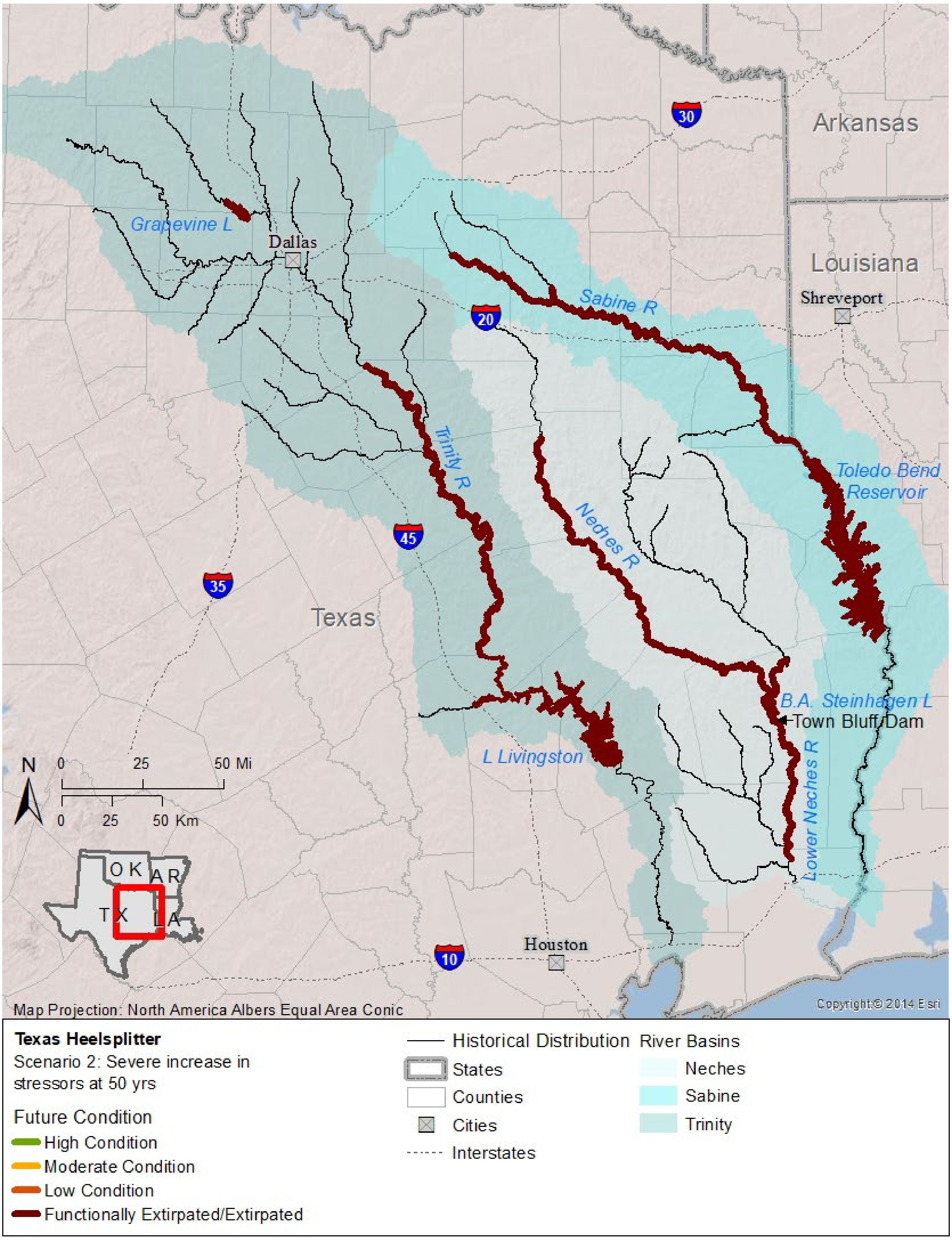
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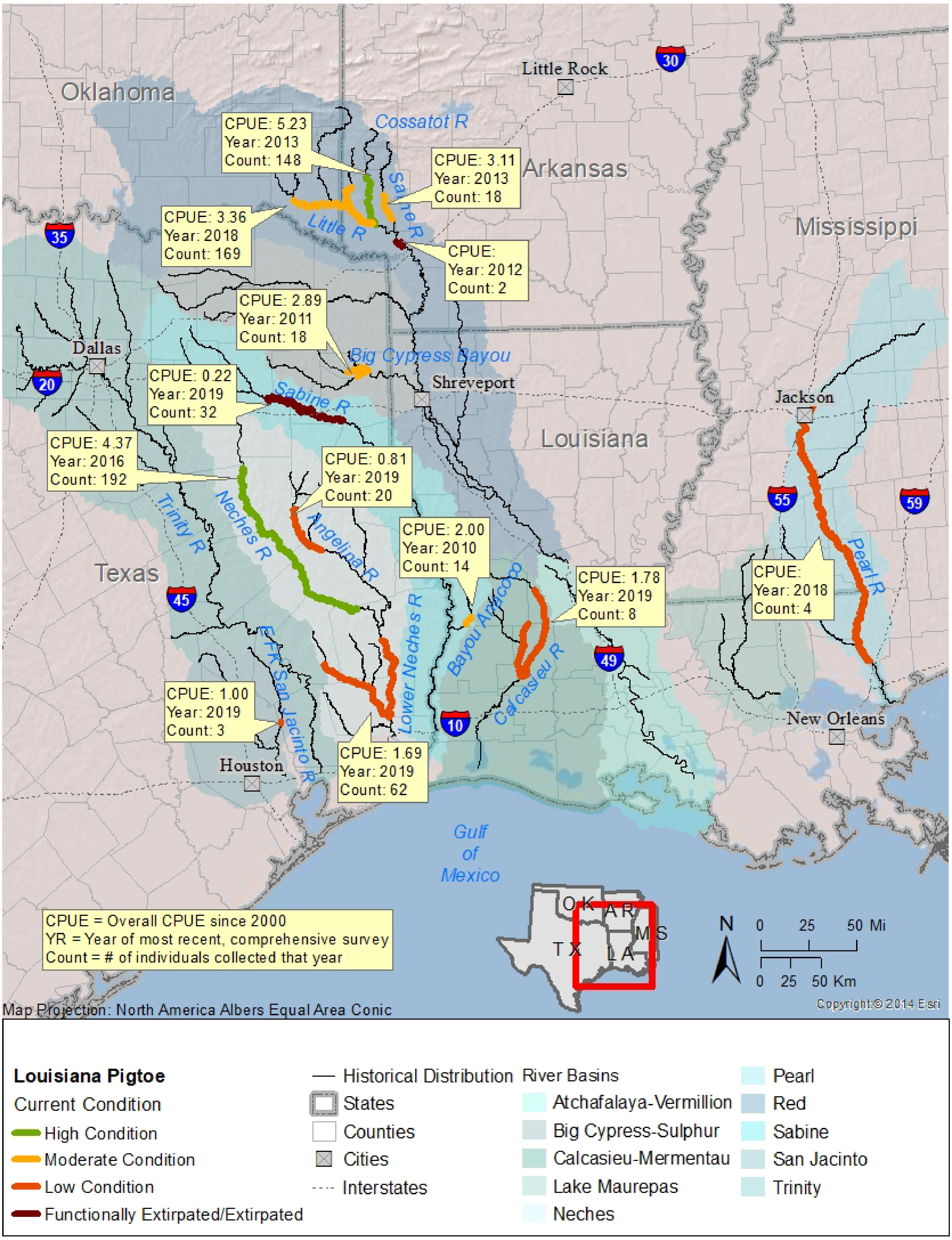
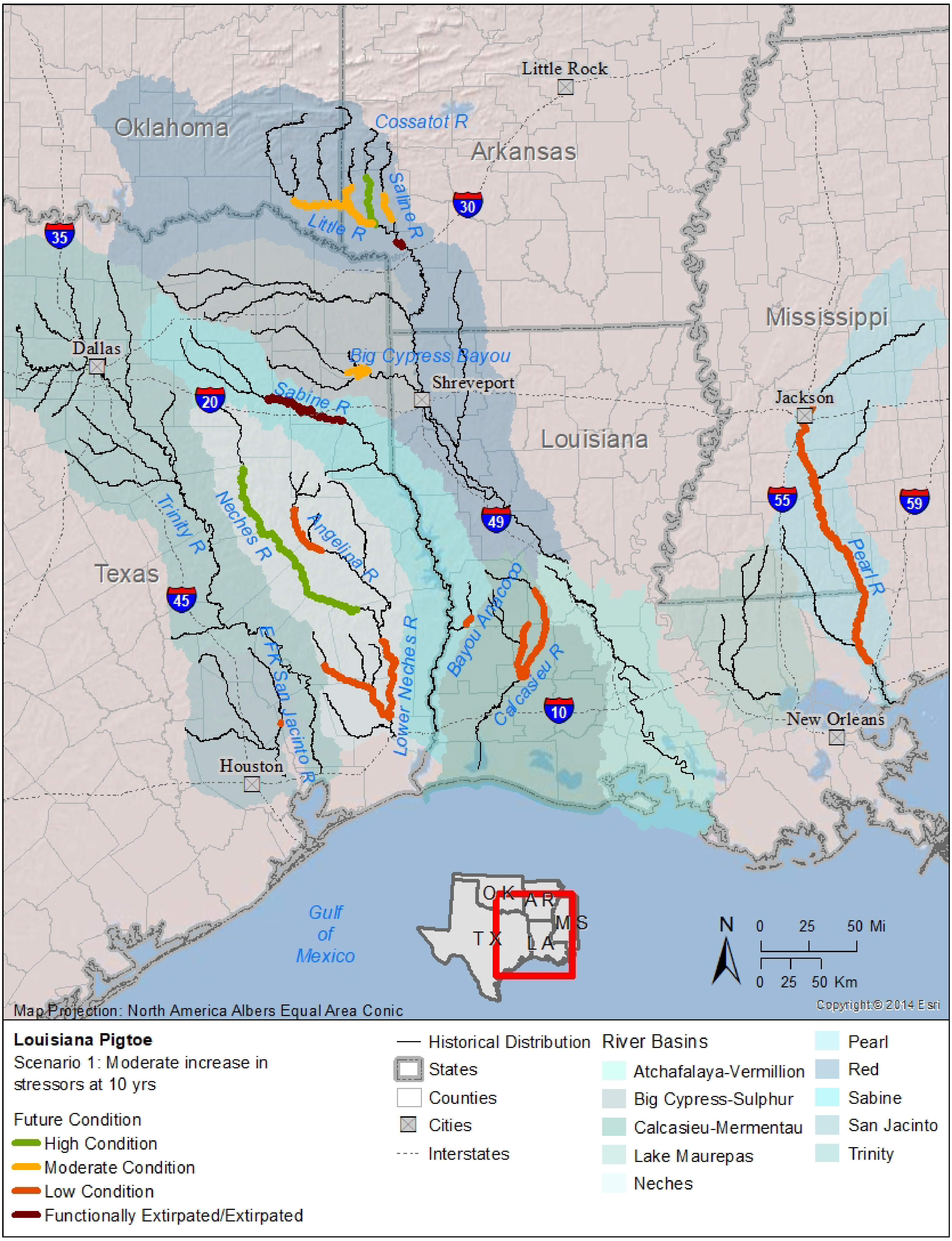
Figure C.2 Large-sized Current and Future Population Condition Maps for Louisiana Pigtoe





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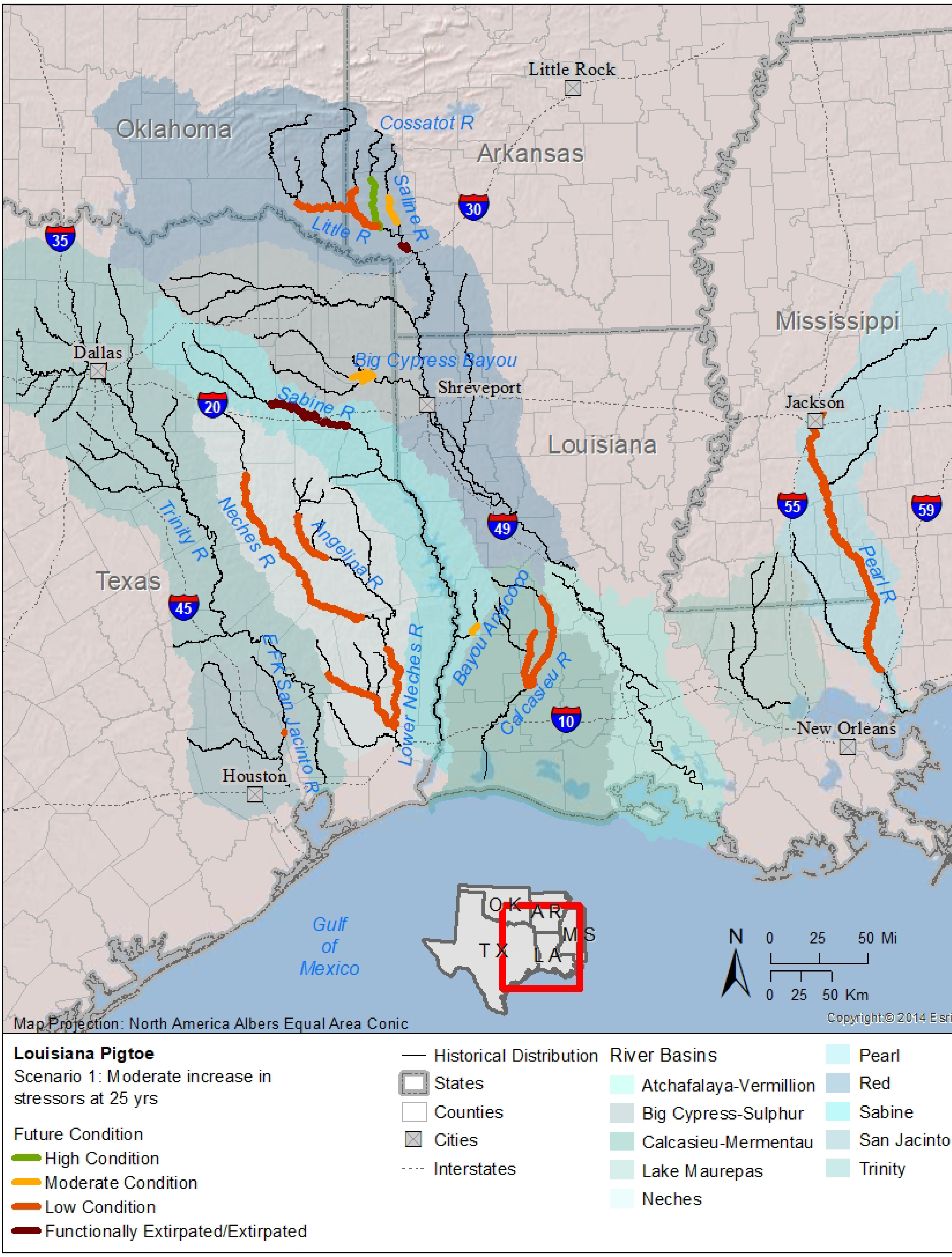

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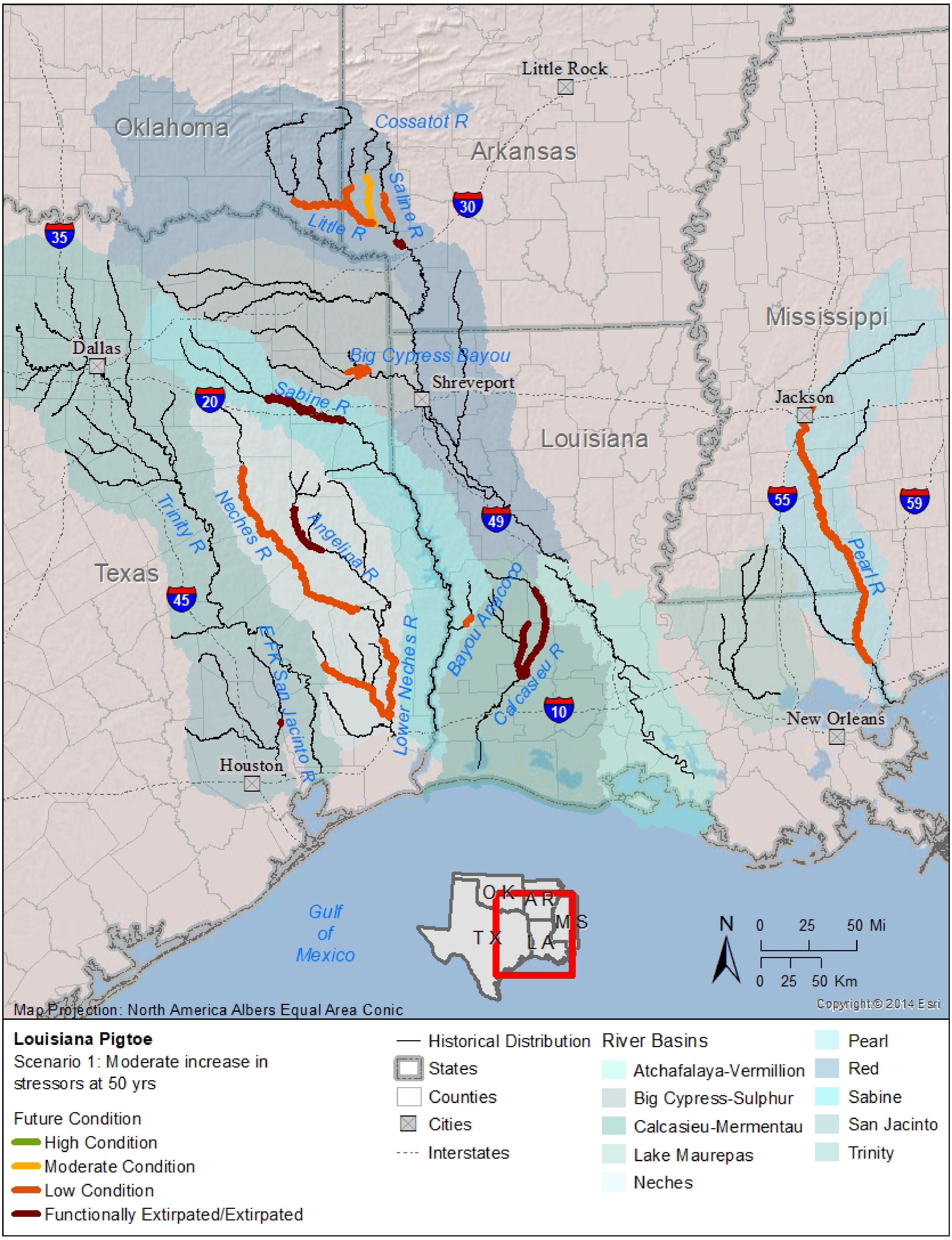

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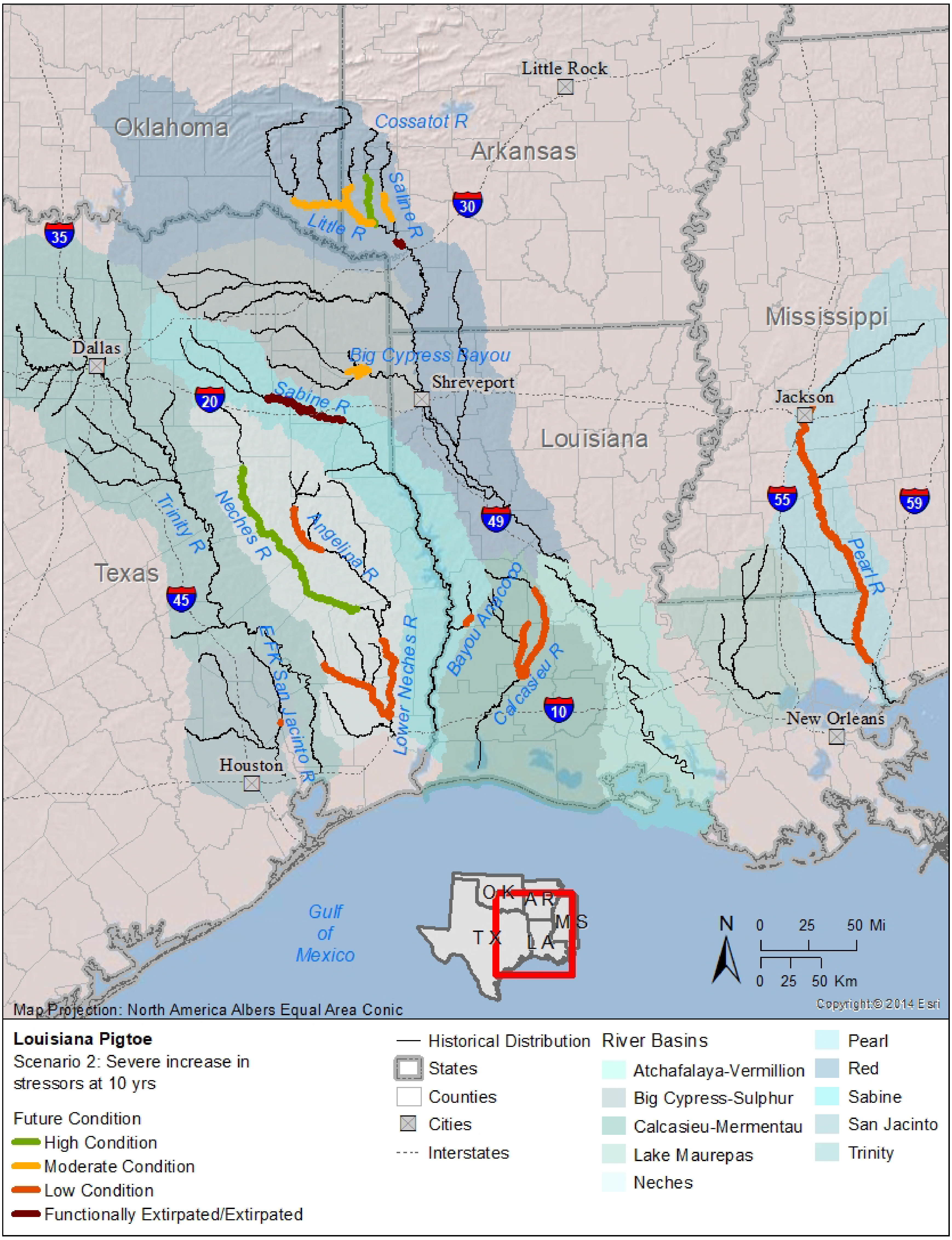

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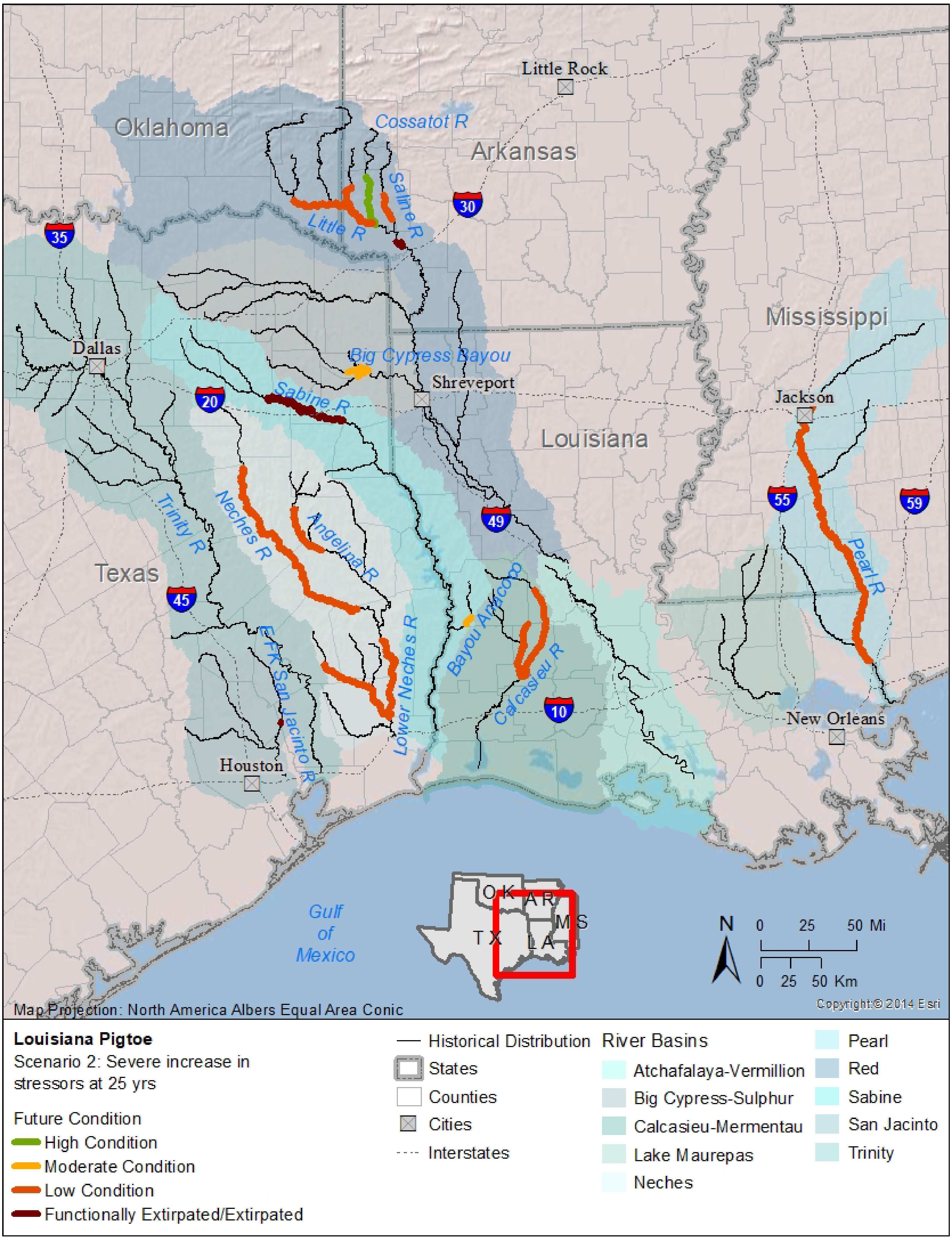

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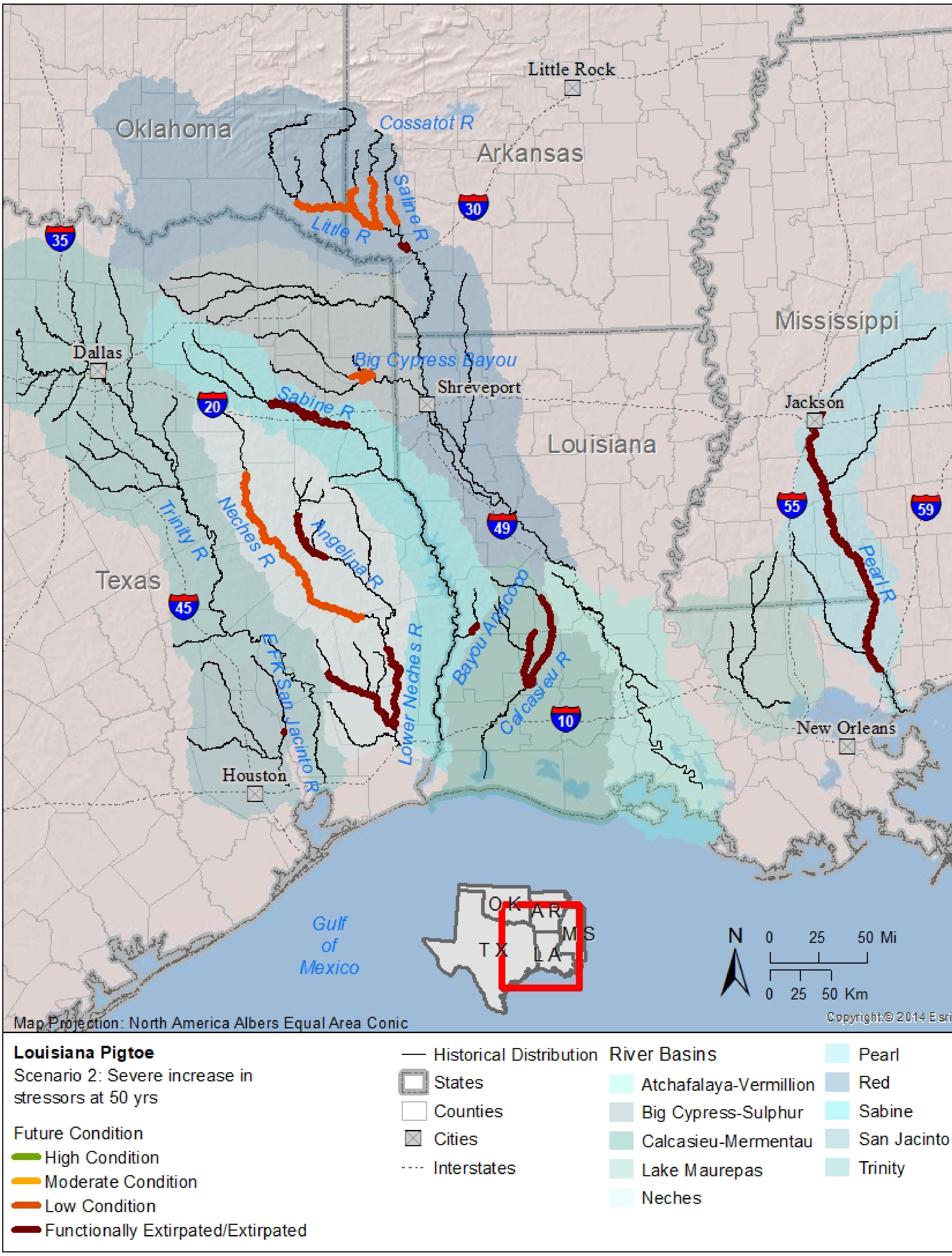
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Scenario 2: Severe increase in stressors at 50 yrs
Future Condition
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Functionally Extirpated/Extirpated
Historical Distribution
States
Counties
Cities
Interstates
River Basins
Atchafalaya-Vermillion
Big Cypress-Sulphur
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Appendix D. Descriptions of Select Indices of Hydrologic Alteration Evaluated as Part of Study: "Assessment and Review of Hydrological Relationships for Mussels in East Texas"

Appendix D: Descriptions for Select Indices of Hydrologic Alteration Evaluated (red boxes).

Group 2: Magnitude and duration of annual extreme flow conditions		
1-day minimum	Annual minimum 1-day mean discharge	ft^3/s
3-day minimum	Annual minimum 3-day mean discharge	ft^3/s
7-day minimum	Annual minimum 7-day mean discharge	ft^3/s
30-day minimum	Annual minimum 30-day mean discharge	ft^3/s
90-day minimum	Annual minimum 90-day mean discharge	ft^3/s
1-day maximum	Annual maximum 1-day mean discharge	ft^3/s
3-day maximum	Annual maximum 3-day mean discharge	ft^3/s
7-day maximum	Annual maximum 7-day mean discharge	ft^3/s
30-day maximum	Annual maximum 30-day mean discharge	ft^3/s
90-day maximum	Annual maximum 90-day mean discharge	ft^3/s
Number of zero days	Number of days having a discharge of zero for each year	Count
Base flow index	Minimum 7-day mean divided by mean annual flow for each year	ft^3/s
Group 3: Timing of annual minimum and maximum flow conditions		
Date of minimum	Julian date of each annual 1-day maximum	Julian date
Date of maximum	Julian date of each annual 1-day minimum	Julian date
Group 4: Frequency and duration of low- and high-flow pulses		
Low pulse count	Number of low-flow pulses within each year	Count
Low pulse duration	Duration of low-flow pulses within each year	Days
High pulse count	Number of high-flow pulses within each year	Count
High pulse duration	Duration of high-flow pulses within each year	Days
Parameter Group 5: Rate and frequency of change in flow		
Rise rate	Median of all positive differences between consecutive daily means	ft^3/s
Fall rate	Median of all negative differences between consecutive daily means	ft^3/s
Number of reversals	Median number of times in which flow switched from a rising period to a falling period or from a falling period to a rising period	Count

Monarch (*Danaus plexippus*)
Species Status Assessment Report, version 2.1
September 2020

Photo: Kelly Nail

Prepared by:

U.S. Fish and Wildlife Service

Acknowledgments

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Suggested Reference

U.S. Fish and Wildlife Service. 2020. Monarch (*Danaus plexippus*) Species Status Assessment Report. V2.1 96 pp + appendices.

*The changes from version 2.0 (July 2020) to 2.1 (Sept 2020) are minor corrections and do not change the results of the SSA analyses.

Executive Summary

The monarch, *Danaus plexippus*, is a species of butterfly globally distributed throughout 90 countries, islands, and island groups. These butterflies are well known for their phenomenal long-distance migration in the North American populations. Descendants of these migratory monarch populations expanded from North America to other areas of the world where milkweed (their larval host plant) was already present or introduced. With the year-round presence of milkweed and suitable temperatures, many of these global monarch populations no longer migrate.

Two North American populations, the migratory populations located east and west of the Rocky Mountains, have been monitored at their respective overwintering sites in Mexico and California since the mid-1990s. While these populations fluctuate year-to-year with environmental conditions, these census data indicate long-term declines in the population abundance at the overwintering sites in both populations (Figure E1). These declining trends led to the petition of the U.S. Fish and Wildlife Service to list the monarch butterfly for protection under the Endangered Species Act of 1973, as amended.

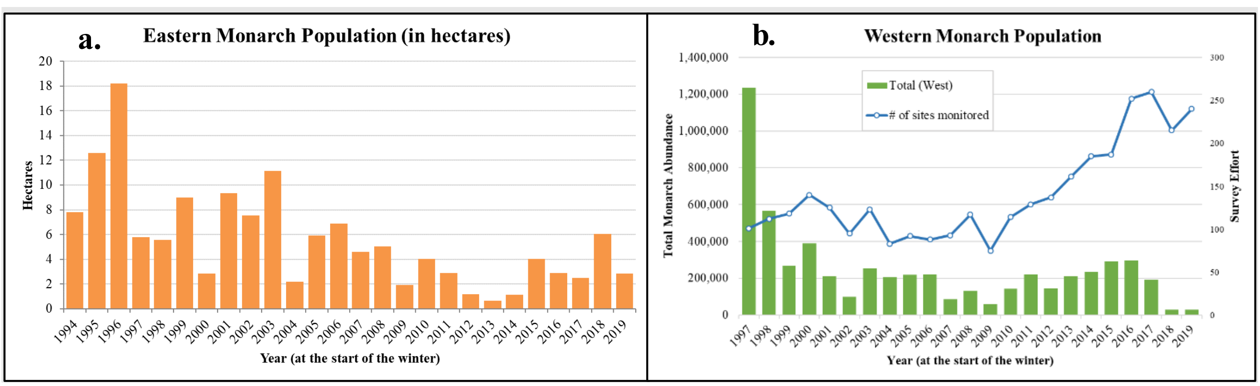

Figure E1. Eastern (a) and western (b) North American monarch population sizes, as measured at overwintering sites in terms of hectares (eastern) and total number (western). The western population count also has a blue line indicating survey effort (number of sites monitored). Horizontal black bars with labels indicate the decadal average population counts.

Using the best available scientific information about monarchs in North America and worldwide, we conducted a species status assessment (SSA). This report summarizes the results of our SSA. We delineated the historical number and distribution of monarch populations, assessed the status and health of the currently extant populations, identified the key drivers of their health, forecasted the likely future change in these drivers and monarch population responses to these changes, and evaluated the consequences of the population responses to monarch viability. Specifically, we evaluated the ability of the monarch to withstand environmental stochasticity (resiliency), catastrophes (redundancy), and novel changes in its biological and physical environment (representation).

We delineated 31 historical populations; of these, 27 are extant and 4 have unknown status. Outside of the 2 migratory North American populations, the health of the remaining 29 populations is undeterminable due to limited information available on population trends and stressors. However, at least 15 of these populations are at risk of extinction due to climate change related sea level rise or unsuitably high temperatures. The results for the two migratory North American populations show that both are facing declining numbers and overall health.

The primary drivers affecting the health of the two North American migratory populations are primarily: loss and degradation of habitat (from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence and incompatible management of overwintering sites in California, urban development, and drought), continued exposure to insecticides, and effects of climate change. Relative to the recent past, both the eastern and western North American populations have lower abundances and declining population growth rates. Using the best available science, we estimated the probability of the population abundance reaching the point at which extinction is inevitable (pE) for each population given their current abundance and growth rate, as well as under projected future conditions. The pE for the western population is high (60% to 68% chance within 10 years, reaching 99% by year 60) under current conditions and increases under projected future conditions. For the eastern population, the pE in 60 years under current conditions ranges from 48% to 69%, and under the projected future conditions, it ranges from 56% to 74%. The range in the estimates represents the best and worst plausible future state conditions of the primary drivers.

Additionally, at the current and projected low population numbers, both the eastern and western populations are more vulnerable to catastrophic events (e.g., extreme storms at the overwintering habitat) than in the past. These risks, however, are not captured in the pE estimates. Similarly, we found that under different climate change scenarios, the number of days and the area in which monarchs will be exposed to unsuitably high temperatures will increase markedly. We were unable to incorporate the effects of high daily temperatures into the extinction analyses, and thus, these risks as well are not fully captured in the pE estimates.

The extinction of either the western or eastern North American migratory population would increase the risk of losing the North American migratory phenomenon, as its persistence would depend solely upon the continued survival of a single population. Moreover, loss of either population would impair the species' ability to adapt into the future. The North American populations are unique in their long-distance migratory ability, and they represent unique sources of genetic and ecological diversity. Further, these two populations represent the historical and current core of the species and the ancestral lineage of the species. The eastern North American population is by far the largest of all populations (both in number of individuals and range size), and the western North American population encompasses as much as 30% of the geographic range occupied by monarch butterflies in North America. Accordingly, loss of these two ACUs would reduce monarch diversity, rendering the species less able to adapt to novel changes in its environment now and in the future and thereby increasing the extinction risk of the monarch. The chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (under future conditions) and drops under 10% within 30 years. Based on this information and other analyses of influences included in this SSA, monarch viability is declining and is projected to continue declining over the next 60 years.

Table of Contents

List of Acronyms

ACU	Adaptive capacity unit
Act	Endangered Species Act of 1973, as amended
CCAA	Candidate Conservation Agreement with Assurances
CDC	Center for Disease Control
CDL	Cropland Data Layer
CMIP5	Coupled Model Intercomparison Project Phase 5
CRP	Conservation Reserve Program
EROS	Earth Resources Observation and Science
FAO	The Food and Agriculture Organization of the United Nations
FSA	Farm Service Agency
IPCC	Intergovernmental Panel on Climate Change
MACA	Multivariate Adaptive Constructed Analogs
MAFWA	Midwest Association of Fish and Wildlife Agencies
MCD	Monarch Conservation Database
MP3	Managed Pollinator Protection Plan
NLCD	National Land Cover Database
NOAA	National Oceanic and Atmospheric Administration
PDSI	Palmer Drought Severity Index
RCP	Representative Concentration Pathways
SSA	Species Status Assessment
USDA	U.S. Department of Agriculture
USDOT	U.S. Department of Transportation
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
VRT	Variable Rate Technology
WAFWA	Western Association of Fish and Wildlife Agencies

Chapter 1: Introduction & Analytical Framework

The Center for Biological Diversity, Center for Food Safety, Xerces Society, and Dr. Lincoln Brower petitioned the U.S. Fish and Wildlife Service (USFWS) to list the monarch (*Danaus plexippus plexippus*) as a threatened species under the Endangered Species Act of 1973, as amended (Act) on August 26, 2014 (Center for Biological Diversity et al. 2014). In December 2014, USFWS found the petition presented substantial scientific or commercial information that indicated listing the monarch may be warranted (79 FR 78775) and initiated a rangewide status review.

This report summarizes the results of a species status assessment (SSA) conducted for the monarch butterfly, and it is intended to provide the biological support for the decision on whether the monarch warrants listing under the Act. Importantly, the SSA report is not a decisional document; rather it provides a review of available information strictly related to the species' biological status. The USFWS will make a listing determination after reviewing this document and all relevant laws, regulations, and policies, and will announce the results of the determination in the *Federal Register*, with appropriate opportunity for public input. This report has undergone peer and state review and incorporates the best available scientific data.

This chapter describes the analytical framework and the conservation principles used to assess monarch viability over time. Chapter 2 summarizes the ecological requirements for survival and reproduction at the individual, population, and species levels. Chapter 3 details the methods underlying our analyses. Chapters 4 and 5 summarize the historical and current conditions of monarch, respectively, and identifies the key factors (referred to as influences) that contributed to the species' current condition. Chapter 6 describes the projected changes in these key influences. Chapter 7 summarizes the projected future condition of the monarch given the plausible projections of the key influences. Lastly, Chapter 8 synthesizes the above analyses and describes how the consequent change in the number, health, and distribution of monarch populations influence monarch viability over time. In this final chapter, we also describe sources of uncertainty and the implications of this uncertainty. Additionally, we include four appendices providing further information on taxonomy, methodology, results, and other drivers considered.

Analytical Framework

Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308-311). Meaning, to sustain populations over time, a species must have a sufficient number of populations distributed throughout its geographic range to withstand:

(1) environmental stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

Viability is a continuous measure of the likelihood of sustaining populations over time and can be defined in relative terms, such as "low" or "high" viability. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is generally better able to adapt to future changes and to tolerate catastrophes, environmental stochasticity, and stressors, and thus, typically has high viability.

Resiliency is the ability of the species to withstand and sustain populations through environmental stochasticity (normal, year-to-year variations in environmental conditions, such as temperature or rainfall), periodic disturbances (fire, floods, storms, etc.), and anthropogenic stressors (factors that cause a negative effect to a species or its habitat) (Redford et al. 2011, p. 40). Simply stated, resiliency refers to a species' ability to sustain populations through favorable and unfavorable environmental conditions and anthropogenic impacts.

Resiliency is multi-faceted. First, it requires having healthy populations demographically (robust survival, reproductive, and growth rates), genetically (large effective population size, high heterozygosity, and gene flow between populations), and physically (good body condition). Second, resiliency also requires having healthy populations distributed across heterogeneous environmental conditions (referred to as spatial heterogeneity; this includes factors such as temperature, precipitation, elevation, and aspect). Because environmental stochasticity can operate at regional scales (Hanski and Gilpin 1997, p. 372), populations tend to fluctuate in synchrony over broad geographical areas (Kindvall 1996, pp. 207, 212; Oliver et al. 2010, pp. 480-482). Spatial heterogeneity induces asynchronous fluctuations among populations, thereby guarding against concurrent population declines. Lastly, resiliency often requires connectivity among populations to maintain robust population-level heterozygosity via gene flow among populations and to foster demographic rescue following population decline or extinction due to stochastic events.

Redundancy is the ability of a species to withstand catastrophes; defined here as highly consequential events (cause population extinction) for which adaptation is unlikely (Mangal and Tier 1993, p. 1083). For all species, a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000, pp. 307, 309-310; Groves et al. 2002, p. 506). Reducing the risk of extinction due to a single or series of catastrophic events requires having multiple populations widely distributed across the species' range, with connectivity among groups of locally adapted populations to facilitate demographic rescue following population decline or extinction. This provides a margin of safety to reduce the risk of losing substantial portions of genetic diversity or the entire species to a single or series of catastrophic events.

Representation is the ability of a species to adapt to both near-term and long-term novel changes in the conditions of its environment, both physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (novel pathogens, competitors, predators, etc.). This ability, referred to as adaptive capacity, is essential for viability because species need to continually adapt to their continuously changing environment (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either 1) moving to new, suitable environments or 2) by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270).

Maintaining a species' ability to disperse and colonize new environments fosters adaptive capacity by allowing species to move from areas of unsuitable conditions to regions with more favorable conditions. It also fosters adaptive capacity by increasing genetic diversity via gene flow, which is, as discussed below, important for evolutionary adaptation (Hendry et al. 2011, p. 173; Ofori et al. 2017, p. 1). Thus, maintaining natural levels of connectivity among populations is important for preserving a species' adaptive capacity (Nicotra et al. 2015, p. 1272).

Maintaining a species' ability to adapt to novel conditions also requires preserving the breadth of genetic variation. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either genetic change or plasticity (see Text Box 1.1). For adaptation to occur, whether through plasticity or evolutionary adaptation, there must be genetic variation upon which selection can act (Hendry et al. 2011, pp. 164-165; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 326). Without genetic variation, the species cannot adapt and is more prone to extinction (Spielman et al. 2004, p. 15263; also see Text Box 1.1).

Text Box. 1.1. ***Species Adaptation***. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either *genetic change* or *plasticity* (Chevin et al. 2010, p. 2-3; Hendry et al. 2011, p. 162; Nicotra et al. 2015, p.1270). *Genetic change,* referred to as evolutionary adaptation or potential, involves a change in phenotypes via an underlying genetic change (specifically, a change in allele frequency) in response to novel environmental cues (Nicotra et al. 2015, p. 1271; Ofori et al. 2017, p. 2). *Plasticity*, unlike evolutionary adaptation, involves a change in phenotypes (phenotypic plasticity) without undergoing changes in the genetic makeup (Nicotra et al. 2015, p. 1271-1272). Plasticity is an important mechanism for species to adapt both in immediate and future time frames. In the immediate time frame, plasticity directly acts to allow species to persist despite novel changes in the environment. In the longer time frame, plasticity contributes to a species' adaptive capacity by buying time for adaptive evolution to occur through genetic changes (referred to as genetic assimilation, see Ghalambor et al. 2007, p. 395; Nicotra et al. 2015, p. 1271). Not all genetic and plastic induced changes are adaptive; changes must lead to improved fitness to be adaptive (Nicotra et al. 2015, p. 1271-1272). Importantly, however, adaptive traits can vary over space and time; what is adaptive in one location may not be adaptive in another, and similarly, what is adaptive today may not be under future conditions and vice versa (Nicotra et al. 2015, p. 1271-1272). Thus, maintaining the full breadth of variation in both plastic traits and genetic diversity is important for preserving a species' adaptive capacity.

Genetic variation that is adaptive is difficult to identify for a species and represents a significant challenge even when there is genetic information available. To denote variation as 'adaptive' we need to identify which loci are under selection, which traits those loci control, how those traits relate to fitness, and what the species' evolutionary response to selection on those traits will be over time (Hendry et al. 2011, p. 162-163; Lankau et al. 2011, p. 316; Teplitsky et al. 2014, p. 190). Although new genomic techniques are making it easier to obtain this type of information (see Funk et al. 2019), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation. One of the easiest proxies to measure is variation in biological traits (also described as phenotypic variation). Phenotypic variation, which on its own can be a mechanism for adapting to novel changes, can be due to underlying adaptive genetic variation (Crandall et al. 2000, p. 291; Forsman 2014, p. 304; Nicotra et al. 2015, p. 3). A second proxy for adaptive genetic variation is neutral genetic variation, which is usually the type of genetic data first reported in species-specific genetic studies (see Text Box 1.2). A third, and more distant, proxy for adaptive genetic variation is disjunct or peripheral populations (Ruckelhaus et al. 2002, p. 322). These populations can be exposed to the extremes in climate tolerances for the species and thus harbor unique and potentially adaptive traits. Similarly, populations that occur across steep environmental gradients can be indicators of underlying adaptive genetic diversity because local adaptation is driven by environmental conditions, which are continually changing at different rates and scales (Sgro et al. 2011, p. 330, 333).

Text Box. 1.2. ***Genetic diversity***. Genetic variation can be partitioned into two types: adaptive and neutral genetic diversity. Both types are important for preserving the adaptive capacity of a species (Moritz 2002, p. 243), but in different ways. Genetic variation under selection underlies traits that are locally adaptive and that determine fitness (Holderegger et al. 2006, pp. 801, 803; Lankau et al. 2011, p. 316); thus, it is the variation that underpins adaptive evolution (Sgro et al. 2011, p. 328). This type of genetic variation is referred to as adaptive genetic diversity and determines the capacity for populations to exhibit an adaptive evolutionary response to changing environmental conditions. Conversely, neutral genetic variation refers to regions of the genome that have no known direct effect on fitness (i.e., selectively neutral) and change over time due to non-deterministic processes like mutation and genetic drift (Sgro et al. 2011, p. 328). Although, by definition, neutral genetic variation is not under selection, it contributes to the adaptive capacity of a species in a couple of ways. First, neutral genetic variation that is statistically neutral in one environment may be under selection--and thus adaptive--in a different environment (Nicotra et al. 2015, p. 1271-1272). Second, neutral markers can allow us to infer evolutionary lineages, which is important because distinct evolutionary lineages may harbor locally adaptive traits (Hendry et al. 2011, p. 167), and hence, serve as an indicator of underlying adaptive genetic variation. Thus, maintaining the full breadth of neutral and adaptive genetic diversity is important for preserving a species' adaptive capacity.

Lastly, preserving a species' adaptive capacity requires maintaining the natural levels of the processes that allow for evolution to occur; namely, natural selection and gene flow (Crandall et al. 2000, p. 290-291; Sgro et al. 2011, p. 327; Zackay 2007, p. 1). Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population via differential survival or reproduction (Hendry et al. 2011, p. 169). To preserve natural selection as a functional evolutionary force, it is necessary to maintain populations across an array of environments (Hoffmann and Sgro 2011, p. 484; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 332; Shaffer and Stein 2000, p. 308). Gene flow serves as an evolutionary process by introducing new alleles (variant forms of genes) into a population, thereby, increasing the gene pool size (genetic diversity). Maintaining the natural network of genetic connections between populations will foster and preserve the effectiveness of gene flow as an evolutionary process (Crandall et al. 2000, p. 293). Along with maintaining large effective population sizes, preserving genetic connections among populations also helps minimize the loss of genetic variation due to genetic drift (Crandall et al. 2000, p. 293). Large population numbers also important to adaptive capacity because the level of diversity is influenced by population size and the rate of evolutionary adaptation is faster in populations with high diversity (Ofori et al. 2017, p.2).

Chapter 2: Species Ecology

This chapter describes the ecological requirements for survival and reproduction at the individual, population, and species levels (the first step of our analytical framework).

Species Description

The monarch, *Danaus plexippus* (Linneaus, 1758), is a species of butterfly in the order Lepidoptera (family Nymphalidae) that occurs in North, Central, and South America; Australia; New Zealand; islands of the Pacific and Caribbean, and elsewhere (Malcolm and Zalucki 1993, p. 3-5; Fig. 4.1). Adult monarch butterflies are large and conspicuous, with bright orange wings surrounded by a black border and covered with black veins. The black border has a double row of white spots, present on the upper side and lower side of forewings and hindwings (Bouseman and Sternburg 2001, p. 222). Adult monarchs are sexually dimorphic, with males having narrower wing venation and scent patches (CEC 2008, p.11; Figure 2.0). The bright coloring of a monarch is aposematic, as it serves as a warning to predators that eating them can be toxic.

Figure 2.0. Male monarch on milkweed. Note the arrow pointing to the black dots known as androconial scent patches on the hind wings. These are not present on female monarchs. Photo by Tim Koerner, USFWS.

Taxonomy

In 2014, a petition was received to list the subspecies of the monarch butterfly (*Danaus plexippus plexippus*) under the Endangered Species Act (Center for Biological Diversity et al. 2014). The petition also requested a determination of whether any new North American subspecies of *Danaus plexippus* should be listed. After careful examination of the literature and consultation with experts, there is no clearly agreed upon definition of potential subspecies of *Danaus plexippus* or where the geographic borders between these subspecies might exist. Given these findings, we examined the entire range of *Danaus plexippus* for this assessment. For more information on taxonomy, see Appendix 1.

Individual-Level Ecology and Requirements

Below we describe the ecological needs for monarch individuals to survive and reproduce; these needs are summarized in Table 2.1. During the breeding season, monarchs lay their eggs on their obligate milkweed host plant (primarily *Asclepias* spp.), and larvae emerge after two to five days (Zalucki 1982, p. 242; CEC 2008, p. 12). Larvae develop through five larval instars (intervals between molts) over a period of 9 to 18 days, feeding on milkweed and sequestering toxic cardenolides as a defense against predators (Parsons 1965, p. 299). The larva then pupate into chrysalis before eclosing 6 to 14 days later as an adult butterfly. There are multiple generations of monarchs produced during the breeding season, with most adult butterflies living approximately two to five weeks; overwintering adults enter into reproductive diapause (suspended reproduction) and live six to nine months (Cockrell et al. 1993, pp. 245-246; Herman and Tatar 2001, p. 2509; Figure 2.1).

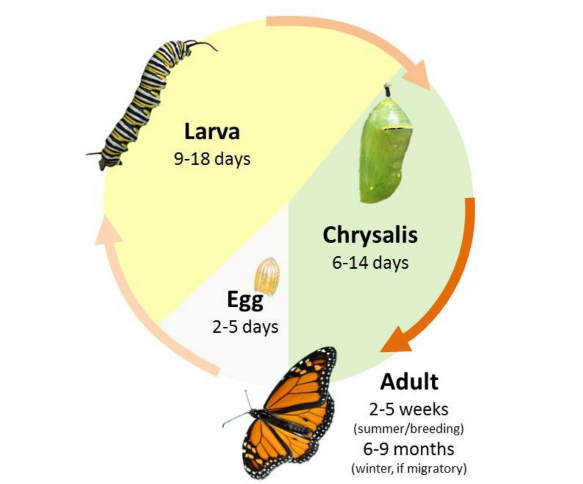

Figure 2.1. Monarch life cycle. Development times calculated from Zalucki (1982) based on temperatures ranging from 22°-32°C. Adult life span based on Herman and Tatar (2001).

The monarch life cycle varies by geographic location. In many regions where monarchs are present, monarchs breed year-round, repeatedly following the above-referenced life cycle throughout the year. Individual monarchs in temperate climates, such as eastern and western North America, undergo long-distance migration, where the migratory generation of adults is in reproductive diapause and lives for an extended period of time (Herman and Tatar 2001, p. 2509). In the fall, in both eastern and western North America, monarchs begin migrating to their respective overwintering sites. This migration can take monarchs distances of over 3,000 km (Urquhart and Urquhart 1978, p. 1760) and last for over two months (Brower 1996, p. 93). Migratory individuals in eastern North America predominantly fly south or southwest to mountainous overwintering grounds in central Mexico, and migratory individuals in western North America generally fly shorter distances south and west to overwintering groves along the California coast into northern Baja California (Solensky 2004, p. 79; see Figure 2.2). Data from monarchs tagged in the southwestern states in the fall suggest that those in Nevada migrate to California, those in New Mexico migrate to Mexico, and those in Arizona migrate to either

Mexico or California (Southwest Monarch Study Inc. 2018). In early spring (February-March), surviving monarchs break diapause and mate at the overwintering sites before dispersing (Leong et al. 1995, p. 46, van Hook 1996, pp. 16-17). The same individuals that undertook the initial southward migration begin flying back through the breeding grounds and their offspring start the cycle of generational migration over again (Malcolm et al. 1993, p. 262).

In eastern North America, monarchs travel north in the spring, from Mexico to Canada, over two to three successive generations, breeding along the way (Flockhart et al. 2013, p. 4-5; Figure 2.2). Individual monarchs disperse as far north as they can physiologically tolerate based on climatic conditions and available vegetation; the most specific predictors of the northern distribution of individual monarchs are monthly mean temperature and precipitation (Flockhart et al. 2013, p. 4; Flockhart et al. 2017, p. 2568). The number of generations of monarchs produced in a given year can vary between three and five and is dependent upon environmental conditions (Brower 1996, p. 100). While a majority of the eastern monarchs shift to the more northern reaches of their range, western monarchs continue to occupy and breed in warmer climates throughout the summer, while also expanding to include the farther reaches of their range. In the spring in western North America, monarchs migrate north and east over multiple generations from coastal California toward the Rockies and to the Pacific Northwest (Urquhart and Urquhart 1977, p. 1585; Nagano et al. 1993, p. 157; Figure 2.2). In the southwestern states, migrating monarchs tend to occur more frequently near water sources such as rivers, creeks, roadside ditches, and irrigated gardens (Morris et al. 2015, p. 100). While the overwintering areas shown in Figure 2.2 represent the sites where most monarchs in North America overwinter in reproductive diapause, there are other sites and overwintering strategies (see *Uncertainties* section in Chapter 8).

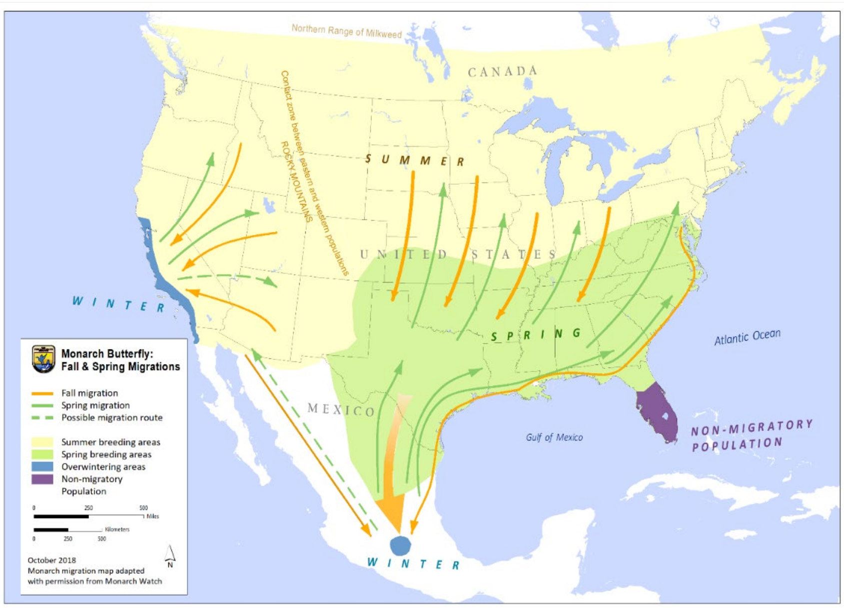

Figure 2.2. North American monarch migration map.

Adult monarch butterflies during breeding and migration require a diversity of blooming nectar resources, which they feed on throughout their migration routes and breeding grounds (spring through fall). Monarchs also need milkweed (for both oviposition and larval feeding) embedded within this diverse nectaring habitat. The correct phenology, or timing, of both monarchs and nectar plants and milkweed is important for monarch survival. The position of these resources on the landscape is important as well (see *Population-Level Ecology* section in this chapter). In western North America, nectar and milkweed resources are often associated with riparian corridors, and milkweed may function as the principal nectar source for monarchs in more arid regions (Dingle et al. 2005, p. 494; Pelton et al. 2018, p. 18; Waterbury and Potter 2018, p. 38; Dilts et al. 2018, p. 8). Individuals need nectar and milkweed resources year-round in non-migratory populations. Additionally, many monarchs use a variety of roosting trees along the fall migration route (Table 2.1).

Migratory individuals of eastern and western North America require a very specific microclimate at overwintering sites. The eastern population of monarchs overwinter in Mexico, where this microclimate is provided by forests primarily composed of oyamel fir trees (*Abies religiosa*), on which the monarchs form dense clusters (Williams and Brower 2015, pp. 109-110). The sites used for overwintering occur in mountainous areas west of Mexico City located between elevations of 2,900 and 3,300 m (Slayback and Brower 2007, p. 147). The temperature must remain cool enough to prevent excessive lipid depletion (Alonso-Mejía et al. 1997, p. 935), while at the same time staying warm enough to prevent freezing (Anderson and Brower 1996, pp. 111-113). Exposure to these cooler temperatures also helps orient the monarchs northward in the spring (Guerra and Reppert 2013, pp. 421-422). The oyamel fir forest provides essential protection from the elements, including rain, snow, wind, hail, and excessive solar radiation (Williams and Brower 2015, p. 109). Many sites also provide a source of hydration via nectar plants or a water source (Brower et al. 1977, pp. 237-238). Most of the observed overwintering sites are located within the Monarch Butterfly Biosphere Reserve, which covers over 56,000 ha (Vidal and Rendón-Salinas 2014, p. 169; Ramírez et al. 2015, p. 158).

Migratory monarchs in the western population primarily overwinter in groves along the coast of California and Baja California (Jepsen and Black 2015, p. 149). The location and structure of these sites provide the specific microclimate (although different from the Mexico overwintering microclimate) needed for survival in the western overwintering areas. There are approximately 400 groves that have been occupied, but only a portion of these sites is occupied in any given year. These sites, typically close to the coast, span approximately 1.225 km of coastline (COSEWIC 2010, p. 10). These groves are populated by a variety of tree species, including blue gum eucalyptus (*Eucalyptus globulus*), Monterey pine (*Pinus radiata*), and Monterey cypress (*Hesperocyparis macrocarpa*) (Griffiths and Villablanca 2015, pp. 41, 46-47), all of which act as roost trees. These groves provide indirect sunlight for the overwintering monarchs, sources of moisture for hydration, defense against freezing temperatures, and protection against strong winds (Tuskes and Brower 1978, p.149; Leong 1990, pp. 908-910, Leong 1999, p. 213). The close proximity to the coast (average distance of 2.37 km ± 0.39 SE) also provides a mild winter climate (Leong et al. 2004, p. 180).

Table 2.1. Individual-level requisites for monarch survival and reproduction.

Life Stage	Requirements	Description
Eggs, Larvae, and Adults – breeding	Milkweed resources	Healthy and abundant milkweed is needed for oviposition and larval consumption.
Adult – breeding and migration	Nectar resources	Sufficient quality and quantity of nectar from flowers is needed for adult feeding throughout the breeding and migration seasons.
Adult – overwintering	Suitable habitat for overwintering	Habitat that provides a specific roosting microclimate for overwintering: protection from the elements (e.g., rain, wind, hail, excessive radiation) and moderate temperatures that are warm enough to prevent freezing yet cool enough to prevent lipid depletion. Nectar and clean water sources located near roosting sites.
Adult – migration	Connectivity & Phenology	Nectar and milkweed resources along the migration route when butterflies are present; the size and spatial arrangement of habitat patches are generally thought to be important aspects, but currently unknown. Roosting sites may also be important for monarchs along their fall migration route.

Population-Level Ecology

The ecological requirements of a healthy monarch population are summarized in Table 2.2. To be self-sustaining, a population must be demographically, genetically, and physically healthy (see Redford et al. 2011, entire). Demographically healthy means having robust survival, reproductive, and growth rates. Genetically healthy populations have large effective population sizes (N_e), high heterozygosity, and gene flow between populations. Physically healthy means individuals have good body condition. Monarchs, like many insects, are sensitive to environmental conditions (temperature and precipitation) and can experience large swings in population numbers year-to-year in response to these conditions (Rendón-Salinas et al. 2015, p. 3; Schultz et al. 2017, pp. 3-4). During favorable conditions, monarch survival and reproductive rates are high and population numbers increase; conversely, when environmental conditions are unfavorable, survival and reproductive rates are low and population numbers can plummet. Thus, to successfully recruit over generations and years, they must be capable of withstanding large swings in population sizes (N). Specifically, they need a robust population growth rate (lambda, or λ). Given that environmental fluctuations vary spatially, robust growth rates likely vary across populations.

To support a strong growth rate, monarch populations require large population sizes and sufficient quantity and quality of habitat to accommodate all life stages. Large population sizes also help maintain genetic health (via large N_e) and facilitate thermoregulation during the winter, which is important for good physical health. It may also be important for mate finding and aposematism (S. Malcolm, pers. comm. 2018). The quality of habitat needed to support healthy demographic rates and physical health is described under *Individual-Level Ecology and Requirements*. The quantity of habitat likely varies among populations, and exact requirements may vary (e.g., the type of trees needed for overwintering).

Migratory monarch populations can have individuals that can fly distances of over 3,000 km (Urquhart and Urquhart 1978, p. 1760; see *Individual-Level Ecology and Requirements* earlier in this chapter). During migration to overwintering sites, most monarchs are in reproductive diapause, but continue to need blooming nectar plants throughout the migratory habitat to provide sugar that is eventually stored as lipid reserves (Brower et al. 2015, p. 117). On their return, monarchs are laying eggs, and thus need both nectar sources and milkweed. This habitat needs to be distributed throughout the landscape to ensure connectivity throughout their range and maximize lifetime fecundity (Zalucki and Lammers 2010, p. 84; Miller et al. 2012, p. 2). However, the specific optimal amount of habitat and its spatial distribution are unknown; more research is needed on optimal distances between habitat patches, as well as optimal patch sizes and milkweed density and characteristics of patches selected for female oviposition (Kasten et al. 2016, p. 1055; Stenoien et al. 2016, p. 8; Grant et al. 2018, p. 48; Waterbury and Potter 2018, p. 48).

Table 2.2. The population-level requisites for a healthy population.

Parameter	Requirements
Population growth rate, λ	The long-term λ must be sufficiently high to rebound from population lows. On average, λ must be >1; how much greater than 1 is dictated by the degree of environmental variability.
Population size, N	Sufficiently large *N* to withstand periodic population lows; the minimum *N* required is dictated by the degree of environmental variability and varies geographically across populations.
Habitat	Sufficient seasonally and geographically specific quantity and quality of milkweed, breeding season nectar, migration nectar, and overwintering resources to support large population sizes.
Connectivity	A matrix of seasonally specific habitat patches throughout the landscape to support breeding and migrating monarchs and allow migration throughout the population's range each year.

Species-Level Ecology

The ecological requisites at the species level include having a sufficient number and distribution of healthy populations to ensure it can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel biological and physical changes in its environment (representation). We describe the monarch's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

Monarch resiliency requires maintaining healthy populations across spatially heterogeneous conditions. Healthy monarch populations are better able to withstand and recover from environmental variability and stochastic perturbations (e.g., storms, dry years) than those populations that are less demographically, genetically, or physically healthy. The greater the number of healthy populations, the more likely it is that the monarch will withstand perturbations and natural variation, and hence, have greater resiliency. Additionally, given the monarch's sensitivity to environmental conditions (experiencing large swings in population numbers year-to-year; Rendón-Salinas et al. 2015, p. 3), monarchs occupying a diversity of environmental conditions and being widely distributed helps guard against populations being exposed to adverse conditions concurrently, and thus, fluctuating in synchrony. Asynchronous dynamics within and among populations minimizes the chances of concurrent losses, and thus, provides species' resiliency. Lastly, maintaining the natural patterns and levels of connectivity between populations also contributes to monarch resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

Monarch redundancy is best achieved by having multiple, widely distributed populations of monarchs relative to the spatial occurrence of catastrophic events. In addition to guarding against a single or series of catastrophic events that extirpate monarch populations, redundancy is important to protect against reducing the species' adaptive capacity. Having multiple monarch populations, occupying areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events.

Representation

The monarch's ability to withstand novel changes is influenced by its adaptive capacity, which is primarily a function of the species' ability to colonize new areas and its breadth of variation in biological traits and genetic diversity (both neutral and adaptive genetic variation). In addition, and as explained in Chapter 1, maintaining large populations across an array of environments as well as the natural networks of genetic connections among populations are important components of preserving a species' adaptive capacity. Below we describe monarch adaptive capacity by using the best available data to incorporate the multiple proxies for adaptive variation described in Chapter 1. These proxies include genetic, morphological, behavioral, and ecological data drawn from published literature and expert knowledge. Based on these data, we delineated eight geographical units, referred to as adaptive capacity units (ACUs), which are depicted in Figure 2.3 and described below.

Table 2.3. Species-level requisites for species' viability (i.e., ability to sustain populations over time).

3 Rs	Species-Level Requisites	Details
Resiliency	Healthy populations distributed across spatially heterogeneous conditions	Healthy populations distributed across a diversity of temperatures, precipitation levels, elevations, and aspects.
Redundancy	Healthy populations distributed across geographical areas with low risks to catastrophic events	Widely spread, healthy populations to ensure all populations are not exposed to a single or series of catastrophic events.
Representation	Having healthy populations distributed across the breadth of genetic and phenotypic diversity; maintaining evolutionary processes	Breadth of variation in biological traits and genetic diversity via persistent populations within the 8 ACUs. Also, functional evolutionary processes via ensuring populations occupy an array of environments, maintaining genetic connections, and ensuring large N_e.

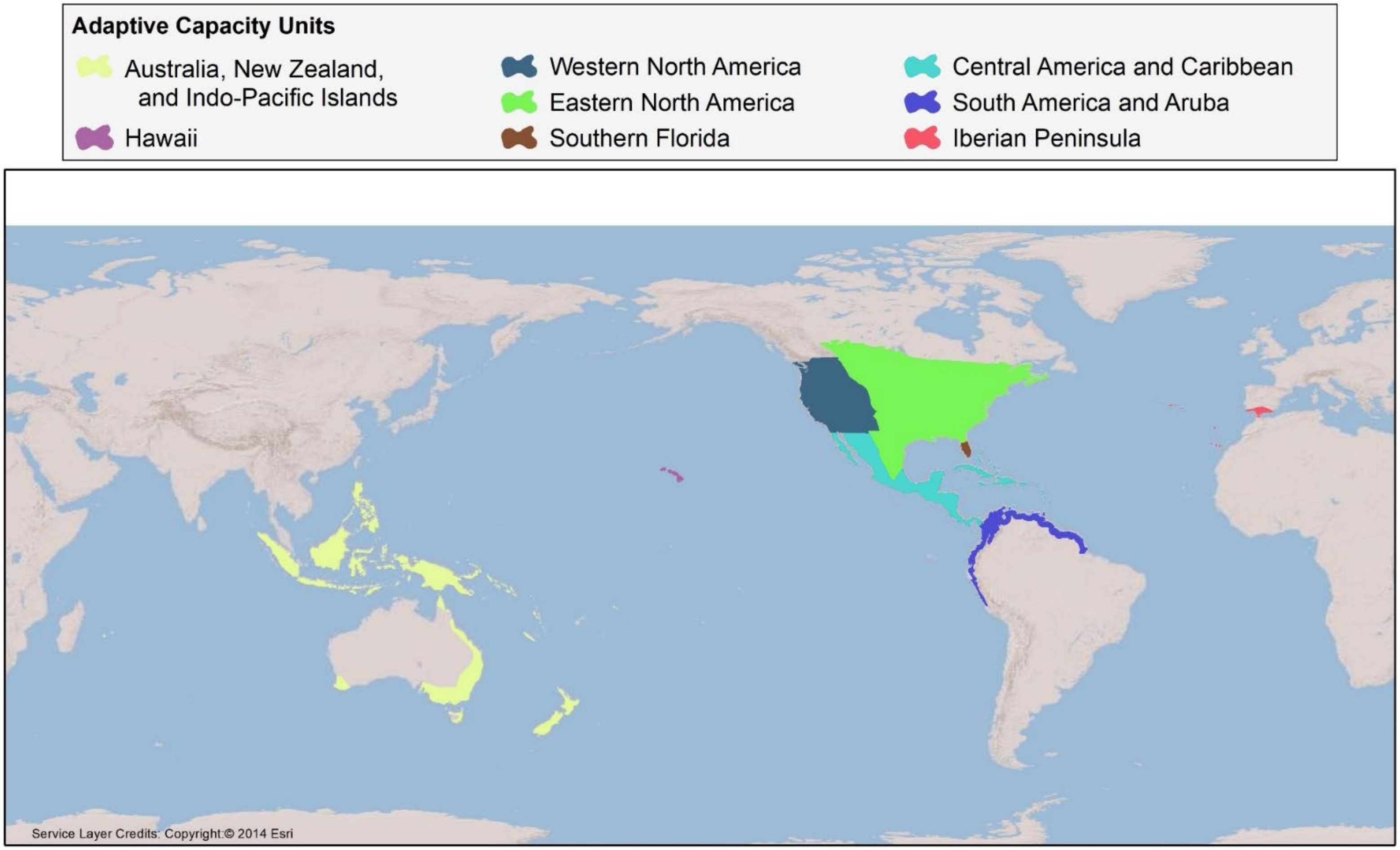

Figure 2.3. Worldwide range of monarchs organized into eight ACUs.

1. Eastern North America:

   Eastern North American monarchs are identified as an ACU because they contribute unique phenotypic variation in long-distance migratory behavior, wing morphology, and disease/parasite infection resistance, in addition to unique genetic variation. They also occupy unique ecological conditions and serve (along with the western North American ACU) as the ancestral origin for the species worldwide.

   Eastern North American monarchs undergo long-distance migration every fall, a behavior that differentiates this population from other non-migratory populations or from migratory populations that fly shorter distances and to different locations. Further, the migratory phenotype of monarchs in the eastern ACU is distinct from monarchs in other ACUs that may have latent migratory phenotypes (Tenger-Trolander et al. 2019, p. 14673). Experimental comparisons between non-migratory and migratory individuals in the Eastern ACU reveal a unique phenotype present only in the migratory monarchs in the Eastern ACU. This migratory phenotype consists of both reproductive diapause and directional flight orientation to the south, and this migratory behavior of monarchs is remarkably sensitive to genetic and environmental change (Tenger-Trolander et al. 2019, p. 14673). In order to maintain full representation within the eastern North American monarch population, it is crucial to conserve the long-distance migratory phenotype for the unique adaptive capacity this behavior and its associated traits may offer.

   Monarchs from the eastern North American migratory population also have unique physical characteristics. They tend to have larger bodies, and larger and elongated wings compared to monarchs from most non-migratory populations (Altizer and Davis 2010, pp. 1023-1025). Relative to monarchs in western North America, eastern monarchs differ at isolated spots in the genome in relation to flight muscles (Kronforst, M. and A. Tenger-Trolander, pers. comm. 2018). Additionally, within the eastern North American ACU, long-distance migrants tend to have redder coloration (Davis 2009, p. 3). Redder coloration is associated with the ability to fly for longer periods of time, although the mechanism for this correlation is unknown (Davis et al. 2012, p. 4). Furthermore, compared to monarchs in the western North American ACU and the southern Florida ACU, eastern North American monarchs have lower rates of infection by the protozoan parasite *Ophryocystis elektroscirrha* (*OE*) (<10%; Altizer et al. 2000, p. 131), which may be due in part to their long-distance migration (Bartel et al. 2011, p. 348). Eastern monarchs migrating to Mexico also have higher lipid reserves than those overwintering in California (Brower et al. 1995, p. 542) and may have a longer diapause compared to monarchs from the western North American ACU (Herman et al. 1989, pp. 56-57).

   Monarchs from the eastern North American ACU also differ from monarchs in other ACUs in their overwintering habitat use and requirements. These monarchs overwinter in the mountainous forests composed primarily of oyamel fir roosting trees (Slayback and Brower 2007, pp. 147-148; Williams and Brower 2015, pp. 109-110), which provide a protective microclimate that is unique relative to those used by overwintering monarchs in other ACUs (Brower et al. 1995, p. 542).

Migratory monarchs in North America are the ancestral population for all other monarch populations around the world (Pierce et al. 2014a, p. 4; Zhan et al. 2014, p. 318). Their unique genetics separate them from non-migratory monarchs within North America (e.g., southern Florida; Pfeiler et al. 2016), as well as populations for the other ACUs described below. While some results show that the monarchs from eastern and western North American ACUs form an admixed population (Lyons et al. 2012, p. 3441), the differences in biological traits and ecological conditions they occupy warrant separating the populations into two ACUs.

2. Western North America:

Western North American monarchs form a separate ACU because they contribute unique variation in migratory behavior, ecology, reproductive behavior, wing morphology, flight performance, and disease/parasite resistance. In addition, along with the eastern North American ACU, the western North American ACU serves as the ancestral origin for the species worldwide (Pierce et al. 2014a; Zhan et al. 2014).

Like the monarchs in the eastern North American ACU, monarchs in the western North American ACU possess the unique migratory phenotype that is absent in the other six ACUs (Tenger-Trolander et al. 2019, p. 14673). Western North American monarchs also migrate long distances, although their migration is shorter than monarchs in the eastern North American ACU. Whereas eastern monarchs may fly well over 3,000 km to reach the Mexican overwintering sites, western monarchs reach the California coast by flying ~500 km to 1,600 km (Yang et al. 2016, p. 1002). Western monarchs occupy warmer climates throughout the summer to include the farther reaches of their range while they continue to breed in the hotter regions (expand their range). Eastern monarchs, in contrast, follow more of a stepping-stone path into the northern states, vacating areas as they warm and recolonizing their range.

Additionally, western monarchs use ecologically different breeding, migrating, and overwintering habitats (Brower et al. 1995, p. 542), and the western North American ACU comprises as much as 30% of the area occupied by monarch butterflies in North America (Dilts et al. 2019, p. 11). Differences in breeding habitat include climate (Zalucki and Rochester 2004, pp. 220-221) and availability and abundance of native nectar and native milkweed plants (Borders and Lee-Mäder 2015, pp. 190-196). It is hotter and drier in the west than the east, and the milkweed and nectar resources used by monarchs in west and east differ (Dilts et al. 2019, entire). In the fall, western monarchs migrate from Canada and states west of the Rockies to overwintering groves located primarily along the California coast south into Baja California, Mexico (Jepsen and Black 2015, pp. 147-156). Roosting tree species used by western monarchs are different than those of the eastern population, and include blue gum eucalyptus, Monterey pine, and Monterey cypress (Griffiths and Villablanca 2015, pp. 43-44). There are fewer monarchs in the western population, spread out among hundreds of overwintering sites compared to fewer than 20 sites in Mexico for the eastern population (Jepsen and Black 2015, pp. 147-156; Vidal and Rendón-Salinas 2014, entire).

In addition to differences in migratory behavior and habitats occupied, the designation of a separate ACU for western North American monarchs is supported by variation in reproductive behavior, wing morphology, flight performance, and disease/parasite resistance. Western North American overwintering monarchs may have a shorter diapause compared to those in eastern North America (Herman et al. 1989, pp. 52-54), and there may also be differences in mating behavior at the western overwintering grounds compared to the eastern overwintering grounds (Brower et al. 1995, p. 542). Eastern and western North American monarchs have divergent wing morphology (see the Eastern ACU discussion above, Freedman and Dingle, 2018, p. 66) and differences in flight performance resulting from differential gene expression related to non-muscular motor activity (Talla et al. 2020, p. 2572-2573). Monarchs in the west have *OE* infection rates (averaging 5-30%) that are lower than most non-migratory populations but higher than the rates of infection in eastern North America (Altizer and de Roode 2015, p. 91).

Thus, in order to maintain representation within the western North American monarch population, it is crucial to conserve the long-distance migratory phenotype in the west for the unique adaptive capacity this behavior and its associated traits may offer.

3. Southern Florida:

Southern Florida monarchs form a separate ACU because they contribute unique variation primarily in genetics and phenotypic characteristics of non-migratory behavior, year-round breeding, and resistance to both high OE loads and a different strain of OE.

Monarchs in southern Florida live in areas where the climate permits year-round breeding, and thus are able to reside continually without migrating. These non-migratory monarchs are genetically distinct from the migratory North American monarchs, although the southern Florida population gets an annual influx of individuals from the eastern monarch population (Knight and Brower 2009, p. 821; Zhan et al. 2014; Pfeiler et al. 2016). Non-migratory Florida monarchs experience some of the highest recorded OE infection rates compared to other monarchs worldwide and particularly high rates compared to eastern and western North America monarch infection rates (75-100% average infection rates in Florida vs. 5-30% infection rates in the western North American population and less than 10% infection rates in the eastern North American population; Altizer and de Roode 2015, p. 91). This may be due both to their inability to escape infected habitat, as well as the non-migratory behavior not leading to any migratory culling (Bartel et al. 2011, entire). Sternberg and colleagues (2013, pp. E239-E241) further determined that in lab settings, monarchs from southern Florida had lower OE spore loads (relative to eastern migratory monarchs) and were less likely to become infected with OE, potentially indicating that non-migratory southern Florida monarchs have increased resistance to OE (however, see also Altizer 2001, p. 622). In cross-population laboratory experiments, the OE parasites from southern Florida caused higher parasite loads than those from the eastern population (Altizer 2001, p. 622). For additional information, see Disease and Natural Enemies in Chapter 6.

4. South America and Aruba:

Monarchs in South America and Aruba are grouped together to form an ACU due to genetic uniqueness.

Monarchs in South America, based on samples from Ecuador, are markedly distinct from other populations of monarchs when analyzing microsatellite markers (Pierce et al. 2014a, 2015). They are occasionally classified as a separate subspecies (Danaus plexippus nigrippus).

While there is some indication that monarchs in Aruba are genetically distinct from South American monarchs (Pierce et al. 2014a), there is also evidence to the contrary (Zhan et al. 2014). Thus, based on this and on expert input suggesting that the small Aruba population is probably not genetically or ecologically distinguishable from South American monarchs, we grouped Aruba (and nearby islands) and South American monarch into the same ACU.

5. Central America and the Caribbean:

   Central American and Caribbean monarchs are grouped together to form an ACU based on genetic and behavioral differences relative to monarch elsewhere.

   Microsatellite analyses showed that Caribbean and Central American monarchs are distinct from South American monarchs and other non-migratory monarchs (Pierce et al. 2014a), and single nucleotide polymorphism analysis showed that Caribbean and Central American monarchs are also genetically distinct from the two migratory North American monarch populations (Zhan et al. 2014). Given that monarchs in Central America and the Caribbean are genetically distinct from these other populations and given the uniqueness of the southern Florida population (outlined above), we classified these monarchs as a separate ACU. Mexican non-migratory monarchs were also included in this unit (rather than the eastern North American ACU), based on similar ecological habitat, behavior (lack of migration), and recent genetic work showing genetic differentiation between migratory and non-migratory Mexican monarchs (Pfeiler et al. 2016).

6. Australia, New Zealand, and other Pacific Islands:

   Monarchs across Australia, New Zealand and other Pacific Islands are grouped together to form an ACU based on genetic characteristics and phenotypic characteristics of migration and disease/parasite resistance.

   Monarchs are found on many islands throughout the Pacific Ocean, including larger populations in Australia and New Zealand. Microsatellite analyses of monarchs in several Pacific island locations (Australia, New Zealand, New Caledonia, Fiji, and Samoa) indicate that these monarchs are genetically distinct from other areas and have lower allelic diversity than North American monarchs (Shephard et al. 2002, entire; Pierce et al. 2014a, p. 4). In addition to genetic differences, monarchs in the Pacific Islands show variation in migratory behavior. Monarchs on most of the smaller islands are non-migratory, but some Australian monarchs in New South Wales have been shown to migrate up to 380 km in autumn (James 1993, p. 193). However, there is little evidence for a regular long-distance migration, making it unique from the migration of the western and eastern North American monarchs (James 1993, p. 190).

   Researchers working with non-migratory Australian monarchs also discovered unique phenotypic responses upon exposure to environmental conditions thought to induce migration. Non-migratory monarchs exposed to cooler temperatures and shorter day lengths showed longer larval development periods, greater adult mass (thought to represent greater lipid reserves), and longer forewing development, all characteristics associated with potentially regaining the migratory phenotype (Freedman et al. 2017, p. 7, 10). Additionally, these responses varied significantly between the offspring of different mothers, suggesting that a migratory phenotype is potentially present within that Australian population (Freedman et al. 2017, p. 7, 10). Finally, incidence of OE in Australia is higher than in most other populations (~66% infection rate; Barriga et al. 2016, p. 76).

7. <u>Hawaii</u>:

   Hawaiian Island monarchs form an ACU because of unique genetic variation and increased disease/parasite tolerance.

   Monarchs exist on all major Hawaiian Islands and are non-migratory. Analysis using single nucleotide polymorphisms shows that monarchs in Hawaii are genetically distinct from other worldwide populations (Zhan et al. 2014). Microsatellite analyses also indicate that Hawaiian monarchs are genetically distinct from populations outside of Hawaii and that they have lower allelic diversity than continental North American monarch populations (Pierce et al. 2014b). Additionally, work indicates that monarchs in Hawaii form an admixed population (suggesting movement among islands; Pierce et al. 2014b). Monarchs in Hawaii persist with only moderate fitness reduction under strains of OE that are both more virulent and more prevalent than that of North American monarchs (Sternberg et al. 2013, p. E239). Thus, monarchs in the Hawaiian ACU contribute unique variation to the species in resistance to OE.

8. <u>Iberian Peninsula (including Spain, Portugal, Morocco, and nearby Atlantic islands)</u>:

   Monarchs on the Iberian Peninsula (Spain and Portugal), along with monarchs in Northern Morocco and nearby Atlantic Islands, form an ACU because of unique genetic variation and ecological and climatic conditions.

   The non-migratory, introduced monarchs in Spain, Portugal, and Morocco form a genetically distinct, derived population based on a single nucleotide polymorphism analysis of the entire monarch genome (Zhan et al. 2014, p. 2). There may be some genetic variation between the Spanish monarchs and the monarchs in Portugal and Morocco based on microsatellite analyses (Pierce et al. 2014a). However, we did not consider Spanish monarchs as a separate ACU because these monarchs occupy very similar ecological and climatic conditions to the rest of the monarchs in this ACU (Fernández-Haeger et al. 2015, entire) but differ from those of other ACUs.

Chapter 3: Methodology

This chapter describes our methods for assessing viability of the monarch over time. The specific methodology for each step of the framework is described below. Briefly, our approach entailed: 1) gathering occurrence data globally, 2) assessing the number, health, and distribution of populations historically and currently, 3) identifying the substantive factors leading to the species' current condition and predicting the future states of these influences, 4) forecasting the health and distribution of populations given the future states of the influences, and 5) evaluating the resulting change in resiliency, redundancy, and representation over time and the implications for the species' viability (Figure 3.1).

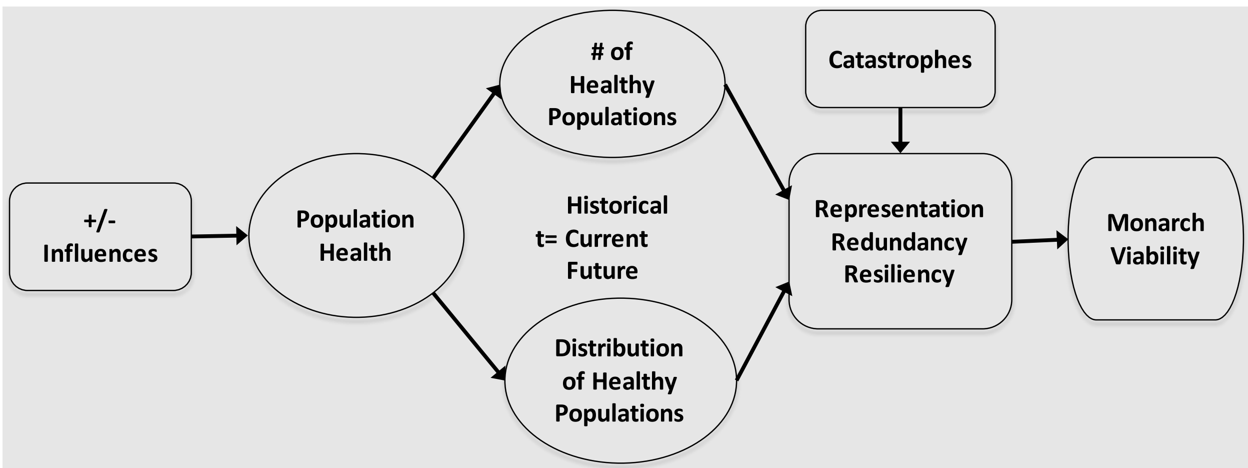

Figure 3.1. Simplified conceptual diagram depicting the analytical framework for assessing monarch viability over time.

Historical Condition: Number, health, and distribution of monarch populations (Ch. 4)

We examined the published literature to determine the historical distribution of the monarch butterfly populations. In order to assess the change in the number, health, and distribution of monarch populations over time, we delineated populations as follows. The monarchs in North America were separated into three populations—eastern, western, and southern Florida—based on distinct behavioral differences and limited movement between these populations. While differences at neutral markers have not been found between the western and eastern populations, a recent genomics analysis indicates low levels of dispersal between eastern and western monarch butterflies suggesting that they are demographically independent (Talla et al., 2020). The monarchs in the Caribbean, Central America, and South America were grouped according to documented genetic differences (Pierce et al. 2015). The remaining monarch locations were delineated based on distance. For monarchs occurring in countries and islands in the Pacific, monarch locations separated by more than 200 miles were considered disjunct populations. Tagging and observational data suggest that monarchs can travel up to approximately 70-75 miles a day during migration, with the longest recorded flight of a tagged eastern North American monarch at 265 miles (Journey North 2018). We thus chose a distance of 200 miles for separating populations because it was at the upper limits of the range of observed distances flown by tagged monarchs, and it is unlikely that monarchs separated by 200 miles or more could successfully move among these locations. If the distance between islands was less than 200 miles, we assumed that movement between islands was plausible and thus did not consider the islands as disjunct populations.

To assess population health, we sought out information on historical population abundance (N) and population growth rate (λ). Population size (N) estimates were derived from published survey counts; eastern North American monarchs have been surveyed yearly using a standardized protocol at the Mexican overwintering sites since 1994 (Monarch Watch 2020) and the western North American population has been monitored since 1997 at coastal overwintering sites in California (Xerces Society for Invertebrate Conservation 2020). The historical population growth rates (λ) for eastern and western North American populations were available from Semmens et al. (2016) and Schultz et al. (2017) for the eastern and western populations, respectively, and we updated both to reflect changes in growth rates since publication. Prior to 1994, we have limited information on population size (N) or growth (λ), but assume both populations were healthy (i.e., λ and N met conditions of Table 2.2) at some point in the historical time period. For all other populations, there are no systematic, multi-year surveys for any time period, so we assume those populations were healthy at some point in the historical time period as well.

Current & Future Conditions: Number, health, and distribution of monarch populations (Ch. 5 and Ch. 6)

To assess the current and future number, health, and distribution of monarch populations, for each population we: 1) determined the current abundance and population trend (λ), 2) identified the current and likely future primary influences, and 3) forecasted the change in health given these influences. We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) drivers of monarch population numbers. We identified the following drivers: disease/natural enemies; herbicides; logging/tree loss; habitat degradation (succession, western overwintering site aging of trees); climate change (drought, storm events, temperature extremes); collection/tourism; grazing/incompatible farming; change in nectar and milkweed resources; loss of urban/greenspace; mowing; insecticides; change in western overwintering habitat. Of these, we identified the subset that are the key drivers influencing monarch dynamics (referred to as influences). We carried this subset through the rest of our analyses. For the worldwide populations, we researched potential issues related to land use change, insecticides, and disease.

Population-specific information for monarchs varies from highly detailed data for the eastern and western North American populations to very limited data (occurrence only) for most of the other 29 populations. To fully apply the best available data, we developed a population model for the eastern and western populations while using a coarser-scaled, qualitative approach for the remaining populations. We refer to the non-eastern, non-western populations as “worldwide populations.”

Worldwide Populations

To assign status, we categorized populations based on last date observed and survey effort. We assumed that all populations in which at least a single monarch has been reported since the year 2000 are extant today and were assigned ‘extant’ status. Populations lacking a sighting since the year 2000 and lacking multi-year survey efforts were assigned ‘unknown’ status (neither extant nor extirpated). Populations lacking sightings with multiple years of surveys were assigned ‘extirpated’ status. We garnered the available data by: 1) searching for records in Google Scholar using each known country with a historical monarch occurrence and the phrase “*Danaus plexippus*” as search terms; 2) requesting personal knowledge and unpublished information regarding monarch occurrence from international entomologists and species’ experts; and 3) searching geotagged photos on Flickr and reports from the citizen science database iNaturalist for monarch records. We did not use these records if we could not verify the species, or if the photo appeared to have been taken in a butterfly exhibit (potentially with non-native butterflies present).

In absence of demographic data, we assessed the current health of each worldwide population by evaluating the past trend in population numbers, the current status of milkweed and nectar resources, the current levels of insecticide exposure, and the current status of overwintering habitat. We compiled these data and assigned a population condition category of ‘high,’ ‘moderate,’ ‘low,’ or ‘unknown’ for each population. Condition categories were assigned using the descriptions presented in Table 3.1 (for similar condition category table approaches, see NatureServe 2013; IUCN 2018; and Puget Sound Stream Benthos 2018). If the information available was insufficient to assign a condition category, the population was marked as unknown status (Table 3.1).

Table 3.1. Categories used to define the health of the worldwide populations. Unknown indicates insufficient information about habitat quality, quantity, and corresponding monarch population trends.

Condition Rating	Past Trend	Current status of Milkweed and Nectar	Current status of Insecticides	Overwintering Habitat
High	$\lambda > 1$	Milkweed/Nectar not thought to be limiting monarch numbers	Current level of insecticide exposure to and/or toxicity of insecticides not thought to impact population-level	Overwintering habitat quality and quantity not thought to be limiting monarch numbers
Moderate	$\lambda \approx 1$	Milkweed/Nectar resources have been lost and are limiting monarch numbers in some portion of the population	Current level of insecticide exposure to and/or toxicity of insecticides limiting monarch numbers in some portion of the population	Overwintering habitat quality and quantity are limiting monarch numbers in some portion of the population
Low	$\lambda < 1$	Milkweed/Nectar resources have been lost and are limiting monarch numbers throughout the entire population	Level of insecticide exposure to and/or toxicity of insecticides are limiting monarch numbers throughout the entire population	Overwintering habitat quality and quantity are limiting monarch numbers throughout the entire population
Unknown	Unknown	Unknown	Unknown	Unknown

To assess future health of the worldwide populations, we searched the published literature and contacted international lepidopterists to identify the primary influences. For most influences (e.g., insecticides, land cover change, etc.), there was insufficient information to make an assessment.

Eastern & Western North American Populations

Unlike the worldwide populations, there are 20+ years of standardized survey data from which we can derive current abundance and population trend (λ) for eastern and western North American monarch populations. Thus, to assess the current and future health of these populations, we used published stochastic, geometric growth models for eastern (Semmens et al. 2016) and western (Schultz et al. 2017) populations. We updated the models with population data obtained since 2015 and incorporated the future state conditions of the influences (Figure 3.2). We briefly describe our models here; for additional detail see Voorhies et al. (2019) and see Appendix 2 for a list of small improvements made since the publication of Voorhies et al. (2019).

Our models assume that next year's population size in their wintering grounds, $Nt+1$, is a function of the monarch population size in the current time-step, Nt, and their log population growth rate, λ. To incorporate future threats and conservation actions into monarch population projections we added an additional term, δ, which represents a net change in population size (N) due to both positive and negative influences. We used published data, expert knowledge, and professional judgment to project the expected future state of each influence. To capture the uncertainty in our future state projections, we identified plausible optimistic and pessimistic changes for each influence. The most optimistic and pessimistic states for each influence were then combined to create composite plausible "best case" and "worst case" scenarios.

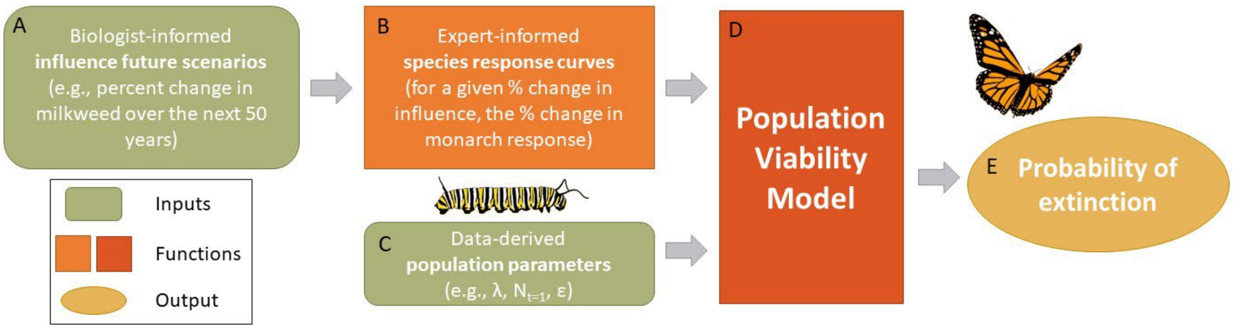

Figure 3.2. An overview of the monarch modeling framework. Biologist-informed scenarios (A) represent expected range in % change in a given influence over time. Expert-elicited population response curves (B) specific to each influence provide the proportional change in monarch response given a proportional change in the influence. Population response curves differ by influence and region (eastern and western populations). Population demographic data (C) were sourced from existing literature and used to initialize the model (D), which also received inputs from (B). Simulation outputs from the population viability analysis were compared against a range of extinction threshold values (E) to estimate the cumulative pE over time.

The health metric, pE, reflects the probability of the population size dropping below a threshold at which extinction would become inevitable (via a mechanism known as an extinction vortex). As others have done (e.g., Flockhart et al. 2015, p. 159; Semmens et al. 2016, p. 2; Schultz et al. 2017, p. 345), the extinction threshold is our primary mechanism for incorporating the consequences of Allee effects and environmental stochasticity at small population sizes. In addition to the extinction threshold, we introduced a population cap to address the limitation of a density-independent growth model which, as noted by Courchamp et al. (1999, p. 408), implicitly assumes populations increase linearly to carrying capacity.

Mechanisms that may trigger an extinction vortex in monarch populations include the following component effects:

- reduced survival on the overwintering grounds (Williams and Brower 2015; Berec et al. 2007, p. 187)
- increased predation on the overwintering grounds (Berec et al. 2007, p. 187; Brower and Calvert 1985, p. 857 and 861; Calvert et al. 1979, p. 849)
- reduced reproduction (e.g., mating depression due to difficulty finding mates [Berec et al. 2007, p. 187] and the subsequent reduction of female overwintering survival due to additional nutrients from multiple matings [Wells et al. 1993, p. 66])
- inability of small population sizes to rebound from sustained threats (Hutchings 2015, p. 6) or natural environmental variation (e.g., poor weather years)

The extinction thresholds for the eastern population were derived from expert-elicited estimates. We defined our lower and upper bounds for the extinction threshold as the median across the experts' "lowest" (0.05 ha) and "highest" (0.61 ha) estimates. For the western population, we used extinction thresholds reported in the literature. Our lower bound was set at 20,000 individuals (Schultz et al. 2017) and the upper bound at 50,000 (Wells et al. 1990). We assumed that all values between the lower and upper bounds were equally probable; thus, we used the upper and lower estimates to set the bounds of a uniform distribution (refer to Voorhies et al. 2019 for further discussion).

We calculated starting population size by taking the average of the last 5 years and calculated population growth rate (λ) and environmental stochasticity value (epsilon; $\mathcal{E}$) by using the Semmens et al. (2016) and Schultz et al. (2017) models, respectively, and updating the population data and time period. All input values are provided in Appendix 2.

Viability (Ch. 8)

To describe monarch viability over time, we evaluated how the change in the number, health, and distribution of monarch populations from historical to present to future influences the resiliency, redundancy, and representation of monarchs.

We used the results from our current and future forecasts--specifically the change in the number, health, and distribution of monarch populations over time--to evaluate the species' resiliency to environmental stochasticity, disturbances, and stressors. To assess monarch's redundancy, we qualitatively assessed how the current and forecasted number and distribution of populations affect the risk of catastrophic losses within each ACU. A catastrophe is an event that is outside the normal range of variation for a stressor and for which adaption is unlikely (Mangal and Tier 1993, p. 1083), and therefore, inevitably leads to population collapse (extinction).

For the eastern North American population, we identified overwintering habitat loss, monarch disease, widespread drought, extreme storm events (both at the Mexican overwintering sites and during migration funnel points), and widespread insecticide spray events as potential catastrophic events. Of these, we found reliable evidence for widespread drought and extreme storm events as sources for causing catastrophic losses, and thus, were carried forward in our analyses. For the western North American population, we identified extreme widespread drought, disease, severe storms events, wildfire, widespread milkweed loss, widespread insecticide spray events, and co-occurrence of a poor environmental conditions and low population abundance as potential catastrophic events. Of these, we found reliable evidence for widespread drought and the co-occurrence of poor environmental conditions and low population abundance as sources for causing catastrophic losses, and thus, were carried forward in our analyses.

For the worldwide populations, we identified climate change induced sea level rise and maximum temperature increases as potentially catastrophic events. We classified risk as either "No Known Risk" or "At Risk" (Table 3.2). Using the Third Assessment Report developed by the International Panel on Climate Change (IPCC), we determined which low-lying islands occupied by monarchs may be at risk of permanent inundation, and used the maximum elevation of those islands to develop thresholds for the risk classifications (IPCC 2001). We also qualitatively assessed where daily maximum surface temperatures exceeding 42°C (a temperature threshold that leads to mortality; Nail et al. 2015b, p. 99) are projected to increase by the year 2069 (~50 years from now) under Representative Concentration Pathways (RCP) scenarios 4.5 and 8.5 using climate projections obtained from the Earth System Grid Federation (Cinquini 2014). Given scale and magnitude of impact (whether population would be exposed to events that would lead to population extinction), this analysis falls under a catastrophic risk.

Table 3.2. Categories used to define the risk of the worldwide populations to predicted climate change impacts.

Future Influence	**Risk Category**	**Definition**
Sea Level Rise	No Known Risk	Not at low elevation (highest point >100m above sea level).
Sea Level Rise	At Risk	Very low elevation (highest point ≤100m above sea level) and single location represents an entire population.
High Temperatures	No Known Risk	Number of days and/or areas with daily maximum surface air temperatures above lethal levels (42˚C) not projected to increase under moderate (RCP 4.5) or severe (RCP 8.5) scenarios.
High Temperatures	At Risk	Number of days and/or areas with daily maximum surface air temperatures above lethal levels (42˚C) are projected to increase under the moderate (RCP 4.5) or severe (RCP 8.5) scenarios.

Lastly, we evaluated the monarch's ability to adapt to novel changes in its physical and biological environment by assessing the likelihood of monarchs persisting in each of the 8 ACUs given the forecasted influences and catastrophes. Specifically, for the eastern and western North American ACUs, we used the results of our population modeling to predict the likelihood of persistence of monarchs within both ACUs over the next 50 years. For the remaining 6 ACUs, we qualitatively express the likelihood of persistence within each of the 6 ACUs over the next 50 years given the risks of catastrophic sea level rise or high temperature conditions.

Chapter 4: Results – Analysis of Historical Condition

This chapter describes the number, health, and distribution of monarch populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in monarch viability over time.

Worldwide

There are no reliable records of monarchs outside of continental North America or the Caribbean before 1840 (Vane-Wright 1993, p. 180). However, by 1883 the monarch was reported as one of the most common butterflies in many Pacific Islands (Walker 1914, p. 187). Host plants used by monarchs in these non-North American locations include *Asclepias* spp., *Gomphocarpus* spp., and *Calotropis* spp. (all either milkweed or closely related genera; Blakley and Dingle 1978, p. 134; Buden and Miller 2003, p. 4). It is generally accepted that both monarchs and milkweed dispersed from North America via human assistance, potentially aided through wind dispersal events (Brower 1995, p. 354). For the purposes of our analysis, we assume that monarchs in locations outside of North America have become naturalized, and thus, these records, along with the North American occurrences, comprise the historical range of the species (Figure 4.1).

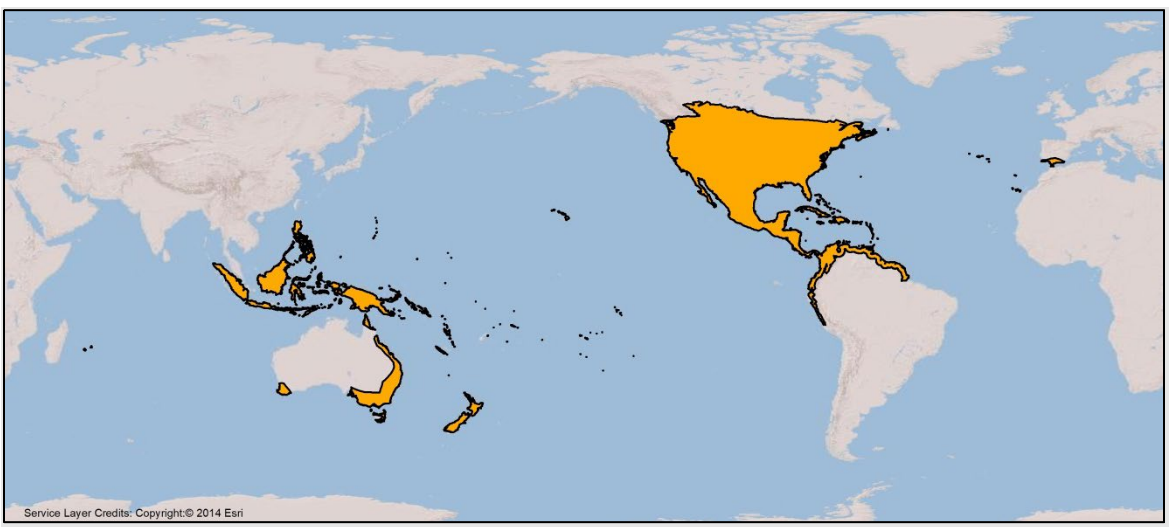

Figure 4.1. Map showing global range of monarchs (orange shows known range).

We found monarch occurrence records in 90 countries, islands, or island groups. We delineated these occurrences into 31 different populations (Table 4.1). We assume that at some point in the past, all populations were healthy. Table 4.1 also shows how these 31 populations are distributed among the eight ACUs (see Chapter 2 for description of the ACUs and how they were delineated). This organization is visually represented in Figure 4.2. While the Australia, New Zealand, and Indo-Pacific Islands ACU appears the largest in spatial extent, the eastern North American population has the most individuals (even accounting for large variation in estimates; Figure 4.3).

Table 4.1. The 31 delineated monarch populations, with their associated ACUs and the countries and islands that comprise each population.

ACU	Population	Countries/Islands within Population
Australia, New Zealand, and Indo-Pacific Islands	Australia	Commonwealth of Australia
	Cook Island	Cook Islands
	French Polynesia	French Polynesia
	Greater Indonesia	Nation of Brunei, Republic of Indonesia, Malaysia, Democratic Republic of Timor-Leste
	Guam & CNMI	Guam, Commonwealth of Northern Mariana Islands (CNMI)
	Johnston Atoll	Johnston Atoll
	Kiribati	Republic of Kiribati
	Marquesas Islands	Marquesas Islands
	Marshall Islands	Republic of the Marshall Islands
	Mascarene Islands	Republic of Mauritius, Réunion
	Micronesia	Federated States of Micronesia
	Nauru	Republic of Nauru
	New Zealand	New Zealand
	Norfolk Island	Norfolk Island
	Palau	Republic of Palau
	Papua New Guinea	Independent State of Papua New Guinea
	Philippines	Republic of the Philippines
	Samoa	American Samoa, Samoa
	South Pacific Islands	Republic of Fiji, New Caledonia, Society Islands, Solomon Islands, Vanuatu
	Tokelau	Tokelau
	Tonga	Kingdom of Tonga
	Tuvalu	Tuvalu
	Wallis & Futuna	Territory of the Wallis and Futuna Islands
Central America & the Caribbean	Caribbean	Anguilla, Antigua and Barbuda, Bahamas, Barbados, Bermuda, Bonaire, British Virgin Islands, Cayman Islands, Cuba, Dominica, Dominican Republic, Grenada, Guadeloupe, Haiti, Jamaica, Martinique, Montserrat, Puerto Rico, Saba, Saint Barthélemy, Sint Eustatius, Saint Kitts and Nevis, Saint Lucia, Saint Martin, Saint Vincent and the Grenadines, Sint Maarten, Turks and Caicos Islands, U.S. Virgin Islands
	Central America	Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua, Panama, Mexico
S. Florida	Florida	United States (FL)
Hawaii	Hawaii	United States (HI)
Iberian Peninsula	Iberian Peninsula	Azores, Canary Islands, Gibraltar, Madeira, Morocco, Portugal, Spain
South America & Aruba	South America and Aruba	Aruba, Colombia, Curacao, Ecuador, French Guiana, Guyana, Peru, Suriname, Trinidad and Tobago, Venezuela
E. North America	Eastern North America	Canada, Mexico, Saint Pierre and Miquelon, United States (East)
W. North America	Western North America	Canada, United States (West), Mexico

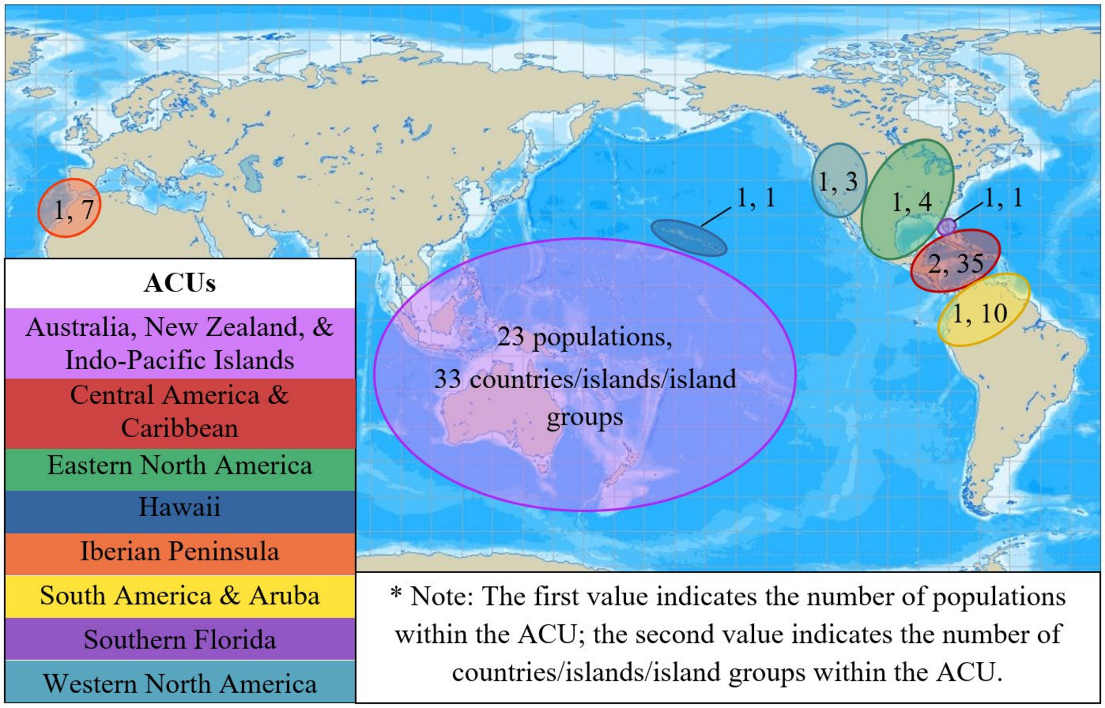

Figure 4.2. Generalized map of the eight ACUs, with the number of populations and countries contained within each ACU provided. Note that the total number of countries/islands/island groups do not add up to 90 because some are present in multiple ACUs.

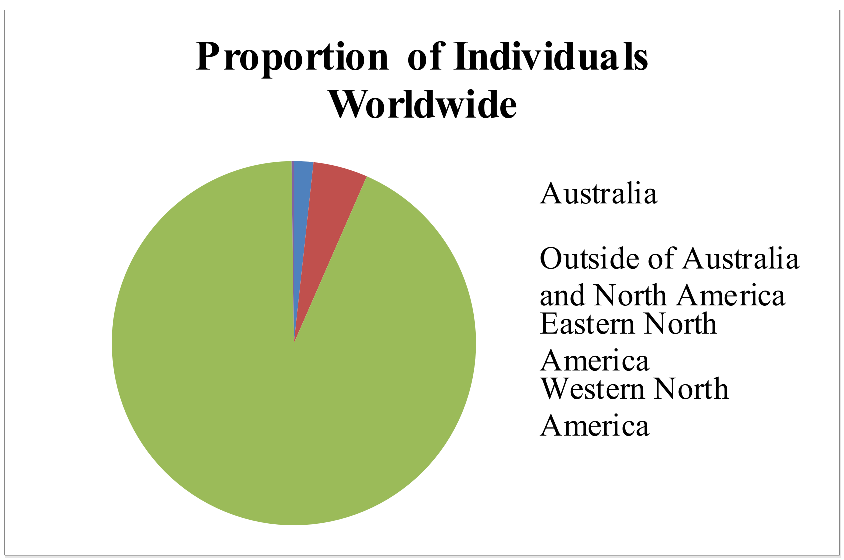

Figure 4.3. Estimated relative proportion of individual monarchs by geographical area. The numbers are based on the following: eastern North America (77,141,600; based on average of last 5 years overwintering estimates, assuming a 21.1 million monarch/ha density), western North America (168,365; based on average of past five years of overwintering counts); Australia (1,424,790; based on estimates from M. Zalucki, pers. comm.); and outside of Australia and North America (4,000,000; based on 3-5 million monarch estimate; M. Zalucki, pers. comm.).

Note that throughout the rest of the document, when the term 'worldwide' is used in relation to monarchs, we are referring to 29 monarch populations excluding the eastern and western North American populations.

Eastern North American Population

The eastern North American monarch population has been systematically censused annually since 1994 (Figure 4.4; Vidal and Rendón-Salinas 2014, pp. 167-168). Although varying year-to-year, monarchs consistently numbered in the hundreds of millions throughout the 1990s and early 2000s (assuming a 21.1 million monarch/hectare density; Thogmartin et al. 2017a, p. 1). There are additional survey data suggesting that monarch populations were as high or higher in the two decades prior to standardized monarch monitoring at the Mexican overwintering sites (Vidal and Rendón-Salinas 2014, p. 172, Calvert and Brower 1986, pp. 167-169).

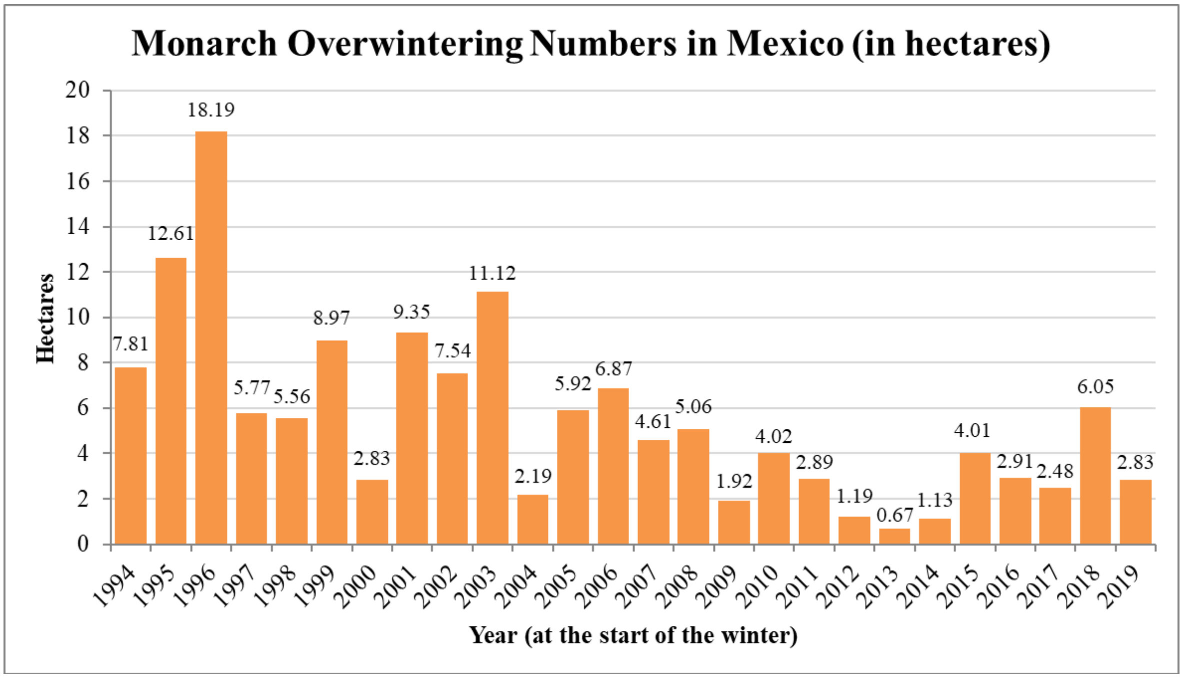

Figure 4.4. Area occupied (in hectares) by eastern North American monarch butterflies at overwintering sites in Mexico (actual hectare measurement displayed above each bar). Year displayed is the beginning year for the winter (e.g., 2017 represents the number for the winter of 2017-2018). Data from Monarch Watch (2020).

Western North American Population

The western North American population has been censused annually since 1997, providing an estimate of annual population size (Figure 4.5). Recent work, using past survey data, gives estimates of millions of butterflies in the mid-1980s (Schultz et al. 2017, p. 3).

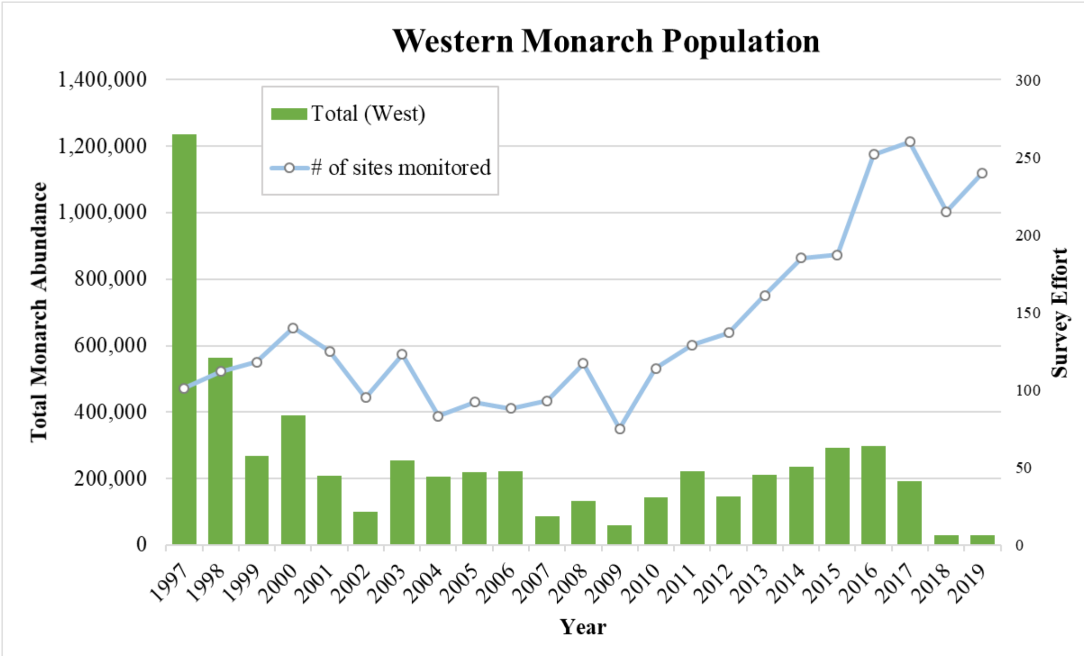

Figure 4.5. Thanksgiving counts showing the number of western North American monarch butterflies observed at overwintering sites (green bars). Blue line shows the number of sites monitored (survey effort) for a given year. Data from The Xerces Society for Invertebrate Conservation 2020, entire.

Chapter 5: Results – Analysis of Current Condition & Current Influences

This chapter describes the number, health, and distribution of monarch populations given current state conditions and describes the influences that have led to this current condition. We present the current condition and influences that led to the condition for the eastern and western North American populations first, followed by the current conditions and influences for the worldwide populations.

Eastern North American Population – Current Condition

Based on the past annual censuses, the eastern North American population has been generally declining over the last 26 years (Figure 4.4). Although the numbers at the overwintering sites have declined, we did not find a corresponding change in the spatial extent of the population during the breeding season. Given its current population size and population growth rate, the

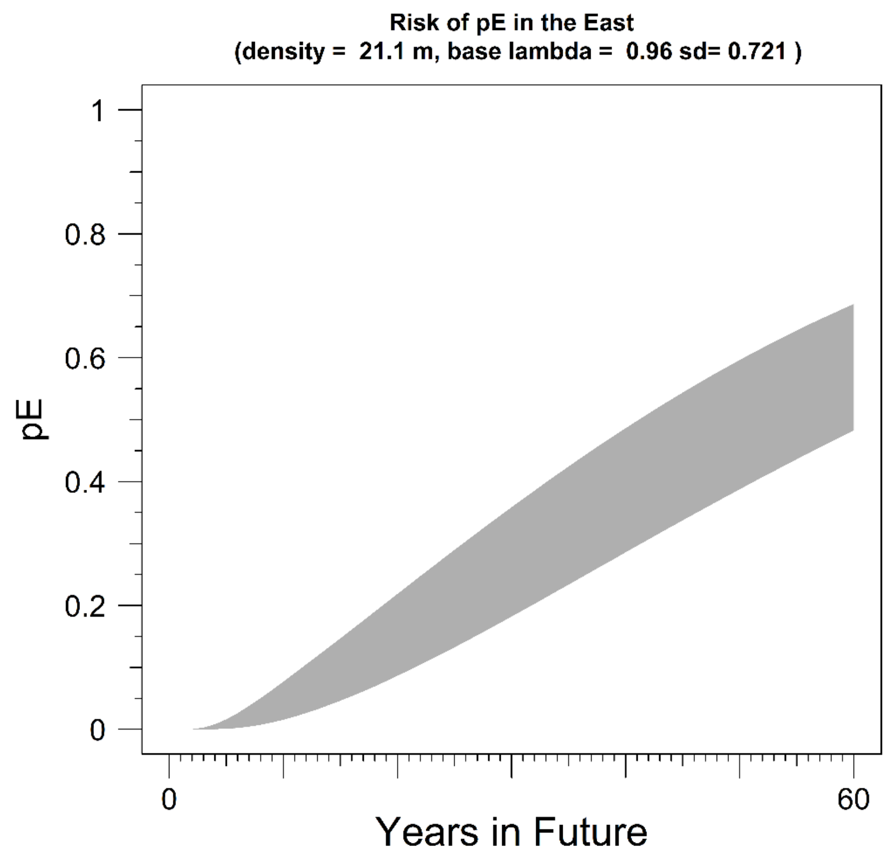

Figure 5.1. pE for the eastern North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.

Western North American Population – Current Condition

Based on the past annual censuses, the western North American population has been generally declining over the last 23 years, despite an increasing number of sites being counted (Figure 4.5). Under current conditions, the risk of extinction over time is predicted to increase sharply, with the pE over 60 years reaching 99% (98%-99%, CI 50%) (Figure 5.2).

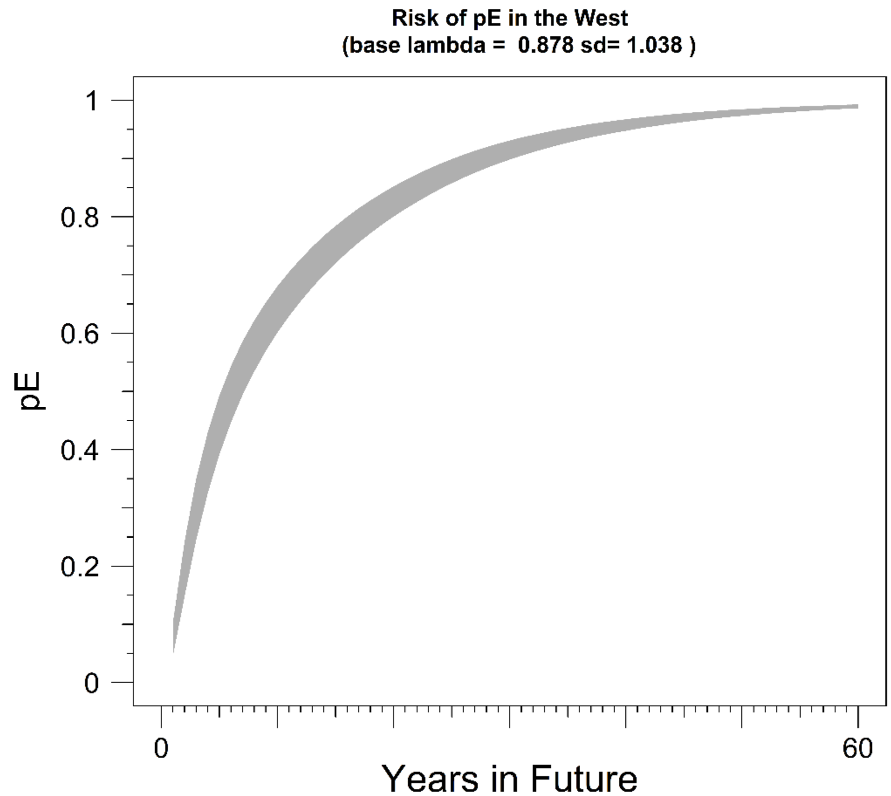

Figure 5.2. pE for the western North American monarch population over time, represented by 50% confidence interval (gray space). Probability based on current trend in growth.

North American Populations – Current Influences

There are a myriad of influences operating on the North American populations. With the assistance of monarch experts, we identified the important factors driving monarch population dynamics for the eastern and western North American populations (Tables 5.1 and 5.2). The primary drivers affecting the health of the two North American migratory populations are changes in breeding, migratory, and overwintering habitat (due to conversion of grasslands to agriculture, urban development, widespread use of herbicides, logging/thinning at overwintering sites, unsuitable management of overwintering groves , and drought), continued exposure to insecticides, and effects of climate change (Figure 5.3). Below, we discuss the key influences on monarch populations—the aforementioned stressors and monarch conservation efforts.

Table 5.1. Expert-elicited rank and extent of impact (% contribution to the decline from the historical period) of the influences on the eastern North American population. % Contribution = median value across experts; the lowest and highest expert judgment among the experts provided in parentheses (see Voorhies et al. 2019, Suppl.2).

Influence	Rank	% Contribution
Availability, spatial distribution, and quality of milkweed	1	25 (10-60)
Availability and quality of overwintering habitat	2	20 (10-30)
Climate (storms, drought, temperatures)	3	12.5 (6-23)
Availability, quality, and spatial distribution of migration resources	4	12 (2-20)
Disease and natural enemies	5	9.5 (1-15)
Insecticides	6	8 (1-10)
Availability, spatial distribution, and quality of nectar resources (breeding)	7	5 (1-10)
Road mortality and pollutants	8	3 (1-5)
Biogeographical scrambling of milkweed spp. (includes non-native spp.)	9	2 (0-4)
Other	10	2 (0-8)
Monarch releases, captive breeding, and translocation	11	1.5 (0-3)

Table 5.2. Expert-elicited rank and extent of impact (% contribution to the decline from the historical period) of the influences on the western North American population. % Contribution = median value across experts; the lowest and highest expert judgment among the experts provided in parentheses (see Voorhies et al. 2019, Suppl.2).

Influence	Rank	% Contribution
Availability, spatial distribution, and quality of milkweed	1	22 (15-25)
Availability, spatial distribution, and quality of nectar resources (breeding)	2	18 (13-20)
Insecticides	3	18 (15-22)
Climate change effects via impacts to habitat	4	17 (10-19)
Availability and quality of overwintering habitat	5	16 (12-18)
Climate change via non-habitat mediated effects	6	8 (3-14)
All others	7	4 (0-7)

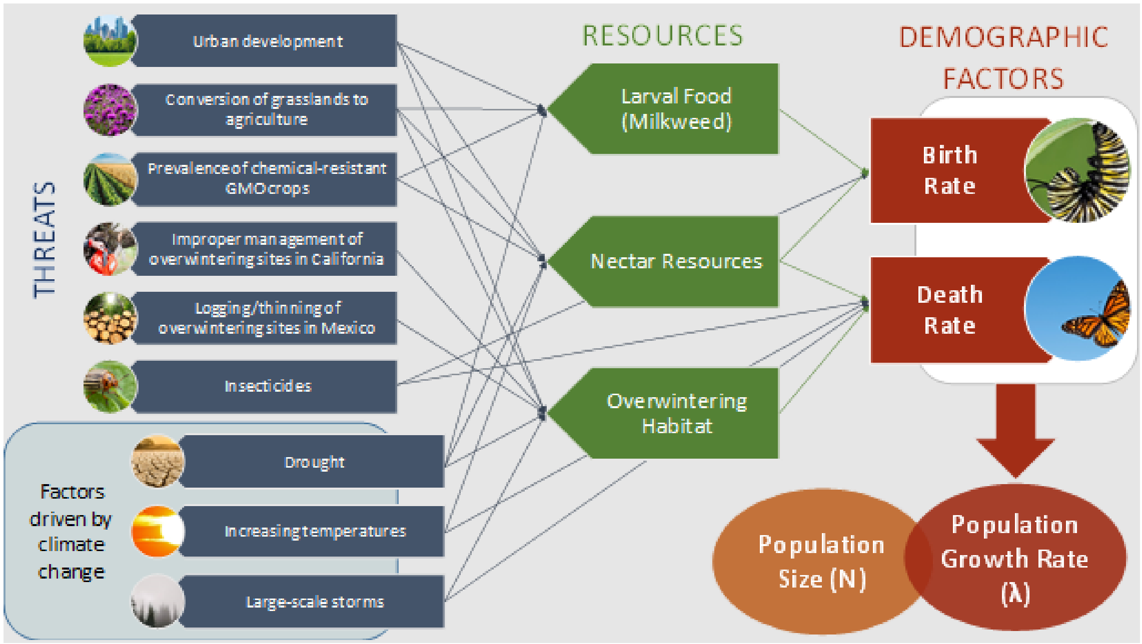

Figure 5.3. Influence diagram showing the key Influences and how they drive monarch population abundance (N) and growth rate (lambda, λ). Note, conservation efforts can decrease all the listed threats and improve all resources for monarchs.

Availability, Distribution, and Quality of Milkweed

The availability of milkweed is essential to monarch reproduction and survival. Reductions in milkweed is cited as a key driver in monarch declines (Brower et al. 2012, p. 97; Pleasants and Oberhauser 2013, p.7; Inamine et al. 2016, p. 1081; Thogmartin et al. 2017b, p.12; Waterbury and Potter 2018, pp. 42-44; Saunders et al. 2019, p. 8612).

A majority of the milkweed loss has occurred in agricultural lands, where intensive herbicide usage for weed control has resulted in widespread milkweed eradication. Pleasants (2017, p. 7), for example, estimated that over 860 million milkweed stems were lost in the Midwest between 1999 and 2014, a decline of almost 40%. Currently, approximately 89% and 94% of corn and soybean crop acreage, respectively, are planted as glyphosate (herbicide)-tolerant crops (USDA 2018). Glyphosate use in western agricultural lands has also increased dramatically since the 1990s, especially within the Central Valley of California, Snake River Plain of Idaho, and the Columbia River Basin, which spans the border between Washington and Oregon (USGS NAWQA 2017; Waterbury and Potter 2018, p. 42). As weed species develop increasing resistance to glyphosate, other herbicide (e.g., dicamba) tolerant crops are developed, which can lead to a corresponding increase in herbicide use. Accordingly, herbicide impacts to milkweed and nectar plants will continue to impact monarch resources.

Milkweed is also lost on the landscape through development and conversion of grasslands (Lark et al. 2015, pp. 3-4). Between 2008 and 2012, a total of 5.7 million acres of grassland were converted to new cropland, including up to 3 million acres of Conservation Reserve Program (CRP) land (Lark et al. 2015, p. 5). Pleasants and Oberhauser (2013, pp. 5-6) estimate that the

loss of agricultural milkweeds in the Midwest has resulted in an 81% decline in monarch production, in part because monarch egg densities were higher on milkweed in agricultural fields (3.89 times more eggs than on non-agricultural milkweed). This particularly impacts the eastern monarch population because more Mexico overwintering monarchs originate from the Midwest crop belt region than any other region (with estimates ranging from 38% to over 85% of all overwintering monarchs originating from the Midwest; Wassenaar and Hobson 1998, pp. 15438-15439; Flockhart et al. 2017, p. 4).
will continue to impact monarch resources available in agricultural lands.

Availability, Distribution, and Quality of Breeding Range Nectar Resources

Reductions in nectar resources are also cited as a potential key driver in monarch declines (Thogmartin et al. 2017b, p.12). Losses of nectar resources are due to same stressors identified above for milkweed resources.

Availability, Distribution, and Quality of Migration Nectar Resources

Losses of nectar sources during migration have also been particularly implicated as a potential key driver in monarch declines (Inamine et al. 2016, p. 1081; Thogmartin et al. 2017b, p.12; Saunders et al. 2019, p. 8612). Losses of nectar resources are due to same stressors identified above for milkweed resources. Additionally, with a warming climate, drought impacts may become more important, especially in the western population and in the migratory bottleneck for the eastern population (see *Climate Change* in Current Influences section within this chapter for more details).

Availability and Quality of Overwintering Habitat

Both western and eastern monarchs rely on the microclimate provided by the trees at their overwintering sites (Leong et al. 2004, entire; Williams and Brower 2015, entire). Loss of trees occurs at overwintering sites in Mexico primarily through small- and large-scale logging, storms, and an increasingly unsuitable climate (see *Climate Change* section below for more details). Most overwintering sites used by eastern monarchs occur within the Monarch Butterfly Biosphere Reserve (Reserve), a 56,259-ha protected area. Within this area, there is a logging ban within the 13,551-ha core zone (Ramírez et al. 2015, p. 158). However, recent logging has occurred both legally (including salvage logging allowed after storms) and illegally at multiple colonies (Vidal et al. 2014, pp. 180-185; Brower et al. 2016, entire).

Logging was estimated by Vidal and colleagues (2014, p. 180) in the core zone of the Reserve from 2002 through 2012. Within this period, 2,179 ha of core zone were deforested (<10% canopy cover remained; 1,254 ha) or degraded (a decrease in canopy cover; 925 ha). Most of these losses were attributed to illegal logging (2,057 ha), with the remaining 122 ha lost due to floods, drought, strong winds, and fire. Current estimates of forest loss throughout the Reserve vary from 0-2.4% per year (Ramírez et al. 2015, p. 163). While anti-logging and reforestation efforts are underway (López García 2011, p. 631), logging is still ongoing within the Reserve (Brower et al. 2016, entire). Although clearcutting of forests destroys habitat directly, thinning of the forest also changes the microclimate needed by overwintering monarchs, making them more susceptible to winter mortality (Brower et al. 2011, p. 43).

Western monarch overwintering habitat along the Pacific Coast has been subject to loss through various forms of development, particularly urban development (Sakai and Calvert 1991, p. 149; Frey and Schaffner 2004, p. 172). Habitat alteration, both natural and anthropogenic, can also alter the microclimate of the western overwintering sites, leading to less suitable habitat conditions (Jepsen et al. 2015, p. 17). There are many other stressors that can work alone or in tandem on the western overwintering sites, including disease and pests that impact the trees used for overwintering, as well as senescence and improper grove management. Fire is also a threat, both indirectly through habitat loss and directly to overwintering monarchs (Pelton et al. 2016, pp. 28, 32). Drought in the West can further exacerbate the stressors on the western overwintering sites (see *Climate Change* section below).

<u>Insecticide Exposure</u>

Insecticides are pesticides with chemical properties that are designed to kill insects, and most are non-specific and broad-spectrum in nature. That is, insects exposed to these insecticides are susceptible to mortality and/or sub-lethal effects. Furthermore, the larvae of many Lepidopterans are considered major pest species and insecticides are tested specifically on this taxon to ensure that they will effectively kill individuals at labeled application rates. Monarchs may also be exposed to insecticides in areas beyond the insecticide application points due to drift (Olaya-Arenas and Kaplan 2019, p. 1; Halsch et al. 2020, p. 3).

The monarch butterfly is widely distributed across the United States, occurring in a variety of urban and rural habitat types that include milkweed plants and other flowering forbs. Insecticide impacts to monarchs are primarily influenced by the extent to which monarchs are exposed to insecticides throughout their range. Although insecticide use is most often associated with agricultural production (for example, between 2005 and 2012, 60% of insecticide applied occurred on agricultural lands, USEPA 2017, p.11), any habitat where monarchs are found may be subject to insecticide use. Insecticides can be used for insect pest control anywhere there is a pest outbreak or for general pest prevention. Homeowners may treat yards and gardens to protect plants from pests or purchase plants from nurseries that sell neonicotinoid-treated plants as ornamentals. Natural areas, such as forests and parks, may be treated to control for insects that defoliate, bore into wood, or otherwise damage trees. Outbreaks of pests such as gypsy moths, Mormon crickets, or grasshoppers may trigger insecticide treatments over larger areas to control populations. Use of insecticides in vector control, especially pyrethroids and organophosphates, may be significant in areas of the country where mosquitoes pose a public health threat or reach nuisance levels. The use of insecticides in the U.S. is ubiquitous; in 2012 for example, expenditures on insecticides topped $5 billion in the United States, with 64 million pounds used for agriculture, home and garden, and other purposes (USEPA 2017, see Tables 2.2 & 3.1).

The most widely used classes of insecticide include organophosphates, pyrethroids, and neonicotinoids. Neonicotinoids entered the market in the mid- to late-1990s (Figure 5.4), and because of their high insecticidal activity at low application rates, they are now the most used class of insecticides in the world (Braak et al. 2018, p. 507). By 2008, for example, neonicotinoid insecticides accounted for 80% of global seed treatment sales (Jeschke et al. 2011, p. 2898), and by 2011, >79% of the corn hectares and 34% to 44% of soybean acreage in the U.S. were planted with neonicotinoid-treated seeds (Douglas and Tooker 2015, p. 5092). Neonicotinoid insecticides are absorbed into plants and distributed throughout their tissues to their stems, leaves, roots, fruits, and flowers. They kill and injure insects by attacking their central nervous system.

Studies looking specifically at dose-response of monarchs to neonicotinoids, organophosphates and pyrethroids have demonstrated monarch toxicity (e.g., Krischik et al. 2015, entire; James 2019, entire; Krishnan et al. 2020, entire; Bargar et al. 2020, entire). Moreover, the magnitude of risk posed by insecticides may be underestimated, as research usually examines the effects of the active ingredient alone, while many of the formulated products contain more than one active insecticide (e.g., Swagger contains bifenthrin and imidacloprid, Krishnan et al. 2020, p. 17, but see Oberhauser et al. 2009, entire). The additional risk posed from compounds added to improve the kill rate (referred to as synergists) are often not assessed. The use of synergists is not uncommon. Olaya-Arenas and Kaplan (2019, p. 13), for example, reported that fungicides (often used as a synergist) were most commonly detected on milkweed samples (e.g., 98% of the milkweed sample in one year contained the fungicide, Propiconazole) and, in many of these cases, co-occurred with insecticides like deltamethrin and thiamethoxam. See *Insecticide Supplemental* for further discussion of the risk of pesticides to the monarch, including data, references, and supporting information.

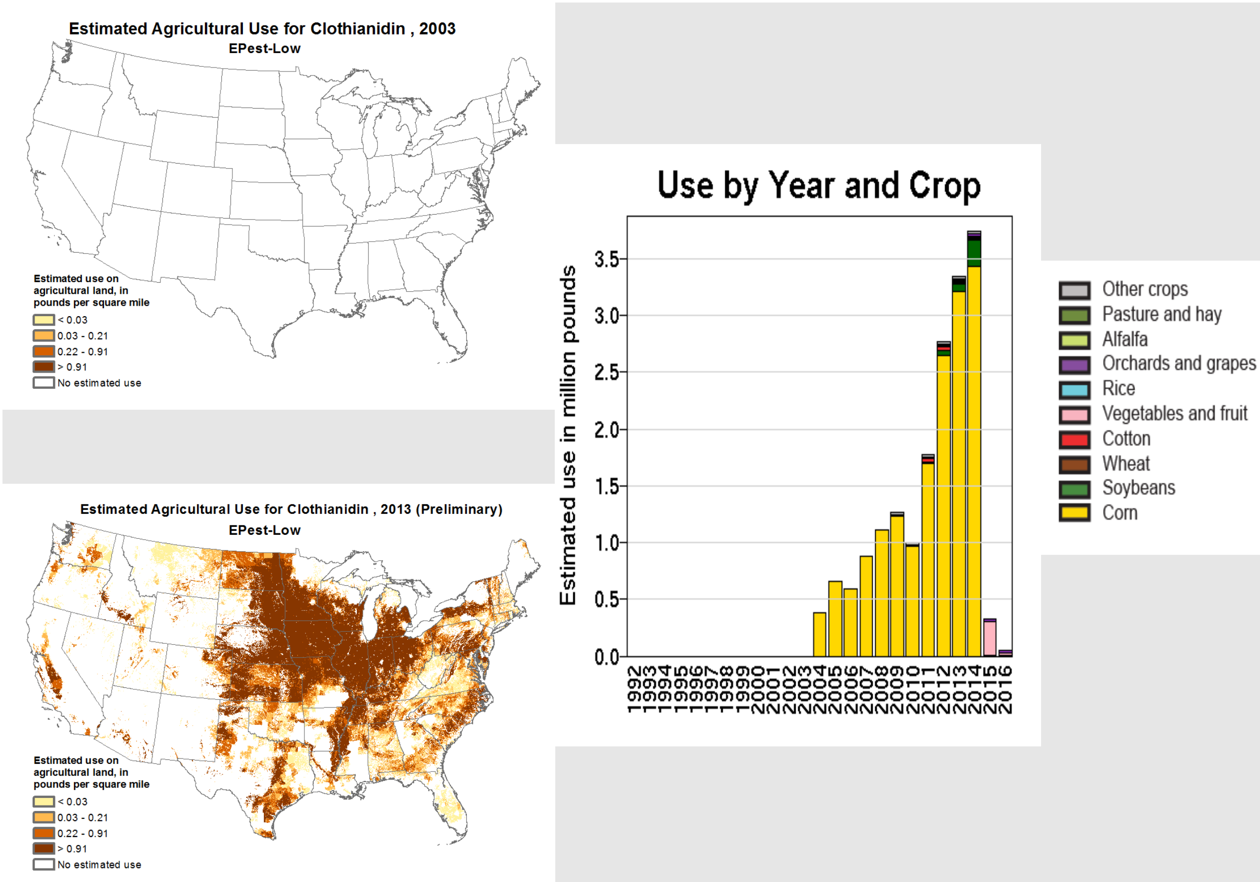

Figure 5.4. Estimated use of Clothianidin (a neonicotinoid) by location in 2003 (top) and 2013 (bottom) and by year (right). USGS National Pesticide Synthesis Project, accessed 2020; https://water.usgs.gov/nawqa/pnsp/usage/maps/

<u>Climate Change Effects</u>

Climate change can affect monarchs both directly and indirectly (Nail and Oberhauser 2015, entire) on both the overwintering and breeding grounds. Increasing storm frequency in the Mexican overwintering colonies can lead to catastrophic (up to 80%) mortality through the freezing temperatures that accompany these storms (Anderson and Brower 1996, p. 112; Brower et al. 2004, entire). Severe storms may become more frequent with precipitation predicted to increase during the winter when monarchs are present in Mexico (Oberhauser and Peterson 2003, p. 14067).

Monarchs need a very specific microclimate at their overwintering sites not just to avoid storm mortality, but also to avoid early lipid depletion (see Chapter 2, *Monarch Life History* section). Additionally, changing precipitation patterns and temperatures may influence the microclimate needed by overwintering monarchs (Williams and Brower 2015, p. 116). Current modeling of the monarch's fundamental niche predicts the loss of 38.6% to 69.8% of current suitable habitat within the Monarch Butterfly Biosphere Reserve (Zagorski 2016, p. 17). In western North America, climate change is predicted to cause a significant change in the distribution of overwintering monarchs in coastal California. Results from climatic niche modeling by Fisher et al. (2018, p. 10) suggest that climate change will result in an inland and upslope displacement of suitable overwintering conditions. The probability of occurrence of suitable overwintering conditions becomes roughly proportional to elevation.

Climate change impacts, particularly increasing temperatures, may impact monarch fecundity (Oberhauser 1997, pp. 168-169), mating success (Solensky and Oberhauser 2009, p.6), and survival during migration and while overwintering (Masters et al. 1988, entire; Alonso-Mejía et al. 1997, entire). Laboratory studies indicate optimal temperatures for monarch range from 27–42°C (Zalucki 1982, p. 243; York and Oberhauser 2002, p. 294; Zalucki and Rochester 2004, p. 225; Nail et al. 2015b, p. 101). Nail and colleagues (2015b) also found nighttime temperatures of 34°C during periods with daytime temperatures of 38°C resulted in lower survival, showing that respites from elevated temperatures are important in allowing monarchs to survive temperature stress (Nail et al. 2015b, p. 104). Temperatures consistently above 33°C to 35°C are unsuitable for monarchs and may account for their general absence from southern U.S. states after spring (Malcolm et al. 1987, p. 78; Zalucki and Rochester 1999, pp. 155- 157). High temperatures and drought conditions may be particularly impactful during the crucial spring migration (Chip Taylor, pers. comm. 2020).

In addition to the impact of climate change on overwintering monarchs directly, the Mexico overwintering sites are predicted to be less suitable for oyamel fir trees, the predominant monarch roosting tree. The overwintering sites are predicted to become increasingly warm throughout the year, potentially making 50% or more of the sites unsuitable for oyamel fir trees in 2030, and completely unsuitable for the oyamel fir trees by 2090 (Sáenz-Romero et al. 2012, p. 102; Ramírez et al. 2015, p. 167). Widespread drought is similarly likely to impact trees in the western overwintering areas both directly and indirectly due to increased susceptibility to pests (Paine and Millar 2002, p. 148).

A warming climate may influence breeding habitat by altering suitable locations for both monarchs (Batalden et al. 2007, pp 1369-1370) and their milkweed host plant (Lemoine 2015, entire). Saunders et al. (2019, p. 8612) suggested that nectar resources during migration may be reduced under climate conditions (decreased precipitation) projected for south-central Texas. Drought may also influence the amount and availability of nectar needed for migrating butterflies (Brower et al. 2015, entire; Stevens and Frey 2010, p. 740; Espeset et al. 2016; p. 826; see *Widespread Drought* section). The coastal non-migratory population may also be impacted by loss of habitat through rising sea levels due to climate change (Tampa Bay Climate Science Advisory Panel 2015, entire). While drought and increased temperatures may reduce monarch habitat in some areas, the climatically suitable niche for monarchs may increase, potentially increasing their summer breeding grounds if both monarchs and milkweed are able to adapt (Lemoine 2015, pp. 10-17).

Climate change may additionally impact monarchs in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of milkweed and nectar sources not aligning with monarch migration; Thogmartin et al. 2017b, p. 13) or range mismatch with associated species (e.g., changed environmental suitability of monarch natural enemies; McCoshum et al. 2016, p. 229-233). Furthermore, recent research suggests that carbon dioxide may impact the medicinal properties of some milkweed species, potentially leading to increased *OE* parasite virulence and decreased monarch tolerance of *OE* infections (Decker et al. 2018, p. 7; see Appendix 2 for more information on *OE*).

Conservation Efforts

While many factors have been implicated in the decline in monarch populations, the loss of milkweed and nectar resources (i.e., breeding and migratory habitat) has been targeted as the threat that can be most easily addressed through conservation efforts. Protection, restoration, enhancement and creation of habitat is a central aspect of recent monarch conservation strategies, thus highlighting the importance of restoring and enhancing milkweed and nectar resources (Oberhauser et al. 2017a, p. 6-8; Pleasants 2017, p. 43; Thogmartin et al. 2017b, p. 2-3; MAFWA 2018, p. 52; Pelton et al. 2019, p. 4-5, WAFWA 2019). Improved management at overwintering sites in California has also been targeted to improve the status of western North American monarch butterflies (Pelton et al. 2019, p. 4; WAFWA 2019).

Major conservation plans and efforts include the Mid-America Monarch Conservation Strategy developed by the Midwest Association of Fish and Wildlife Agencies (MAFWA), the Western Monarch Butterfly Conservation Plan developed by the Western Association of Fish and Wildlife Agencies (WAFWA), and the Nationwide Candidate Conservation Agreement for Monarch Butterfly on Energy and Transportation Lands (CCAA/CCA) developed by entities from the energy and transportation sectors and the Energy Resources Center at the University of Illinois – Chicago. The Mid-America Monarch Conservation Strategy established a goal of adding 1.3 billion stems of milkweed on the landscape by 2038 (MAFWA 2018). The 1.3 billion stem goal is an estimated goal for adding enough habitat to support 6 hectares of overwintering population for the eastern North American population, per Pleasants and Thogmartin et al. (2017; 2017c). Twenty states—including Arkansas, Illinois, Indiana, Iowa, Kansas, Kentucky, Maryland, Michigan, Minnesota, Missouri, Nebraska, New York, North Dakota, Ohio, Oklahoma, Pennsylvania, South Dakota, Texas, West Virginia, and Wisconsin—have agreed to participate in the effort to reach the 1.3 billion stem goal, which will require contributions from multiple sectors of society, including private land owners, agricultural and non-governmental organizations, rights-of-way organizations, and federal, state and local governments. The Western Monarch Butterfly Conservation Plan currently encompasses the states of Arizona, California, Idaho, Nevada, Oregon, Utah, and Washington, which comprise the core of the western monarch range (WAFWA 2019). The plan includes short-term goals of: 1) protecting and managing 50% of all currently known and active monarch overwintering sites, including 90% of the most important overwintering sites by 2029; and 2) providing a minimum of 50,000 additional acres of monarch-friendly habitat in California's Central Valley and adjacent foothills by 2029. It also includes overwintering and breeding habitat conservation strategies, education and outreach strategies, and research and monitoring needs. The monarch CCAA will also contribute to the goals of these plans by coordinating and providing guidance to businesses and organizations in the energy and transportation sectors seeking to implement conservation efforts for monarchs. In exchange for implementing voluntary conservation efforts and meeting specific requirements and criteria, those businesses and organizations enrolled in the CCAA will receive assurance from the USFWS that they will not have to implement additional conservation measures should the species be listed. The goal of the CCAA is enrollment of up to 26 million acres of land in the agreement, providing over 300 million additional stems of milkweed (Cardno, Inc. 2020).

There are many other conservation efforts implemented under agreements, such as the Farm Service Agency's Conservation Reserve Program and the Natural Resource Conservation Service's Environmental Quality Incentives Program, Wetland Reserves Program, and Conservation Stewardship Program, which will be critical for meeting MAFWA and WAFWA's stated goals. Additionally, multiple federal, state and local governments, non-governmental organizations, and private businesses and individuals have provided information about regional and local monarch conservation plans and efforts. Although not associated with any formal plans or agreements, we have also obtained information on thousands of small and backyard pollinator gardens through organizations such as Monarch Watch.

Several land managers who oversee overwintering sites in California have developed and implemented grove management strategies (e.g., Ardenwood Historical Farm, Lighthouse Field) or have added monarch groves in their general management plans (e.g., Vandenberg Air Force Base). Others are in the process of developing grove management plans for which funding has already been established (e.g., Ellwood Mesa Complex). At this time, grove management plans have been implemented by at least three overwintering sites and are currently being developed for at least seven more. An additional 37 overwintering sites are on public land that has a general management plan that specifically includes protections for monarch groves (IELP and Xerces Society 2012, entire). Management and restoration of these sites may include activities such as replacing dead trees, modifying canopy structure, planting fall- and winter-blooming shrubs as nectar sources, and addressing monarch predation issues (Jepsen et al. 2017, entire).

The USFWS developed the Monarch Conservation Database (MCD) to capture information about monarch conservation plans and efforts to inform the listing decision. As of June 1, 2020, there are 48,812 complete monarch conservation effort records in the MCD that have a status of completed, implemented, or planned since 2014, and 113 monarch conservation plans. These efforts constitute a total of 5,635,992 acres of land area in the continental United States and Hawaii (5,534,451 acres and 97,949 acres in the eastern and western populations, respectively) enhanced or created for monarchs, with the most common conservation effort being direct planting of milkweed and other nectar resources [note that these values include all completed, implemented, and not yet completed efforts; completed and implemented efforts to-date total 4,542,323 acres nationally].

Worldwide – Current Condition

Today, there are 30 extant populations and 1 presumed extant (Table 5.3, 5.4). The current health of these populations, however, is unknown, as there is insufficient information available (with the exception of the eastern and western North American populations, described above; Table 5.5).

Table 5.3. The current status (extant; unknown or presumed extant; or extirpated) of ACUs, populations, and countries/islands.

Status	# ACUs	# Pops	# Countries/ Islands	Definition
Extant	8	27	69	Observed since 2000
Unknown or Presumed Extant	0	4	21	Not observed since 2000, but lacking multi-year survey efforts
Extirpated	0	0	0	No observations despite multi-year survey efforts

Table 5.4. Current status of monarchs in 90 known countries, islands, or island groups occurrences and 31 populations worldwide. Status = presumed extant (P), known extant (E).

Population	Country/Island	Status	Population	Country/Island	Status
Australia (E)	Australia	E	Guam & Commonwealth of the Northern Mariana Islands [CNMI] (E)	CNMI	E
Caribbean (E)	Anguilla	E		Guam	E
	Antigua and Barbuda	P	Hawaii (E)	Hawaii	E
	Bahamas	E	Iberian Peninsula (E)	Azores	P
	Barbados	E		Canary Islands	E
	Bermuda	E		Gibraltar	E
	Bonaire	E		Madeira	E
	British Virgin Islands	P		Morocco	E
	Cayman Islands	P		Portugal	E
	Cuba	E		Spain	E
	Dominica	E	Johnston Atoll (E)	Johnston Atoll	E
	Dominican Republic	E	Kiribati (E)	Kiribati	E
	Grenada	E	Marquesas Islands (E)	Marquesas Islands	E
	Guadeloupe	E	Marshall Islands (E)	Marshall Islands	E
	Haiti	E	Mascarene Islands (E)	Mauritius	P
	Jamaica	E		Réunion	E
	Martinique	E	Micronesia (E)	Federated States of Micronesia	E
	Montserrat	P	Nauru (E)	Nauru	E
	Puerto Rico	E	New Zealand (E)	New Zealand	E
	Saba	E	Norfolk Island (E)	Norfolk Island	E
	Saint Barthélemy	P	Palau (E)	Palau	E
	Saint Kitts and Nevis	P	Papua New Guinea (E)	Papua New Guinea	E
	Saint Lucia	P	Philippines (P)	Philippines	P

Population	Country/Island	Status	Population	Country/Island	Status
	Saint Martin	E	Samoa (E)	American Samoa	P
	Saint Vincent & Grenadines	P		Samoa	E
	Sint Eustatius	E	South America and Aruba (E)	Aruba	E
	Sint Maarten	E		Colombia	E
	Turks and Caicos Islands	P		Curaçao	E
	US Virgin Islands	E		Ecuador	E
Central America (E)	Belize	E		French Guiana	P
	Costa Rica	E		Guyana	E
	El Salvador	E		Peru	E
	Guatemala	E		Suriname	P
	Honduras	E		Trinidad and Tobago	E
	Nicaragua	E		Venezuela	E
	Panama	E	South Florida (E)	South Florida*	E
Cook Islands (E)	Cook Islands	E	South Pacific (E)	Fiji	E
E. North America (E)	Canada (also part of the W. N. America population)	E		New Caledonia	E
	Mexico (also part of W. N. America and Central American populations)	E		Society Islands	E
	Saint Pierre & Miquelon	P		Solomon Islands	E
	E. United States	E		Vanuatu	E
Austral Islands (E)	Austral Islands	E	Tokelau (P)	Tokelau	P
Greater Indonesia (P)	Brunei	P	Tonga (E)	Tonga	E
	Indonesia	P	Tuvalu (E)	Tuvalu	E
	Malaysia	P	Wallis & Futuna (P)	Wallis and Futuna	P
	Timor-Leste	P	W. North America (E)	W. United States*	E

*Country that is listed multiple times, but not counted again (note that countries may be counted multiple times if they have distant islands; e.g., Hawaii is counted separately from the contiguous United States.)

Table 5.5. Population health: current status, past trend in population size (N), current status of milkweed & nectar resources, current status of insecticides, and overwintering habitat

ACU	Population	Status	Trend in N	MW/Nectar	Insecticides	OW Habitat	Overall Condition
Australia, New Zealand, and Indo-Pacific Islands	Australia	Extant	Unknown	Unknown	Unknown	Unknown	Unknown
	Cook Island	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	French Polynesia	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Greater Indonesia	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Guam and CNMI	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Johnston Atoll	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Kiribati	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Marquesas Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Marshall Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Mascarene Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Micronesia	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Nauru	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	New Zealand	Extant	Unknown	Unknown	Unknown	Unknown	Unknown
	Norfolk Island	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Palau	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Papua New Guinea	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Philippines	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Samoa	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	South Pacific Islands	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Tokelau	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
	Tonga	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Tuvalu	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Wallis & Futuna	Unknown	Unknown	Unknown	Unknown	N/A	Unknown
Central America, Caribbean	Caribbean	Extant	Unknown	Unknown	Unknown	N/A	Unknown
	Central America	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Southern Florida	Florida	Extant	Unknown	Unknown	Unknown	N/A	Unknown

ACU	Population	Status	Trend in N	MW/Nectar	Insecticides	OW Habitat	Overall Condition
Hawaii	Hawaii	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Iberian Peninsula	Iberian Peninsula	Extant	Unknown	Unknown	Unknown	N/A	Unknown
South America, Aruba	South America and Aruba	Extant	Unknown	Unknown	Unknown	N/A	Unknown
Eastern North America	Eastern North America		Extant	*See Eastern North American Population section below*			
Western North America	Western North America		Extant	*See Western North American Population section below*			

Worldwide – Current Influences

There is little to no information on the status and health for most of these populations, as well as information regarding positive or negative influences acting upon these populations. Below we discuss what little information is known or can be assumed.

There is limited information on predation, parasitism, and disease outside of eastern and western North American populations. Parasitism rates from Tachinid flies have been documented in Australia, Hawaii, throughout Central America, and Brazil. In Australia, the rates fluctuate throughout the year, ranging from very low to up to 100% of sampled monarchs in February (Smithers 1973, p. 38). Another parasitoid, the wasp *Pteromalus puparum*, is also known to attack monarch pupae in other locations (Ramsay 1964, p. 15). The protozoan parasite, *OE*, infects monarchs throughout Australia, Central and South America (Altizer et al. 2000, p. 135), and Hawaii (Pierce et al. 2014b, p. 1). Thus, given this limited information, we are unable to ascertain to what extent predation, parasitism, and disease impact worldwide monarch populations. Similarly, while data suggest global use of insecticides is increasing, we are unable to estimate the degree of overlap with monarch populations and thus derive a credible projection of impact on the worldwide monarch populations.

Chapter 6: Results –Future Influences and Catastrophic Events

This chapter describes our projections for the future states of the influences. To capture the uncertainty in our future projections, we identified both plausible optimistic and pessimistic changes for each influence. These optimistic and pessimistic states for each influence were then combined to create composite plausible "best case" and "worst case" scenarios. Additionally, we describe the events that are likely to be catastrophic should they occur.

North American Populations – Future Scenarios

To assess the future condition of monarch populations, we organized the key factors driving monarch population dynamics into 5 categories: 1) milkweed availability, 2) nectar availability, 3) migration nectar availability, 4) climate change effects, and 5) insecticide exposure. We then forecasted how each of these five influences is expected to change (i.e., its expected future state condition). We described the expected changes as the percent change from current state conditions (Figure 6.1 & 6.2). Lastly, we combined the most optimistic and pessimistic expected state conditions of each influence to form composite plausible best and plausible worst scenarios, respectively. The range of plausible future state conditions for each influence is described below and summarized in Table 6.1 (eastern population) and Table 6.2 (western population).

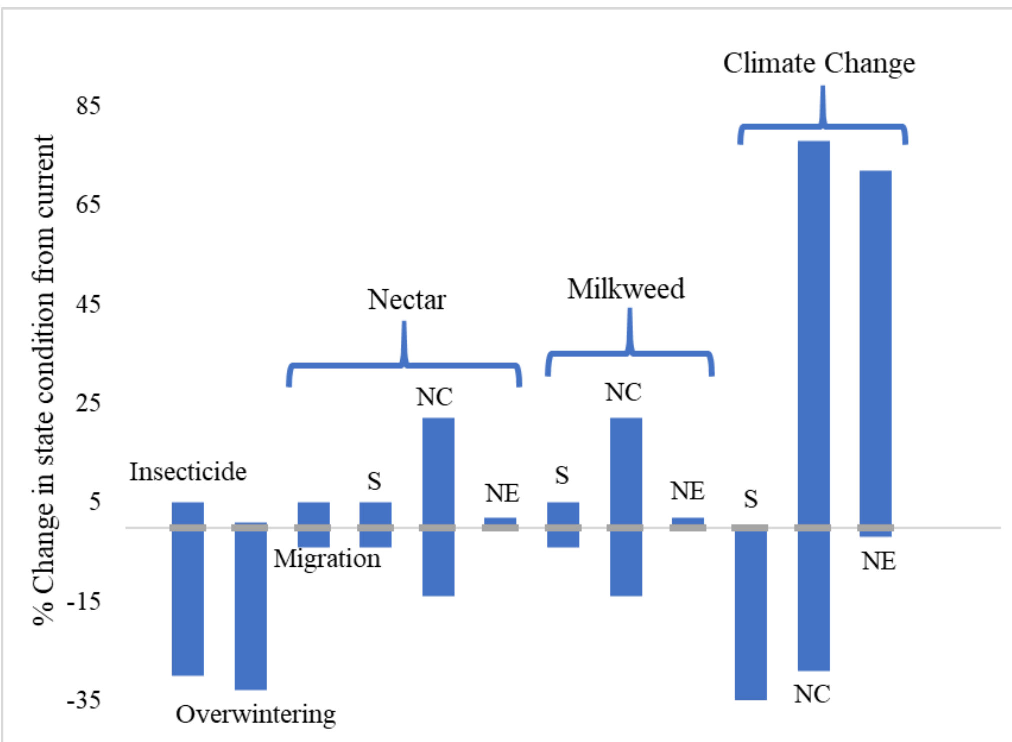

Figure 6.1. Range of forecasted % change from current state conditions for eastern population influences. Bars above and below the x-axis represent positive and negative changes, respectively, relative to monarch numbers. S, NC, and NE represent the Southern, Northcentral, and Northeastern subregions of the breeding range, respectively.

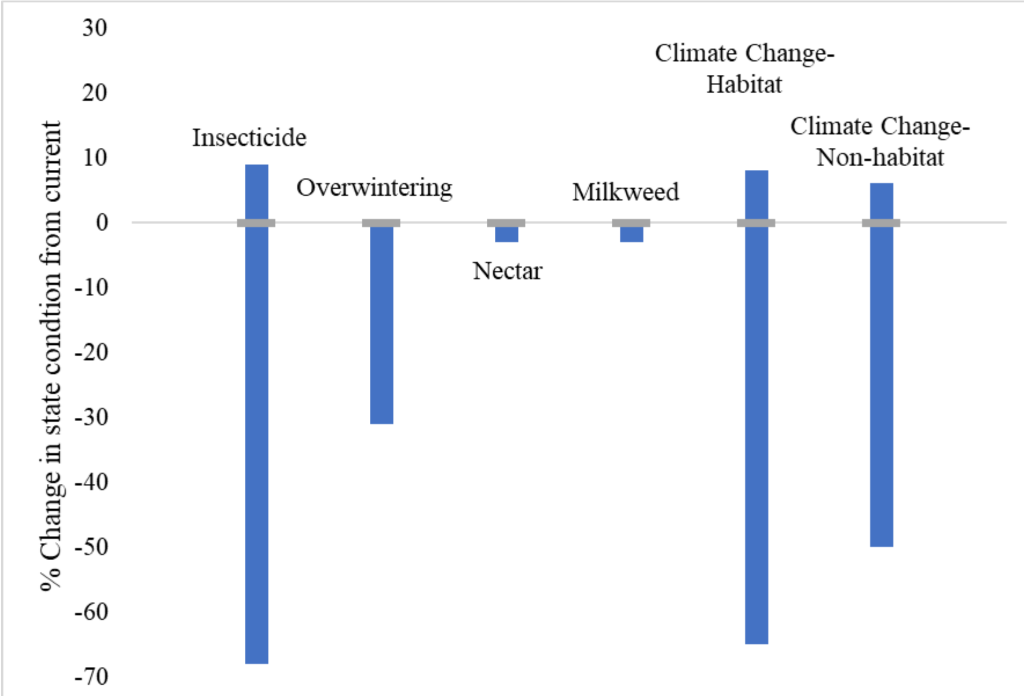

Figure 6.2. Range of forecasted % change from current state conditions for western population influences. Bars above and below the x-axis represent positive and negative changes, respectively, relative to monarch numbers.

Table 6.1. Description of the future state conditions for the influences for the eastern population. Time = the time period over which the change will occur. % Change = estimated % change in influence. NC = northcentral, NE = northeast, S = south.

Influence	**Time/% Change**	**Description**
Milkweed	18 years	
Best	NC: 22% increase NE: 3% increase S: 5% increase	Successful implementation of the Mid-America Monarch Conservation Strategy and other planned efforts, alongside gains of CRP habitat (22% increase in CRP acreage relative to 2018 levels), and a 2% milkweed stem gain driven by future land cover change, results in widespread habitat gains, primarily occurring in the North Core geography.
Worst	NC: 11% increase NE: 1% loss S: 6% loss	Successful implementation of the Mid-America Monarch Conservation Strategy and other planned efforts, occurring alongside losses of CRP habitat (35% decline compared to 2018 CRP levels), with no impact from future land cover change, results in modest habitat gains overall but variable by geography.
Nectar Resources	Same as Milkweed	*Same as Milkweed conditions*
Migration Nectar	18 years	
Best	S: 5% increase	*(Same as Milkweed "Best")*
Worst	S: 6% loss	*(Same as for Milkweed "Worst")*

Influence	Time/% Change	Description
Overwintering habitat	25 years	
Best	1% gain	Natural forest regeneration caused by reduced illegal logging and grazing pressures is projected to result in very slight gains of habitat over time.
Worst	33% loss	Losses of trees due to large-scale illegal logging and climatic factors are projected to continue at rates that have been observed in the recent past. This assumes that funding and programs implemented very recently are not sustained.
Climate change - Habitat	60 years	
Best	NC: 78% increase NE: 72% increase S: no change	Climate change drives increased habitat suitability and northward range expansion, up to a boundary of approximately 50°N latitude, resulting in widespread habitat increases throughout the eastern geography, particularly in Canada.
Worst	NC: 29% loss NE: 2% loss S: 83% loss	Climate change reduces overall habitat suitability across the current range; monarchs and milkweed do not effectively shift their range northward to track changing climatic conditions, resulting in habitat losses occur across the range, most notably in the southern geography.
Insecticides	25 years	
Best	5% decrease	Increasing attention for monarch conservation via MAFWA, CCAA, and MP3 plans, as well as increasing opportunities for VRT & newer equipment with the shift to larger farming operations.
Worst	30% increase	Increasing demand for food production leading to increases pest management; increasing trend in crop and disease-vector pests leading to aggressive insecticide response to prevent crop damage (e.g., soybean aphid) and disease outbreaks (e.g. Zika, West Nile).

Table 6.2. Description of the future state conditions for the influences on the western population. Time = the time period over which the change will occur. % Change = the % change estimated.

Influence	Time/% Change	Description
Milkweed	50 years	
Best	2% loss	Incorporates a low human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Worst	3% loss	Incorporates a high human growth scenario and conservation efforts implanted via the WAFWA plan and nonprofit groups.
Nectar Resources		*Same as Milkweed conditions*
Overwintering habitat	50 years	

Influence	Time/% Change	Description
Best	18% loss	Projected losses of overwintering habitat are decreased from of those losses observed caused by urban development between 1990-1998 due to increased coastal development regulations and recent decreasing population growth rate in California. Conservation efforts implanted via the WAFWA plan are also included.
Worst	31% loss	Projected losses of overwintering habitat are consistent with those losses observed caused by urban development between 1990-1998 due to continued increasing population in California. Conservation efforts implanted via the WAFWA plan are also included.
Climate change – habitat	20 years	
Best	8% decrease	Increases in suitable climate niche due to projected increases temperatures.
Worst	65% increase	Losses of breeding and overwintering habitat due to projected increases in drought intensities & frequencies; the combined effect of dry spring conditions and warmer summer temperatures.
Climate change – non-habitat	20 years	
Best	6% decrease	Projected increases in *minimum* temperatures may expand the amount of time available for western monarch reproduction, thereby allowing for more generations per year to be produced and boosting monarch numbers.
Worst	50% increase	Reductions in reproduction and survival due to projected increases maximum daily temperatures, and hence, the number of days where temperatures exceed critical monarch thresholds.
Insecticides	20 years	
Best	9% decrease	Increasing attention for monarch conservation via WAFWA, CCAA, and MP-3 plans, as well as increased awareness of pollinator declines could lead to reduced and more targeted insecticide use.
Worst	68% increase	Increasing demand for food and projected land conversion from rangeland to agriculture; significant overlap of agricultural lands and the areas of most important to monarch production--CA Central Valley; and lack of standardize, broad-scale efforts and difficulty regulating use needed to reduce exposure

Availability, Distribution, and Quality of Milkweed

Eastern Population

Future scenarios for milkweed and nectar resources for the eastern population include a combination of 1) projected conservation effort, 2) projected changes in CRP acreage, and 3) other habitat change driven by projected land cover change. Scenarios are described in terms of percent change in "habitat" as indicated by milkweed stem estimates (with habitat assumed to consist of both milkweed and nectar resources, effectively co-occurring in a 1:1 ratio on average at broad scale), where percent change is reported relative to 2020 milkweed estimates, respectively for each subregion (Northcentral, Northeast, and South). "Baseline" (2020) habitat estimates were derived from the USGS "seamless" land cover spatial data (Rohweder and Thogmartin 2016; see Appendix 2 for additional methodological details), also including all completed and implemented efforts reported since 2014 via the national MCD.

For the eastern population, our future milkweed scenarios incorporated all not yet implemented (i.e., future) formalized conservation efforts reported to the MCD. For each subregion, the same level of formalized future conservation effort was projected for both the upper and lower bounds. For the Northcentral subregion, projected future formalized conservation effort associated with the Mid-America Monarch Conservation Strategy results in an additional 1.3 billion milkweed stems. We assumed conservation efforts occurring since 2014 effectively contribute to that goal. For our upper bound, we assumed achieving that goal would also include projected gains in CRP, meaning that any increase in CRP acres (in this case a 22% gain relative to 2018 levels; 156,485,213 stems) are not additive (beyond the 1.3 billion stem target) but rather are a contribution toward the overall target in the Northcentral subregion. For the lower bound in the Northcentral subregion, we assumed a similar level of effort would occur compared to the upper bound but with 35% less CRP contributions. Lacking any comparable overarching multi-state plan for much of the South and Northeast, we assumed CRP changes would be additive to future formalized conservation efforts in those subregions. For the conservation effort component of the eastern population future scenario, relative to 2020 levels, we projected an estimated 17% increase for the Northcentral, a 0.28% increase for the South, and a 0.03% increase in in milkweed/nectar in the Northeast subregions.

For CRP, we relied on USDA agricultural projections (USDA 2020), along with national CRP trend data and expert input from USDA-Farm Service Agency (Skip Hyberg, retired Senior Economist; personal communications). US Farm Bill programs are inherently difficult to predict, occurring at roughly 5-year legislative cycles and reflecting national and global economic and policy drivers that influence commodity prices and agricultural land values. We used current USDA projections (USDA 2020) for CRP to inform our upper bound, assuming that CRP increase under their stated assumptions could occur linearly over the next 18 years. Relative to 2018 CRP acreage, our upper bound scenario projected a 22% increase in CRP habitat and our lower bound scenario projected a 35% decline in CRP acres, respectively for each subregion. The lower bound CRP scenario was based on 10-year national CRP acreage declines (2008-2018). For purposes of milkweed stem estimates, future CRP losses/gains were assumed to change to/from cropland land cover.

For broader land cover change, we used the USGS FORE-SCE (Sohl et al. 2018) spatial data projections, which are informed by International Panel on Climate Change Special Report on Emissions Scenarios (IPCC 2000), to evaluate predicted milkweed stem change, respective to each subregion. Our scenarios account for land cover change occurring independent of conservation effort and CRP changes. Milkweed stem estimates, by land cover type, were based on a modified interpretation of Thogmartin et al. (2017c) where a subset of land cover types were lumped or split when necessary to align with the land cover classification scheme available in the FORE-SCE spatial data. We assumed land cover change would occur roughly linearly; therefore, we annualized the projected rate of change relative to the 2050 model output provided by FORE-SCE. For the land cover change component of our future scenarios, we estimated a 4% increase in milkweed stems in the Northcentral subregion, a 5% increase in the Northeast subregion, and a 4% increase in the South subregion over 40 years for the upper bound (primarily driven by urbanization trends). For the lower bound, we assume no habitat change due to projected land cover change.

When conservation effort, CRP, and land cover were considered holistically, overall projected changes in milkweed and nectar habitat range from a 11-22% increase in the Northcentral subregion, a 1% decrease to 3% increase in the Northeast subregion, and a 6% decrease to 5% increase in the South subregion (Table 6.1).

Western Population

The western population future state conditions are predicated upon projections of 1) human population growth rate in California and corresponding changes in landuse/cover and 2) conservation efforts throughout the West. California's Central Valley is an important production area for western monarchs (Crone et al. 2019, p. 10) and important migration pathway. Thus, the availability of milkweed or nectar resources in this area greatly influences the western population dynamics. Hence, we primarily relied on trends in California—and the Central Valley, in particular—to project the future state condition of milkweed and nectar availability. Loss of rangelands (an important land cover for monarchs) represented the largest land cover change in California's Central Valley, with a loss of approximately 1,054 km^2 (~260,450 acres) between 1980 and 2000 (Sleeter 2016). To project future trends, we used the results from Sleeter et al. (2017) analyses. They projected future land use change in California under three human population growth projections, and we chose the low and high human population growth scenarios to bound the range of plausible human population growth and the associated land use projections to estimate the change in monarch breeding habitat. The human growth projections were developed by the California Department of Finance (2019), which monitors human population growth trends at state and county scales. We believe that the methods used to develop these projections were scientifically rigorous, and thus, the scenarios represent the best available data and realistic projections of human population growth in California. In the low human population growth scenario, by the year 2070 approximately 2,600 km^2 will be converted from grassland or shrubland habitat to land use types that do not support monarchs. This represents a loss of 1.7% from the current amount of grassland and shrubland habitat currently available in California. In the high human population growth scenario, by the year 2070 approximately 5,300 km^2 will be converted from grassland or shrubland habitat to land use types that do not support monarchs. This represents a loss of 3.4% from the current amount of grassland and shrubland habitat currently available in California.

To forecast plausible future conservation efforts, we relied upon the WAFWA plan (2019, p. 39) and ongoing projects by nonprofit groups. Under the WAFWA plan, a minimum target of 202 km^2 of breeding habitat and adjacent foothills will be restored by 2029. The key drivers in realizing the plan's restoration goals are adequate funding and partner willingness. These issues are discussed within the plan and we agree with the rationale given for why these targets are plausible (WAFWA 2019, pp. 86-87). We also believe that additional conservation will be achieved by nonprofit groups and use information from the Xerces Society as proxy for estimating the quantity of habitat restored to project habitat restoration into the future. The Xerces Society has received funding to restore 2.65 km^2 of breeding habitat over the next five years and we use this value to project restoration by nonprofit groups over the next 50 years (an estimated total of 26.5 km^2). It is reasonable to expect similar levels of effort and funding for nonprofit groups to continue because supporting organizations such as the Monarch Joint Venture have shown that they are committed to furthering the conservation of the species in the West by funding these projects into the future. Thus, under both scenarios, we assumed 228 km^2 of habitat will be restored, yielding 2,384 km^2 (-2%) and 5,116 km^2 (-3%) for the best (low population growth) and worst (high population growth) case scenarios, respectively (Table 6.2).

Availability, Distribution, and Quality of Breeding Range Nectar Resources

Milkweed stem density is assumed to be a reasonable proxy for the availability, abundance, and phenological diversity of nectar resources. Monarch conservation best management practices generally tend to focus on producing more milkweed alongside diversified vegetation composition and structure, leading to more abundant and more diverse nectar resources that may be available for extended periods of the growing season (additionally, milkweed itself serves as a nectar source throughout a portion of the year). The ratio of milkweed:nectar outputs is largely unknown, is difficult to quantify, and likely varies by land cover, sector, conservation practice, geography, and climatic conditions. While some efforts may produce disproportionate changes in milkweed or nectar resources, 1:1 the relationship between nectar and milkweed is generally assumed to be correlated on average over broad spatial scales. As the mechanisms affecting the availability of nectar and milkweed are assumed to be the same, our future projections for them are proportionally the same as well.

Availability, Distribution, and Quality of Migration Nectar Resources

See the previous section "Nectar Resource Availability" for our rationale on why our southern milkweed scenario is a suitable proxy for nectar. As the mechanisms affecting the availability of nectar and milkweed are generally assumed to be the same, our future projections for them are proportionally the same as well.

Availability and Quality of Overwintering Habitat

Eastern Population

The future projections of the availability of overwintering habitat are largely predicated upon the analyses within Honey-Rosés et al. (2018), Vidal et al. (2014), and Flores-Martínez et al. (2019), the key findings of which are described below.

Under the best case scenario (1% increase, Table 6.1), we assumed that: 1) forest regeneration within the Monarch Butterfly Biosphere Reserve continues at the current rate (0.04% annually), and 2) the negative effects from illegal logging and climate change will lessen over time. Honey-Rosés and colleagues (2018) estimated 0.04% gains in reforestation annually due to natural forest regrowth and concerted replanting efforts. The current regeneration rate is driven largely by reduced logging and grazing pressures, a trend we can plausibly foresee continuing over two or more decades. We also assumed that this rate captures any loss of overwintering habitat (and regeneration outweighs these negative stressors, assuming that illegal logging will continue to decline as well and both oyamel fir trees and monarchs will adapt to the projected environmental conditions under climate change; see Sáenz-Romero et al. 2012).

Under the worst case scenario (33% decrease, Table 6.1), we assumed that: 1) illegal logging returns to rates observed prior to involvement and funding by stakeholders, and 2) the recent loss of habitat due to climatic factors continues. Vidal and colleagues (2014) observed a high percentage of loss due to illegal logging between 2001 and 2012 (2,179 hectares of core zone were impacted due to illegal logging over 11 years; Vidal et al. 2014). Flores-Martínez and colleagues (2019) observed the highest recently recorded rate of habitat loss due to climatic factors between 2012 and 2018 (125 hectares impacted due to climatic factors over 6 years) and we can foresee this trend continuing over two or more decades. Combined, these factors result in an annual loss of approximately 219 ha of overwintering habitat per year (5,473 ha by the year 2045). We assumed that the recent reductions in illegal logging (Flores-Martínez et al. 2019) do not continue or are no longer effective going forward, and thus, rates of illegal logging revert to levels previously observed (since 2000). This is plausible because many of these improvements rely on funding and programs offered by the government and outside entities; if they can no longer be funded, then both large- and small-scale logging operations are expected to resume (Flores-Martínez et al. 2019, p. 7).

Western Population

The future projections of the availability of overwintering habitat are predicated upon: 1) forecasts of urban development and associated monarch habitat loss along coastal California and 2) conservation efforts under full implementation of the WAFWA plan. There is a strong interest by the State and conservation groups to protect and manage key monarch overwintering sites, and thus, under both scenarios, we assume that the actions proposed by WAFWA and conservation groups will be fully implemented. Under the WAFWA plan, 50% of all known overwintering sites will be protected and managed for monarchs by 2029 (WAFWA 2019, p. 35). It is reasonable to expect the WAFWA plan to be fully implemented because the plan outlines the steps required and identifies the key players (WAFWA 2019, pp. 87-88) and the State of California continues to further legislation designed to support implementation of the plan (State of California 2018).

Although the current rate of monarch overwintering habitat loss is unknown, rate of loss from 1990 to 1998 (due primarily to urban development) was 12% (Griffths and Villablanca 2015, entire). The threat of urban development in coastal California remains. Given continued increases in the human population (California Department of Finance 2019), we expect loss of overwintering sites due to urban development to continue. However, we can foresee the rate of habitat loss decreasing because California's population growth rate has been below 1.0 percent since 2005, with the 2019 growth rate being the lowest since 1900 (California Department of Finance 2019). Given this, we can foresee a reduction from the rate of overwintering habitat reported by Griffiths and Villablanca (2015) to 6% loss every 9 years, which is half of the rate observed in the 1990s. Under this foreseeable best case scenario, considering protection and maintenance of 50% of the overwintering sites starting in 2029 and a decreased rate of habitat loss at the remaining 50% of sites, we estimate a total loss of 18% of habitat over the next 50 years (Table 6.2). Under the foreseeable worst case, considering protection and maintenance of 50% of the overwintering sites starting in 2029 and continued loss of habitat at the observed rate (Griffiths and Villablanca 2015), we estimate a total loss of 31% over the next 50 years (Table 6.2).

Climate Change Effects

Eastern Population

Our future scenarios for habitat related climate changes were derived primarily from the model results of Lemoine (2015, entire). For the best case climate change scenario, suitable habitat increases by 78% in the Northcentral subregion, increases by 72% in the northeast subregion, and has no gain or loss in the southern subregion. This was based on the slightly modified monarch and milkweed ecological niches as modeled by Lemoine (2015), using the moderate B2 emission scenario. While Lemoine (2015) found an overall increase in suitable breeding habitat for eastern monarchs, we assume that this increase will ultimately be constrained by the current northern extent of the monarch's range (approximately 50°N). This is reasonable to expect because while there could be some northward expansion in suitable habitat driven by climate change, there are simultaneous factors that limit the degree to which milkweed and monarchs will be able to fully realize a northward range expansion (particularly in terms of population-level outcomes). First, northern expansion of milkweed is expected to lag behind changing climatic conditions, both because of the time it takes the species to colonize large, new areas and because of other potential differences in suitable habitat (e.g., different soil types or competing vegetation). Second, monarchs are mobile, but northward expansion might also be limited for physiological reasons (e.g., lack of directional flight after certain dates, insufficient energetic resources, etc.; Taylor, pers. comm. 2020). Third, even if monarchs and milkweed were able to effectively colonize beyond their current northern limit (~50°N), these monarchs would not be able to successfully migrate such a long distance to Mexico, as evidenced by the limited tag returns from similarly far away areas in the north and northeast (Taylor, pers. comm. 2020). Furthermore, those monarchs that did successfully make the extended journey to the overwintering grounds might subsequently have lowered fecundity due to the increased energetic constraints relative to monarchs that migrated from more optimal core breeding grounds. Thus, we assumed future range expansion will be limited to 50°N latitude.

Under the same moderate emissions scenario, Lemoine (2015) estimates that the southern subregion of the current eastern population breeding range will have a loss of the southernmost portion of the range but backfilling in the more northern part of the southern subregion. Overall, there was more backfilling than loss of southern habitat (for a potential 34% increase); however, this does not account for the importance of the southern portion of the breeding range, particularly for migratory demographic connectivity (Flockhart et al. 2015, p. 5). Thus, for this likely best case scenario, we took a moderate approach and assumed neither an increase nor decrease in the suitable habitat in the southern subregion (Table 6.1).

For the worst case scenario, we used Lemoine's more severe modeled climate change scenario (A2 emissions scenario), but again we constrained monarch expansion to 50°N latitude. Under this climate change scenario, habitat losses will occur in all 3 subregions: 29% loss in the Northcentral subregion, 2% loss in the northeast sub-region, and 83% loss in the southern subregion (Table 6.1).

Western Population

We relied upon expert predictions and other information to quantify the change in impacts from climate change to western monarchs over the next 20 years. We elicited the current and predicted future influence of non-habitat and habitat mediated effects of climate change on monarch numbers (Voorhies et al., 2019, Suppl. 2).

<u>Non-habitat mediated climate change effects</u>

The median (across experts) predicted percent change in influence from the current condition ranged from a 6% decrease in impact to a 50% increase in impact over the next 20 years (Table 6.2). The key underlying premise for the experts' predicted lessening impact from climate change effects is predicated upon recent findings suggesting increases in temperatures could improve reproduction. Svancara et al. (2019), for example, found that the projected increases in minimum temperatures in Idaho will expand the amount of time available for western monarch reproduction (by a half to a full month), thereby allowing for more generations per year to be produced and boosting monarch numbers.

The key underlying premises for predicting increasing impact from climate change include increasing maximum daily temperatures and severe precipitation events. Increasing temperatures–extremes and nighttime temperatures–can hinder reproduction and lead to increased mortality when temperatures exceed critical thresholds (38ºC and 42ºC, respectively; see the climate change section under Influences above). Projected changes in climate show continued and accelerated increases in temperature across the western U.S. through the twenty-first century (Sillmann et al. 2013, entire). In California, for example, statewide warming of 2-4°C (RCP 4.5) to 4-7°C (RCP 8.5) is projected by the end of the century (Pierce et al. 2018, pp. iv, 17-18); extreme temperature events are predicted to increase as well (Pierce et al. 2018, p. 22-28; see also Climate Change discussion under the Current Influences section above).

The experts also forecasted increased mortality from increasing intensity of strong precipitation events at overwintering sites. Unlike the temperature projections, regional changes in precipitation are more variable among global climate models (Kharin et al. 2013, entire). However, climate models generally project an increase in extreme precipitation events in California, including the overwintering coastal areas for monarch (Pierce et al. 2018, p. 26; Swain et al. 2018, entire).

We believe the experts' projections are supportable given the climate change projections available and the knowledge on monarch critical temperature thresholds. Under the best case scenario, the experts assumed that with projected increases in temperature, the number of generations and thus number of monarchs will increase and the number of days where the maximum temperatures exceeds critical thresholds will not increase. Under the worst case scenario, the experts forecasted increased mortality and reductions in reproduction given projected increases in maximum temperatures and the intensity of "most intense" precipitation events at overwintering sites.

Habitat-mediated climate change effects

The median (across experts) predicted percent change in influence from current condition ranged from an 8% decrease to a 65% increase over the next 20 years (Table 6.2). The experts' predictions are predicated upon anticipated changes in: 1) drought frequencies and severities, 2) the suitability of monarch overwintering habitats along coastal California, and 3) the suitability of monarch breeding habitat throughout the West.

The experts' prediction for a reduction in impact is predicated upon recent analyses that show monarch distribution being largely a function of milkweed occurrence (Dilts et al. 2019, p. 6; Lemoine 2015, p. 11; Svancara et al. 2019, p. 14), and with increasing temperatures, the area of suitable climate niche may expand (Svancara et al. 2019, p. 15).

The experts' prediction of an increasing impact is predicated on increasing drought intensities and or frequencies, which will reduce milkweed and nectar plant availability throughout the West. Stevens and Frey (2010, entire) found moisture regime acts as a strong bottom-up driver of monarch abundance patterns via resource availability in the West. Drought indices for California, Idaho, Nevada, and Oregon (but not Arizona, Utah, or Washington) were each significantly associated with monarch wintering abundance patterns, with California exhibiting the strongest relationship. Variation in moisture availability within a block of three contiguous central California climate divisions (Sacramento Drainage, San Joaquin Drainage, and Southeast Desert Basin) significantly predicted inter-annual abundance of migrant generation monarchs. Similarly, Espeset et al. (2016, p. 824, 826) found a positive effect of precipitation and western monarch numbers at focal sites. These findings suggest that precipitation may be a limiting factor and thus increased drought—frequency or intensity—will negatively affect western monarchs.

Even though annual precipitation changes due to climate change are predicted to be modest, year-to-year variability is predicted to increase due to the wetter winter conditions and drier spring conditions in California (Pierce et al. 2018, p. 27). The overall result is an increase in the frequency of dry years due to fewer wet days, but more precipitation on wet days (Pierce et al. 2018, p. 27). In addition, maximum July temperatures are expected to increase and heat waves may span longer durations (Pierce et al. 2013, entire). This could lead to increased evapotranspiration (Diffenbaugh et al. 2015, p. 3994) and a greater likelihood of monarch habitats drying, both inland breeding and coastal overwintering (Pierce et al. 2018, p. 25). The combined effect of dry spring conditions and warmer summer temperatures would reduce the amount of milkweed and nectar resources across the landscape available for nectaring and egg-laying, particularly in the early part of the year when western monarchs are migrating away from the overwintering sites to produce the first generation. These overwintering monarchs have low energy reserves and lack the flexibility to continue moving if resources are not immediately available. Thus, they may die before finding suitable breeding habitat.

The experts indicated that severe drought can cause overwintering tree loss and degradation, decreasing the availability and quality of roosting habitat for monarch butterflies in the West (Pelton et al. 2016, p. 29). Many groves are dominated by one or a few tree species, especially blue gum eucalyptus, which are not native to California and are considered drought sensitive (Marcar et al. 1995, p. 46). Drought-stressed eucalyptus trees are vulnerable to infestation by insect borers, exacerbating tree loss in these groves (Paine and Millar 2002, p. 148), thereby reducing roosting habitat and wind protection. Stressed blue gum eucalyptus may also cease flowering, eliminating the main source of nectar available to monarchs during the overwintering season at some sites. Other dominant trees, such as Monterey pines and Monterey cypress, are more resistant to drought, but these species are the primary species in fewer than 25% of groves.

Furthermore, Fisher et al. (2018, entire) modeled the future location of western monarch overwintering habitat under climate change scenarios in Santa Barbara County, California. They found a substantial shift in predicted overwintering habitat distribution. Monarchs currently overwinter along the coast to take advantage of the mild winter temperatures (Leong 1990, p. 906; Weiss et al. 1991, p. 173), and if temperatures in California are predicted to rise through the year 2100, then similarly cool temperatures, and overwintering monarchs, should be found at higher elevations later this century. Under a plausible scenario (RCP 4.5), the probability of occurrence of overwintering habitat directly reflects elevation, with coastal regions having a reduced probability relative to today, and higher elevation sites increasing in probability. Under a more extreme scenario (RCP 6.0), high probability sites are located only along ridgelines and in mountaintop regions of the county.

We believe the experts' projections are reasonable given (1) there may be small increases in milkweed availability in some portions of the range, and (2) greater losses of monarch habitat from increased temperatures and drought.

Insecticide Exposure

We relied upon expert judgments to quantify the change in insecticide impact, i.e., the expected change in the insecticides state conditions and monarch response for the eastern and western populations (see Voorhies et al. 2019, Supplemental 2). Using the experts' estimates and other information, we devised future projections for the percent change in impact to monarchs. We briefly describe key underlying premises and supporting evidence here; see Insecticide Supplemental for further detail.

Eastern Population

The expert-elicited projected future percent change in the magnitude of impact (monarch population-level response) is a 5% decrease to 30% increase over the next 25 years (Table 6.1). The expert's range is predicated upon the three key premises: 1) there will be no change due to changes in the insecticide doses applied to kill insect pests that reduce crop yields, land use patterns, residential practices, or monarch use of milkweed across the various land uses, 2) there will be a small decrease due to changes in farming practices, and 3) there will be small to high increases in impacts due to additional applications of insecticides because of new agricultural pests that threaten crop yields, new human health threats, and increased vigor of insect pests.

Insecticides are used across a diversity of sectors, with agriculture being the largest source of insecticide exposure for the eastern monarch population (the agriculture comprises 30% of land use within eastern monarch population range and 60% of insecticide use nationwide). The Food and Agriculture Organization of the United Nations (FAO) estimates that a 50% increase in food production by 2050 is needed to meet the demand of the growing human population (FAO 2017, p. 46). In response, corn and soybean production is projected to increase by 16% and 33%, respectively, over the next 10 years (USDA 2020, p. 30, Table 5; p. 35, Table 10). Because only nominal increases in agricultural land expansion is expected in the eastern U.S. (USDA 2020, p. 29; see Milkweed & Nectar Resources section above), this demand will be met primarily through increased yields. Crop production can be greatly diminished by pests. Crop and forest production losses from invasive insects and pathogens in the U.S., for example, have been estimated at $40 billion/year (Paini et al. 2016, p. 7575); similarly, corn and soybean yield losses from pests are estimated to be 54% and 46%, respectively (USDA 2014, p.7). Thus, it is reasonable to foresee efforts to control insect pests intensifying over the next 30 years to meet the increasing demand for food. Additionally, increasing insecticide use among other sectors (e.g., homeowners, forestry, vector control districts) beyond agriculture is expected as well. The number of insect-borne diseases in the U.S., for example, tripled from 2004 to 2016 (CDC 2018), and the causes (e.g., land use changes, increasing transcontinental movements, warming climate) underlying these trends are accelerating (Bradshaw et al. 2016, p. 4-5, FAO 2017, p. 56, 58; Petersen et al. 2016, p. 280).

Moreover, a warming climate is expected to exacerbate insect-borne diseases and pest burden via: 1) improved overwintering survival and faster development and hence increased pest population growth, 2) increased number of generations per season, 3) earlier arrival of migratory pests, and 4) expanding suitable climate envelopes leading to novel pest outbreaks (Caminade et al. 2019, p. 158; Sangle et al. 2015, p.3581; Sharma and Prabhakar 2014, p. 25). Deutsch et al. (2018, p. 918, figure 3) projected, for example, 18% and 32% increase in wheat and corn losses due to insect pests, respectively, with 2°C rise in global temperatures. Although the response of insect pests to climate change will vary, the preponderance of evidence suggests that warmer temperatures in temperate climates will yield more types and higher populations of insect pests and pathogens (Sangle et al. 2015, p. 3580, Wolfe et al. 2008, p. 568). These data indicate an increasing impact from escalating insecticide use into the future.

Some of this increased impact will be mitigated through efforts (e.g., MAFWA, MP3, Rights-of-ways CCAA) to reduce monarch exposure by promoting monarch-specific conservation efforts and increased awareness of the potential harm of insecticides to pollinators, in general. Additionally, the trend towards larger farming operations—which have the capital and capacity to more fully integrate newer technology such as variable rate technology (VRT) and upgrade to newest equipment—may also reduce the monarch's exposure to insecticides. This reduction, however, is likely to be modest as small and mid-size farms still represent a large fraction of acres farmed (e.g., based on a nationwide sample [n=19,600] in 2015, 71% of land was operated by small and mid-size farms; USDA 2016, p. 4).

Given the demand for increasing crop yields and the continued increasing trend in insect pests and insect-borne diseases, increases in insecticide use is foreseeable. Conservation efforts, via reduced exposure potential, are likely to prevent the full impact of these increases from occurring. Thus, we believe the expert's 5% decrease to 30% increase represents a plausible projection of insecticide impacts on the eastern population over the next 25 years.

Western Population

The expert-elicited projected future percent change in the magnitude of impact (monarch population-level response) is a 9% decrease to 68% increase over the next 20 years (Table 6.2). The experts' range is predicated upon the three key premises: 1) areas with high insecticide use overlap significantly with areas most important to monarch production—California's Central Valley, eastern Washington, southern Idaho; 2) the trend in land conversion from rangeland to agriculture will lead to an increasing demand for insecticides by the agricultural sector, and 3) despite California having the strongest pesticide registration in the country, ability to regulate exposure is difficult.

Insecticide use is widespread across the most important breeding areas (Figure 6.4) for the western monarch, and it has been implicated as one of the key drivers in the decline of the western monarch population (Crone et al. 2019, p. 10; Forister et al. 2016, entire; Halsch et al. 2020, entire). Based on volume alone, exposure to insecticides is greatest on or near agricultural lands. Between 2005 and 2012, the agricultural sector, for example, accounted for 60% of insecticide use (USEPA 2017, p. 11). Given the overlap of agriculture and monarch breeding areas, the trend in insecticide use on agriculture greatly influences monarch exposure to insecticides. The increasing demand for food production is expected to expand trade for all the projected agricultural commodities (USDA 2020, p. 55). California is the leading U.S. state in cash farm receipts, and its agricultural production includes more than 400 commodities representing over a third of the United States' vegetables and two-thirds of the country's fruits and nuts (California's Managed Pollinator Protection Plan (MP3); CDPR 2018, p. 1) and ~15% of U.S. agricultural exports for 2017. In the western U.S., this demand for food will be met by expanding agricultural lands (Sleeter et al. 2017) and through increased yields (Popp et al. 2013, p.253), both of which will increase insecticide use in the western U.S.

In addition, insecticide exposure is occurring across a wide variety of land use sectors. A study in the central valley of California, for example, detected pesticides in all land use types (Halsch et al. 2020, p. 13). Insecticides are used by: homeowners to control pests in yards and gardens or planting neonicotinoid-treated ornamentals from garden centers; municipalities to control mosquito populations (WAFWA 2019, p. 16) to prevent the spread of infectious diseases (i.e., West Nile virus, Zika virus); and federal, state, and private entities to control pest irruptions on rangelands (WAFWA 2019, p. 16). These data indicate an increasing impact in the future due to increasing use of and exposure to insecticides.

We expect that some of this impact will be mitigated through efforts (e.g., WAFWA, MP3, Rights-of-ways CCAA) to reduce monarch exposure by promoting monarch-specific conservation efforts and increasing awareness of the potential harm of insecticides to pollinators, in general. The WAFWA plan, for example, points to monarch-specific BMPs and training for all sectors. Additionally, the states of California and Washington have MP3 plans in place and Idaho has a similar plan. The purpose of these plans is to mitigate the pesticide risk to bees, but in doing so, can also lead to reduced monarch insecticide exposure.

Given the increasing demand for agricultural products and the substantial overlap of agricultural lands with important monarch breeding areas, increases in insecticide use or toxicity are foreseeable. Conservation efforts, via reduced exposure potential, are likely to prevent the full impact of these increases from occurring. Thus, we believe the experts' 9% decrease to 68% increase represents a plausible projection of insecticide impacts on the western population over the next 20 years.

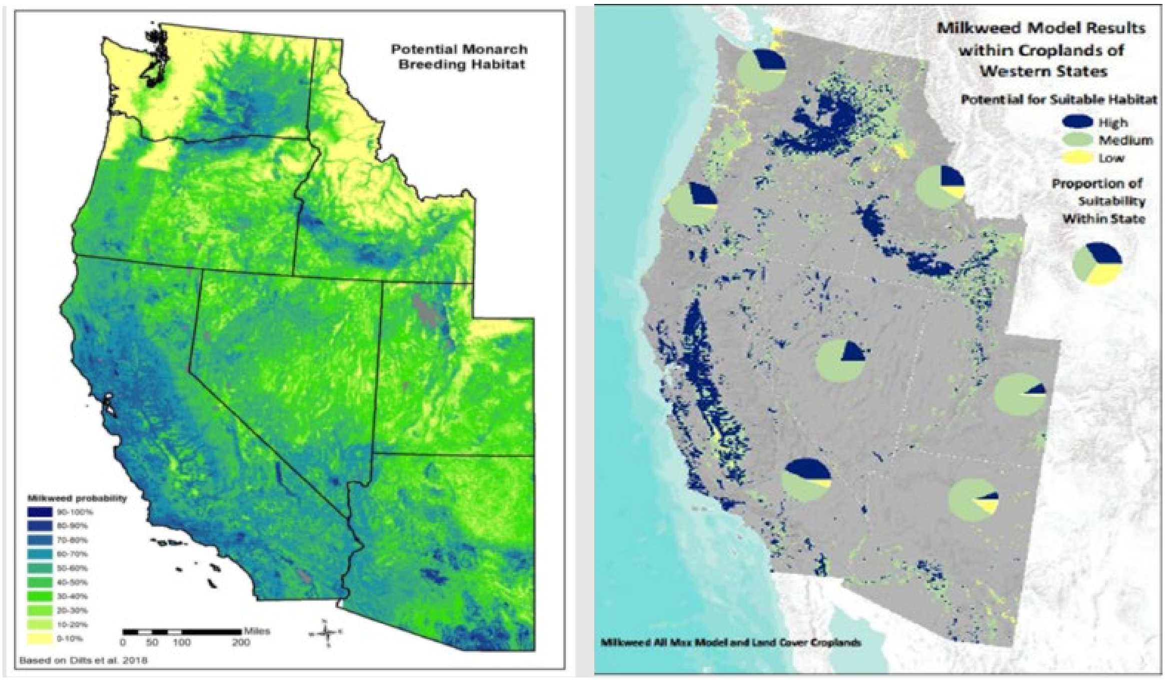

Figure 6.4. A Predicted distribution of milkweed and thus extent of potential monarch breeding areas—derived from a habitat suitability model (Dilts et al. 2018). B. Suitable habitat—milkweed potential—overlaid with croplands in western U.S. (WAFWA 2019, Fig. 6, p. 15). Dark blue spots correlate with the important for breeding areas -- the Central Valley, Columbia River, and Snake River Plain.

North American Populations – Catastrophic Events

We defined catastrophic event as an event that is expected to extirpate the population should the event occur. We evaluated several potential events to determine if they were of sufficient magnitude and severity to cause a population collapse. Below, we describe the events that are likely to be catastrophic should they occur.

<u>Eastern North American Population</u>

We assessed the following events for their potential to cause catastrophic losses: overwintering storms, widespread drought, fire, habitat loss, broad-scale insecticide spray events, and monarch disease and predation. Of these, we determined that two—extreme storm events and widespread drought— have sufficient magnitude (scope) and severity (causing population collapse) potential to pose a catastrophic risk to the eastern population.

Extreme Storm Mortality

Storms during the annual cycle can cause high levels of mortality when monarchs are congregated (during migration and at the overwintering grounds). During migration, storms could be catastrophic if they occurred in areas where monarchs are funneled together (e.g., Texas, where the eastern migratory population funnels through in the spring and fall). However, after an extensive literature search, we found only a few documented incidences of storm mortality during migration (but see Howard and Davis 2012, entire). Moreover, although large numbers of monarchs funnel through at the same time, it is unlikely that storms will cover the relatively large area occupied at any time during migration and thus, not likely to rise to the level of causing population-level losses. Given this, we have insufficient information that the magnitude and severity of storms during migration pose a catastrophic risk.

There is, however, well-documented mortality events at the Mexican overwintering sites from storms (e.g., mortality upwards of 80% has been documented [Brower et al. 2004, p. 158]). Monarchs are particularly sensitive to storms in Mexico because once wetted, monarchs freeze at a warmer temperature (approximately -4°C for wet butterflies, compared to -8°C for dry; Larsen and Lee 1994). Monarch freezing mortality from storms at overwintering sites has been documented during the winters of 1980-1981, 1995-1996, 1999-2000, 2001-2002, 2003-2004, 2009-2010, and 2015-2016 (Oberhauser and Peterson 2003, p. 14063, Brower et al. 2005, p. 970, Fink et al. in prep). Given the potential severity and the high magnitude across the relatively

population. A previous model shows a potential increase in precipitation events in the winter (Oberhauser and Peterson 2003, p. 14066-14067). However, other modeling efforts show a potential decline in freezing storm events due to warming temperatures (Flockhart et al. 2015, p. 160). Additionally, with logging and climate change negatively impacting the oyamel overwintering forests, freezing events may be more likely and more severe because of the loss of the protective effects of an intact forest (Williams and Brower 2015, entire). When combined with a decreasing population size, there is a higher risk that extreme storms of magnitudes similar to previously documented storms would now be catastrophic.

Widespread Drought

Monarchs can be affected by drought at multiple points during their migratory cycle, including during the breeding season as both larvae and adults, and as adults nectaring along their migratory route (nectar can be converted to stored lipids for use while overwintering; Brower et al. 2015). Water availability can affect both milkweed quality and milkweed and nectar availability (Brower et al. 2015, Couture et al. 2015; see also Widespread Drought section under the Western North American discussion below). Given the expansive breeding ground, drought events are unlikely to affect a large enough area to evoke a population level response, and hence not likely to pose a catastrophic risk to the eastern breeding population.

Eastern migratory monarchs funnel through Texas and Mexico in the fall, where it is imperative that they consume enough nectar to be converted to lipids and used as needed throughout their overwintering period (when nectar resources are scarce; Brower et al. 2015). Brower and colleagues (2015) found that monarchs in Texas nectaring on wildflowers during a drought had lowered lipids (compared to monarchs nectaring on flowers from an irrigated garden at the same time). However, they also found that monarchs arriving at Mexican overwintering sites that same year had higher lipid reserves, suggesting that non-drought areas in Mexico may provide sufficient nectar even when Texas is in a drought. This area is also important in the spring, as monarchs funnel through this same area and rely on milkweed and nectar sources as they lay the first generation of the new year. Thus, monarchs in the spring could be similarly impacted by drought. Given the above, it is possible that drought conditions in Texas or Mexico pose a catastrophic risk for the eastern monarch population.

Western North American Population

We assessed the following events for their potential to catastrophic losses: widespread drought, wildfire, extreme overwintering storm events, and co-occurrence of poor environmental conditions and low population numbers. Of these, we determined that two—widespread drought and co-occurrence of poor environmental conditions and low population abundance—have sufficient magnitude (scope) and severity (causing population collapse) potential to pose a catastrophic risk to the western population.

Widespread Drought

Severity and intensity of drought have been suggested as a major driver of monarch populations in the West (Stevens and Frey 2010, p. 740). Severe drought affects both milkweed and nectar resources, and overwintering habitat resources. The frequency of years with precipitation "much below normal" in California and Nevada has increased from 1910 to current (Figure 6.6) and are predicted to increase with climate change (Diffenbaugh et al. 2015, p. 3934; Williams et al. 2015, p. 6826; Cook et al. 2015, p. 6). Under climate change projections, wetter winter conditions and drier spring conditions will lead to greater year-to-year precipitation variability and an overall increase in the frequency of dry years due to fewer wet days (Pierce et al. 2018, p. 27). Additionally, the forecasted higher maximum July temperatures and increased duration of heat waves (Pierce et al. 2013, entire) is likely to increase evapotranspiration (Diffenbaugh et al.

California coast (Pierce et al. 2013, p. 843).

If the tolerance threshold of milkweed and nectar resources to consecutive years of drought is reached, this could result in catastrophic breeding and migratory habitat degradation and loss. A decrease in nectar resources could result in starvation and reduced reproductive output of adults. Milkweed with limited water availability can have more viscous latex, which has been shown to negatively influence larval performance (Bell 1998, p. 133). A decrease in milkweed resources may leave monarchs with fewer resources on which to feed and lay their eggs, resulting in decreased recruitment for the population. However, the majority of milkweeds are deciduous perennials that have adapted to seasonal dry conditions (Borders et al. 2013, p. 7). A mild drought or one that was limited in extent or duration would likely reduce the availability of milkweed to breeding individuals, but the effects to the overall distribution of milkweeds would be short-term. Though a single year of drought could cause fecundity to decline sharply, only a drought that was severe, widespread, and sustained would be catastrophic for a population of monarch butterflies. The breeding ground is widespread for the western population, but large-scale drought could be as equally as widespread (Williams et al. 2020, entire), such that it could occur throughout most of the breeding grounds. Given the above, extreme drought affecting milkweed and nectar resources poses a catastrophic risk for the western monarch population. When combined with a decreasing population abundance, there is a higher risk that drought would be catastrophic.

Severe drought can also cause tree loss and degradation, decreasing the availability and quality of overwintering roosting habitat (Pelton et al. 2016, p. 29). Many groves are dominated by one or a few tree species; one of the most prevalent—blue gum eucalyptus—is drought sensitive (Marcar et al. 1995, p. 46). Drought-stressed eucalyptus trees are vulnerable to infestation by insect borers, which can exacerbate tree loss in these groves (Paine and Millar 2002, p. 148). Eucalyptus loss and degradation reduces availability of roosting habitat, lessens wind protection, and eliminates the primary overwintering source of nectar at many sites. Other dominant trees, such as Monterey pines and Monterey cypress, are more resistant to drought, but are the primary species in fewer than 25% of overwintering sites. Although overwintering grounds are widespread, drought could be equally as widespread, such that it could occur throughout many or most of the overwintering sites simultaneously. Given the above, extreme drought at

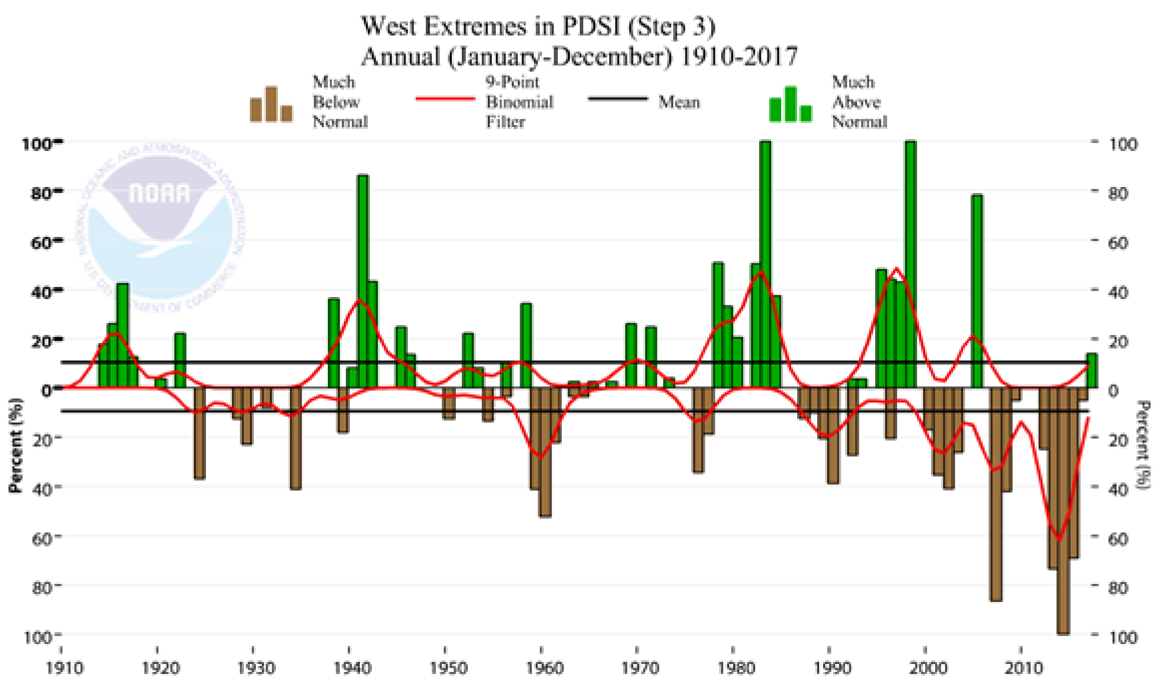

Figure 6.6. Extremes in the Palmer Severity Drought Index (PDSI) for the western U.S. (i.e., California and Nevada). Figure from the National Oceanic and Atmospheric Administration (NOAA 2018).

Co-occurrence of Poor Environmental Conditions with Low Abundance

If the large population fluctuations that were observed in the 1990s (presumably due to poor environmental conditions) were to occur when the population abundance is low (as it has been in recent years), extinction of the western North American population is likely. Given that environmental variability, and thus large swings in abundance, will increase with a changing climate (Pierce et al. 2018, entire) and given that the population has remained at lowest ever abundances for the last 2 years, co-occurrence of poor environmental conditions and low population abundances numbers

Worldwide – Future Scenarios & Catastrophic Events

Due to a lack of information on current influences, we were unable to forecast future scenarios for these populations.

We identified, however, two potential catastrophic events—both of which are climate change effects: sea level rise and lethal high temperatures. To forecast future changes in temperature and sea levels, we relied upon the Third Assessment Report developed by the International Panel on Climate Change (IPCC) to identify the low-lying islands that are at risk of permanent inundation and used the maximum elevation of those islands to develop thresholds for the risk classifications. To forecast changes in daily temperatures, we used downscaled General Circulation Model under RCP scenarios 4.5 and 8.5 obtained from the Earth System Grid Federation (CORDEX 2018; Cinquini 2014). Using these data, we assessed where daily maximum surface temperatures would exceed 42°C (a temperature threshold that leads to significantly reduced monarch larvae survival; Nail et al. 2015b, p. 99) by the year 2069 (see Appendix 2 Methods – Climate change projections for further details).

<u>Sea Level Rise</u>

Several low-lying islands in the pacific region are at risk of permanent inundation according to the Third Assessment Report from the IPCC (IPCC 2001). Many of these low-lying islands are inhabited by monarch butterflies. Additionally, many of these islands are remote and represent an entire population of monarchs. A mix of elevations occurs on these islands. We assumed that monarch populations on islands with higher elevations are at a lower risk level. However, we do not have any data on the population size or extent of habitat on these islands.

<u>Unsuitably High Temperatures</u>

In addition to sea level rise, temperatures are expected to increase throughout parts of the monarch's range (IPCC 2001). While monarchs can tolerate a range of thermal conditions, there are known upper limits (Nail et al. 2015b). Therefore, we also examined future predicted temperatures throughout the global range of monarchs, presuming that areas exceeding these lethally high thermal thresholds would have catastrophic losses of monarchs.

Chapter 7: Results – Analysis of Future Condition

This chapter describes the forecasted health of monarch populations over time. We first describe the results from our analysis of direct effects from high temperatures due to climate change. Next, we provide the forecasted health of the North American populations given the best and worst case scenarios. Lastly, we provide the results of the catastrophic events analysis for the worldwide populations.

Eastern North American Population – Future Condition

Under both best and worst case scenarios, the population continues to decline ($\lambda < 1$, Figure 7.1). The greatest impact on the population occurs during the first 20 years for both scenarios; lambda increases by 1.5% from 0.960 to 0.975 under the best case scenario and declines by -4.5% from 0.960 to 0.917 under the worst case scenario (Figure 7.1). As expected under a declining trajectory, the pE increases over time (Figure 7.2). By year 60, pE ranges from 56% to 74% (see Appendix 3, Table 3A3 for decadal projections).

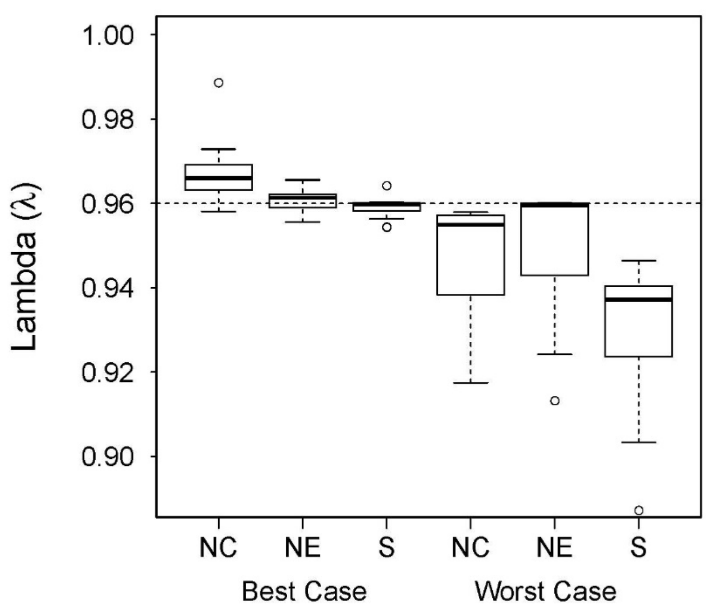

Figure 7.1. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for each of the subregions of the eastern population (NC=Northcentral, NE= Northeast, S=South). The dashed line represents the current population growth rate (λ=0.96).

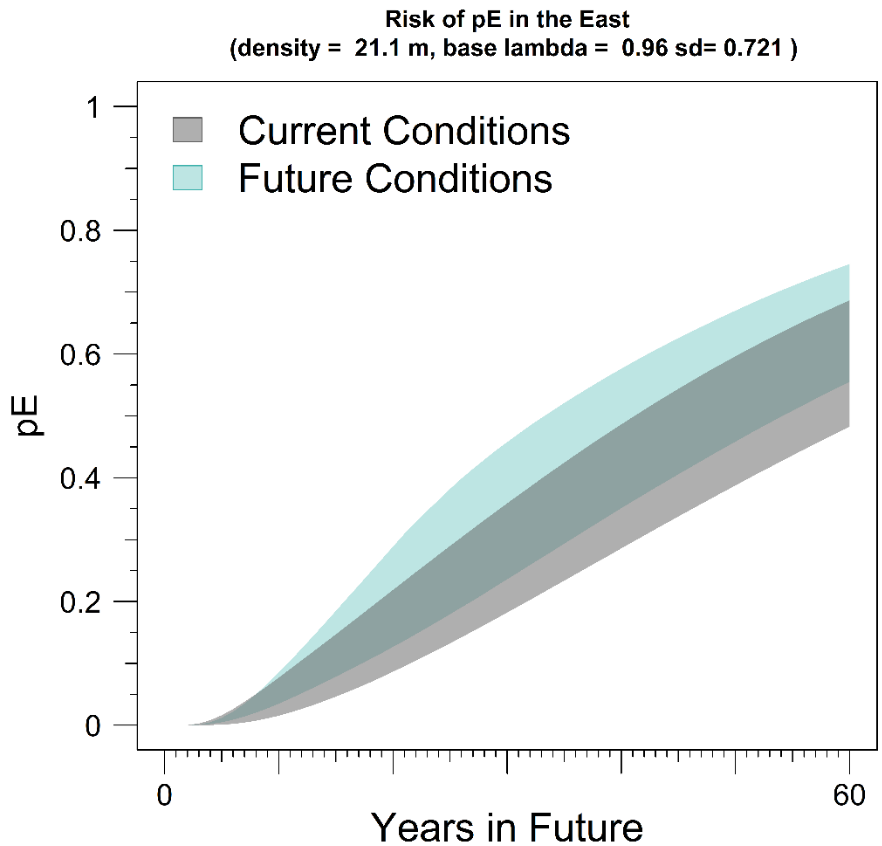

Figure 7.2. pE, for the eastern North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 60, pE ranges from 56% to 74% under the best and worst case future scenarios, respectively.

<u>Direct Effects from High Temperatures & Catastrophic Events</u>

We were unable to incorporate direct effects from increasing temperatures and catastrophic risks into the population models, so we qualitatively discuss the implications of these factors on the future condition of the population. We evaluated the change in the spatial extent and number of "cell days" (i.e., raster grid cells) with projected temperatures above thermal thresholds during critical time periods in monarch migration (see Appendix 2 - Climate change projections for further details). Under the RCP 4.5 scenario, both the spatial extent and the average number of >38°C days (sublethal and moderate survival reductions) are projected to decrease in the northcentral subregion but markedly increase in the south (94% and 331%, for area and number of days, respectively) and northeast subregions in April and May (Figure 7.3, see Appendix 3 for values for all subregions). The spatial extent and average number of cell days above the lethal threshold (42°C) are projected to increase dramatically for the south (6,630% and 8,147%, respectively) during the same period (Figure 7.3). Given these results, monarch reproductive success and survival rates of the first generation of monarchs are likely to decline, although the extent of which these rates will decline is unknown.

Similarly, given the projections of monarch health described above, the eastern population will be increasingly vulnerable to catastrophic losses due to both extreme storm and widespread drought events. Although we cannot quantify this increased risk, the longer the eastern population remains unhealthy, the more likely it is that catastrophic losses will occur and the greater the extinction risk for the eastern population.

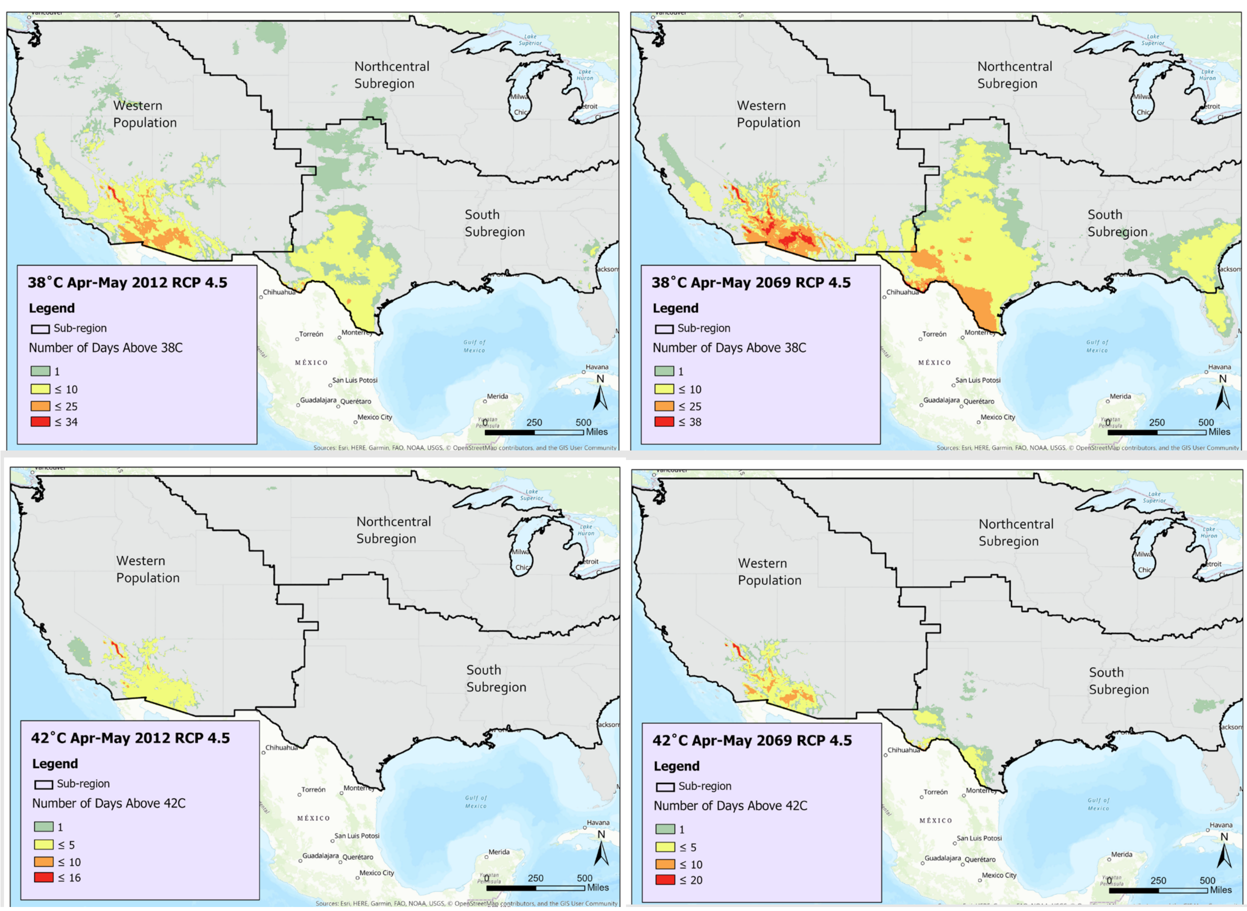

Figure 7.3. The projected spatial extent and average number of cell days between April and May where temperatures >38˚C (top) and 42˚C (bottom) in 2012 (left) and 2069 (right) under RCP 4.5. Colors represent number of cell days above >38˚C and 42˚C.

Western North American Population – Future Condition

Under both scenarios, the population continues to decline ($\lambda < 1$, Figure 7.4). Under the best case scenario, greatest positive effect occurs in years 21-50 when lambda slightly increases by 0.3% from 0.878 to 0.881; under the worst case scenario, the population is most affected during the first 20 years when lambda decreases -5.8% from 0.878 to 0.828. As would be expected with a declining growth, the pE increases over time (Figure 7.5). At year 10, pE ranges from 66 to 71% and reaches 99% by year 60 (see Appendix 3, Table 3A3 for decadal projections).

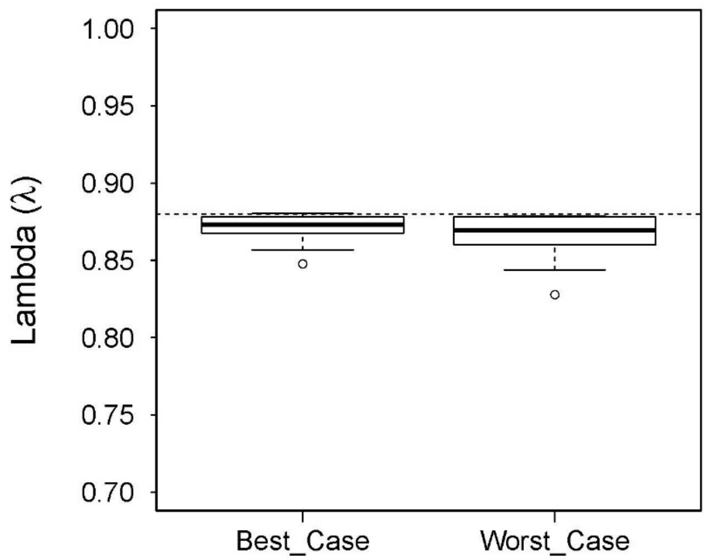

Figure 7.4. Box plot for population growth rate (lambda, λ) under the best and worst case scenarios for the western population. The dashed line represents the current population growth rate (λ=0.878).

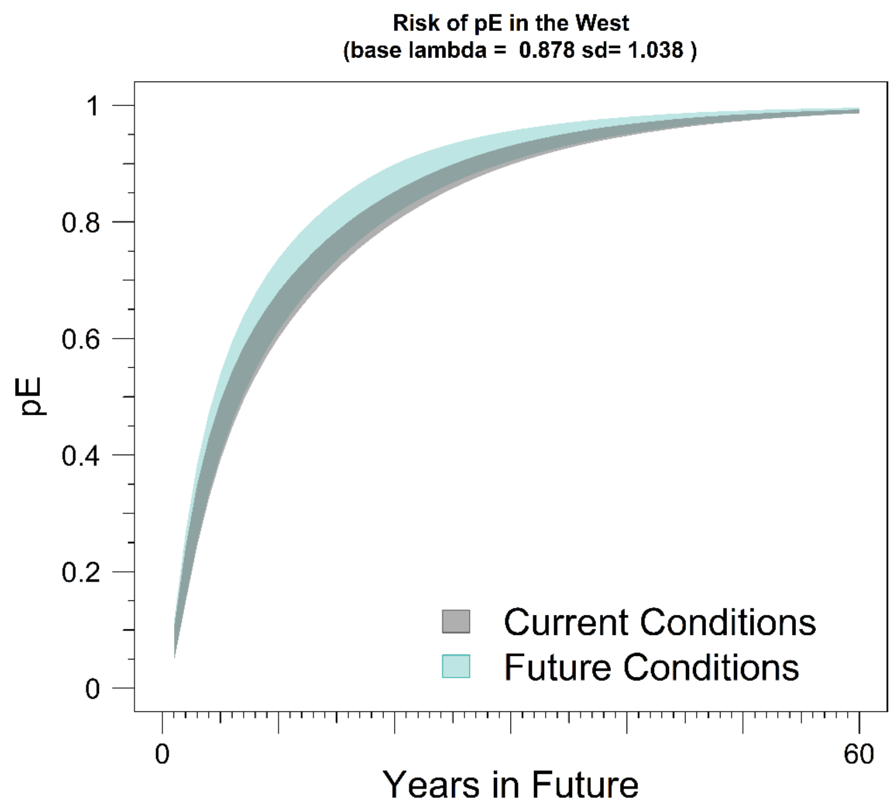

Figure 7.5. pE, for the western North American monarch population over time, given both current (gray band) and projected changes in state conditions (blue band). By year 60, pE reaches 99% under the best and worst case future scenarios.

Direct Effects from High Temperatures & Catastrophic Events

Under the RCP 4.5 scenario, the spatial extent of the area over which the average number of degree days >38°C and >42°C is projected to decrease (-23% and -11%, respectively), while increases in the average number of days >38°C (38%) and >42°C (11%) are projected (see Figure 7.1 and Appendix 3, Tables 3A1-A3 for further results). Given these results, monarch reproductive success and survival rates are likely to decline, although the extent of which these rates will decline is unknown.

Similarly, given the projections of monarch health described above, the western population is vulnerable to catastrophic losses due to both widespread drought events and the co-occurrence of poor environmental conditions and low population abundance. The risk of extinction due to these events increases the longer the population remains at the current low abundances.

Worldwide Populations – Risks due to Catastrophic Events

We qualitatively assessed the impact due to predicted climate change effects. Fifteen of the 29 populations are classified as being "at risk" to extinction due to sea level rise or due to increasing temperatures (Table 7.1).

Table 7.1. Qualitative expression of risk due to predicted sea level rise and high temperatures. See definitions of terms in Table 3.2.

ACU	Population	Status	High Temps	Sea Level Rise
Australia, New Zealand, and Indo-Pacific Islands	Australia	Extant	At Risk	No Known Risk
	Cook Island	Extant	No Known Risk	No Known Risk
	French Polynesia	Extant	No Known Risk	No Known Risk
	Greater Indonesia	Unknown	At Risk	No Known Risk
	Guam & CNMI	Extant	No Known Risk	No Known Risk
	Johnston Atoll	Extant	No Known Risk	At Risk
	Kiribati	Extant	No Known Risk	At Risk
	Marquesas Islands	Extant	No Known Risk	No Known Risk
	Marshall Islands	Extant	No Known Risk	At Risk
	Mascarene Islands	Extant	No Known Risk	No Known Risk
	Micronesia	Extant	No Known Risk	No Known Risk
	Nauru	Extant	No Known Risk	At Risk
	New Zealand	Extant	No Known Risk	No Known Risk
	Norfolk Island	Extant	No Known Risk	No Known Risk
	Palau	Extant	No Known Risk	No Known Risk
	Papua New Guinea	Extant	At Risk	No Known Risk
	Philippines	Unknown	At Risk	No Known Risk
	Samoa	Extant	No Known Risk	No Known Risk
	South Pacific Islands	Extant	No Known Risk	No Known Risk
	Tokelau	Unknown	No Known Risk	At Risk
	Tonga	Extant	No Known Risk	No Known Risk
	Tuvalu	Extant	No Known Risk	At Risk
	Wallis & Futuna	Unknown	No Known Risk	No Known Risk
Central America & the Caribbean	Caribbean	Extant	At Risk	No Known Risk
	Central America	Extant	At Risk	No Known Risk
Southern Florida	Florida	Extant	At Risk	No Known Risk
Hawaii	Hawaii	Extant	No Known Risk	No Known Risk
Iberian Peninsula	Iberian Peninsula	Extant	At Risk	No Known Risk
South America & Aruba	South America and Aruba	Extant	At Risk	No Known Risk
Eastern North America	Eastern North America	Extant	See E. North American pop below	
Western North America	Western North America	Extant	See W. North American pop below	

Chapter 8: Synthesis – Implications for Viability

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences of the change in the number, health, and distribution of populations over time for the viability of the monarch. We assessed monarch viability by evaluating the species' ability to withstand environmental stochasticity (resiliency), catastrophes (redundancy), and changes in its environment (representation). We also discuss the key uncertainties and their implications for the analyses.

Viability

Monarch viability depends upon its ability to sustain populations in the face of normal environmental stochasticity, catastrophes, and novel changes in its environment. The species' ability to do so is influenced by the health and distribution of its populations. Demographically and physically healthy populations are better able to withstand and recover from environmental variability and disturbances and are more likely to withstand and recover from events that would otherwise be catastrophic. Populations spread across heterogeneous conditions are unlikely to be exposed at the same time to poor environmental conditions, thereby guarding against synchronous population losses. Lastly, populations spread across the breadth of genetic and phenotypic diversity help to preserve species' adaptive capacity, which is essential for adapting to their continuously changing environment (Nicotra et al. 2015, p. 1269). Without such variation, species are less responsive to change and more prone to extinction (Spielman et al. 2004, p. 15263). Additionally, as populations with higher genetic diversity can more quickly adapt to novel changes, species with genetically healthy populations (large N_e, which begets genetic diversity) are better able to adapt (Ofori et al. 2017, p.2).

Historically, monarchs were widely distributed across 90 countries, islands, and island groups. Currently, monarchs remain widespread with 27 extant populations and 4 with unknown status. Despite being widespread across a diversity of habitats, environmental gradients, and climates, we found 15 of the worldwide populations are 'at risk' of extinction, and the populations comprising the core of the species—eastern and western North American populations—have declining growth rates and increasing extinction risks. While the North American migratory populations naturally fluctuate year-to-year with environmental conditions, they have declined over the last 20 years (Figure 8.1). These declines are due primarily to: (1) loss and degradation of habitat [from conversion of grasslands to agriculture, widespread use of herbicides, logging/thinning at overwintering sites in Mexico, senescence and incompatible management of overwintering sites in California, urban development, and drought]; (2) continued exposure to insecticides; and (3) effects of climate change. Because monarch populations fluctuate with environmental conditions, populations must be large and have strong population growth potential to withstand natural environmental variation and disturbances. Given their current low population sizes and declining growth rates, these populations will likely continue to decline without threat abatement. The magnitude or frequency (or both) of these threats, are expected to increase (Figures 6.1 & 6.2) further exacerbating declines (in abundance and growth rates) and increasing extinction risks (Figures 7.3 & 7.5). The recent steep decline of the western population may be a consequence of small population effects (i.e., an extinction vortex due to Allee effects and increased sensitivity to environmental stochasticity); in which case, amelioration of threats may not be enough to stall extinction. The western population trajectory may portend the future for the eastern population if its declining population trend is not reversed (i.e., insufficient resiliency to rebound from poor years resulting in steep and rapid declines). The health of the North American populations is declining, rendering both less able to withstand and recover from poor environmental conditions and withstand stressors. Under future state conditions, the resiliency of these populations will continue to decline as reflected in their increasing pE (the probability of the population abundance reaching the point at which extinction is inevitable) estimates over time.

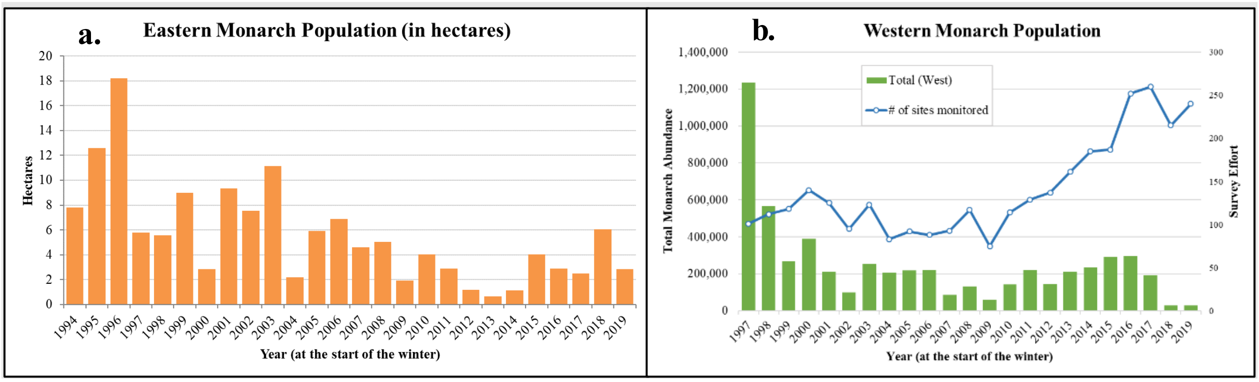

Figure 8.1. Eastern (a) and western (b) North American monarch population sizes, as measured at overwintering sites in terms of hectares (eastern) and total number (western). The western population counts also has a blue line indicating survey effort (number of sites monitored).

Moreover, the estimates of pE do not include risks from large, consequential stochastic events and direct effects of high temperatures due to climate change. At their current low abundances, these populations are more vulnerable to events that would otherwise be non-catastrophic. For example, had either of the two potentially catastrophic storms (where estimated mortality exceeded 70%) on the Mexico overwintering sites occurred during a low abundance year, the eastern North American population may have been extirpated. The longer these populations remain unhealthy (i.e., impaired growth potential and low abundance), the greater their risk to extinction due to stochastic events alone. Additionally, under climate change projections (both RCP 4.5 and RCP 8.5), the number of days with—as well as the spatial extent where monarchs will be exposed to—lethally high temperatures is projected to increase markedly and thus reduce monarch survival and reproductive rates in the affected subregions. Neither the risks from catastrophic events nor high temperature effects are fully captured in our pE estimates.

The extinction of either the western or eastern North American migratory population would increase the risk of losing the North American migratory phenomenon, as its persistence would depend solely upon the continued survival of a single population. Moreover, loss of either population would impair the overall ability of the species to adapt in the future. Although each of the 8 delineated ACUs represent unique sources of adaptive diversity, and therefore individually contribute to the monarch's adaptive capacity, the eastern and western ACUs are especially important. In addition to being genetically distinct and possessing greater allelic diversity than all other ACUs, monarchs in the eastern and western North American ACUs exhibit the long-distance migratory phenotype, occupy different climates and habitat niches, and differ in reproductive behavior and possibly disease resistance. Further, these North American populations represent the historical and current core of the species and the ancestral lineage of the species. Accordingly, loss of these two ACUs would reduce monarch diversity, rendering the species less able to adapt to novel changes in its environment now and in the future and thereby increasing the extinction risk of the species. The chance of *both* populations persisting above the extinction threshold over the next 10 years is 27% to 33% (under future conditions) and drops under 10% within 30 years.

Much of this risk is due to the poor condition of the western population. The western North American population comprises approximately 30% of the area occupied by monarch butterflies in North America and contributes unique variation in migratory, overwintering, and reproductive behavior; ecology; wing morphology; and flight power. Western monarchs expand outward from their overwintering sites, while monarchs in the eastern population shift the range northward. Western overwintering monarchs may have a shorter diapause and may also differ in mating behavior. Western monarchs differ in their ecology from eastern monarchs in their use of different species of nectar and milkweed plants and different roosting tree species. Lastly, differences are seen in divergent wing morphology and flight power between eastern and western monarchs. Additionally, a recent genomics analysis indicates low levels of dispersal between eastern and western monarch butterflies, suggesting that they are demographically independent. So, although unquantifiable, the loss of the western population would reduce the monarch's diversity and likely its ability to adapt to changes in its environment, thereby increasing the extinction risk of the North American monarchs.

Based on this information and other analyses included in this SSA, monarch viability is declining and projected to continue declining over the next 60 years.

Uncertainties

Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios, adding environmental stochasticity to our model, applying stochastic extinction thresholds, and making reasonable assumptions. These assumptions, albeit necessary, impact the results of our analyses. Here, we highlight the key uncertainties, our accompanying assumptions, and our assessment of the relative influence they impose on the results. When we say that these key assumptions impact the analysis of monarch viability, we mean they may directly impact estimates of the monarch's (a) ability to withstand environmental stochasticity (resiliency), (b) ability to withstand catastrophic events (redundancy), (c) ability to adapt to novel changes in their environment (representation), and (d) vulnerability to extinction.

Historical Conditions

The historical range of monarch includes sites outside of North America, with monarchs documented throughout this range from the mid- to late-1800s. We know monarchs were present in North America prior to the 1800s, but we do not know the full extent of their range. We assume that monarchs that are present outside of North America have become naturalized. This assumption may overestimate the historical viability of monarchs worldwide.

Current Conditions

The key uncertainties that impact our ability to interpret current monarch viability include: (1) current status and health of worldwide populations, (2) current, independent population growth of North American populations (the lack of links between their population numbers), (3) extinction thresholds for both eastern and western populations, and (4) density estimates for the eastern population.

Worldwide Populations Status and Health

There is a paucity of data on monarch occurrence over time, distribution, and habitat use. We assumed that all populations in which at least a single monarch has been documented since the year 2000 are currently extant (either known or presumed). To assume these worldwide populations are extant will overestimate the current representation and resiliency of monarchs globally and, subsequently, overestimate the viability of the species.

Exchange of Individuals among the North American Populations

Marking data from Morris et al. (2015, pp. 100, 102) indicate that at least some individuals migrate from the western United States to overwintering grounds in Mexico and that monarchs can return from Mexico to the western United States to breed (Brower and Pyle 2004, p. 155; Dingle et al. 2005, p. 498), but we do not know at what rate. We also know that some monarchs that migrate south through the eastern United States to overwinter in Mexico break diapause to breed in the Gulf region (Howard et al. 2010, p. 2) and likely supplement non-migratory populations that breed year-round in southern Florida (Knight and Brower 2009, p. 819). Similar to other models (Semmens et al. 2016, Schultz et al. 2017), our model does not include immigration and emigration parameters from our population models for the eastern and western North American populations. This assumption of lack of connectivity could underestimate the current resiliency of each population and thus underestimate monarch viability. This uncertainty and its corresponding assumption also apply for future conditions and again likely underestimate monarch viability into the future.

Alternate Overwintering Strategies

It is believed that a majority of eastern and western North American monarchs overwinter in reproductive diapause in Mexico and along the California and Mexican coast, respectively (see *Individual-Level Ecology and Requirements* in Chapter 2). However, there are known exceptions to this overwintering pattern. There are monarchs that remain or become reproductively active and breed throughout the winter along the Gulf Coast, the southern Atlantic Coast, and the southern Pacific Coast (Howard et al. 2010, p. 3; Satterfield et al. 2016, p. 346). These monarchs are more likely to be infected with *OE* (Satterfield et al. 2016, 2018, p. 347, p. 1676, respectively), and there is some question of whether some of the offspring of these individuals might emerge in diapause and continue to Mexico or California overwintering sites later in the season (Batalden and Oberhauser 2015, p. 223).

Additionally, there are other, smaller overwintering areas for the eastern and western North American population that exist with monarchs overwintering in diapause. For the eastern population, these include small colonies east of Mexico City (e.g., a site with small aggregations along western slopes of the Popocatépetl volcano; Calvert and Brower 1986, p. 171), and along the coast of North Carolina (where 94 monarchs were captured during overwintering dates over the course of 13 years; McCord and Davis 2010, p. 413). For the western population, these include several small inland California and Arizona overwintering sites (Morris et al. 2015, p. 98; Pelton et al. 2016 p. 10). Because of the relatively small number of monarchs at these sites and their transient nature, we have assumed that Mexico and California annual counts represent the large majority of the eastern and western monarch populations, respectively.

Density Estimates for the Eastern North American Population

The density (# of overwintering monarchs/ha) at the overwintering grounds in Mexico is uncertain and fluctuates within and among years. Because monarch overwintering population size in Mexico is measured in hectares, the assumed density value determines the initial population size estimate, *N(t)OW*, which can influence model results. Published estimates of these densities range from 6.9-60.9 million monarchs per hectare (Calvert 2004, p. 125); Thogmartin et al. (2017a) estimated that the 95% credible interval ranges from 2.4 - 80.7 million monarchs per hectare. We used the median density estimate of 21.1 million (Thogmartin et al. 2017a, p. 10) for our initial population size estimates, and we assumed that density, as reported by annual monitoring efforts, has remained consistent year to year. The chosen density greatly influences the probability of persistence estimates, and thus, likely monarch viability. Monarch viability could be over or underestimated due to our choice in density estimate.

Extinction Threshold

Another key uncertainty is the population size in which environmental stochasticity and Allee effects begin to override the population dynamics (i.e., reinforcing processes drive the population downward towards extinction, extinction vortex). The model samples extinction thresholds from a uniform distribution defined by two sources: expert elicitation for the eastern population (Voorhies et al., 2019, Suppl. 2) and Schultz et al. 2017 and Wells et al. 1990 for the western population. Therefore, we could be either overestimating or underestimating extinction risk under current conditions depending on the accuracy of the thresholds. This uncertainty and its assumptions also apply to future conditions.

<u>Future Conditions</u>

Most of our uncertainty related to monarch viability rests with our analyses of future conditions. These key uncertainties include (1) the future health and persistence of global populations, (2) the relationship between threats and population responses, (3) extinction thresholds for the migratory eastern and western North American populations, and (4) the correct way to account for the multi-generational growth of the migratory eastern North American monarch population.

Worldwide Populations Status and Health

Similar to current conditions, there is a lack of monitoring or survey data necessary to predict future population growth trends for worldwide populations. We are unable to evaluate the impact of threats like habitat loss (land-use change) or pesticide use because we lack information on the specific locations of monarchs within these worldwide geographies. We do assume that monarchs will be extirpated from islands that are completely drowned due to sea level rise. In all other cases, we assume that monarch populations will persist into the future and this may lead to overestimating the viability of the species.

The Relationship between Influences and Population Response

Outside of milkweed and breeding, we lack direct and causal relationships between monarch population size and threats. We assume that our expert-elicited response curves and scenarios accurately represent these unknown relationships. Additionally, we assume that influences are additive and that their rates remain constant over time, an assumption mirrored in a retrospective threats analysis done by Thogmartin et al. 2017b (threats analysis). To assume influences can be simply added and remain constant over time (rather than including interactions or rate changes), likely leads to an underestimate of the vulnerability of extinction of both eastern and western populations. These assumptions in our eastern and western population models likely lead to an overestimate of monarch viability by increasing the resiliency of eastern and western populations.

Furthermore, we overestimate the resiliency of eastern and western populations through our assumptions addressing uncertainties in climate and insecticide influences on these populations. For climate change, we assume that the newly available monarch habitat will be in the northern portion of its current breeding range and beyond and that the migration success rates will be unchanged. We assume that they will be able to take advantage of this habitat and successfully migrate, and we also assume that the large scale modeled niche is indicative of suitable microclimate for monarchs. For insecticide use, we lack information on changes to effectiveness of insecticides or societal pressure to reduce insecticide use. Therefore, we assume very little change in the influence of insecticides on monarch populations into the future.

Extinction Thresholds

Just as in current conditions, the extinction thresholds for both eastern and western populations are a source of uncertainty. This uncertainty follows the same discussion and rationale as described in the current conditions section. Therefore, we could be either overestimating or underestimating extinction risk under future conditions depending on the accuracy of our expert-informed thresholds. The uncertainty of extinction thresholds will impact our estimate of monarch resiliency and possibly overestimate or underestimate the viability of the monarch as a species.

Multi-Generational Growth of the Eastern Monarch Population

Published models of monarch population growth vary in accounting for the multi-generational migration and growth of the eastern monarch population. Some only estimate growth of the overwintering population (Semmens et al. 2016) while others model the growth of subregions within the eastern monarch population (Flockhart et al. 2015, Oberhauser et al. 2017a). Here we assume that modeling population growth at the sub-regional level (Northcentral, Northeast, and South regions) is appropriate (as done in Oberhauser et al. 2017a and published in Voorhies et al. 2019). Experts who participated in our expert elicitation provided estimates of the relative importance of each of these regions to the Mexico overwintering population used in our modeling. This assumption leads to redundancy in influences (both negative and positive) in the different subregions. This in turn, can lead to either an under- or overestimation of the vulnerability to extinction of the eastern population. This redundancy occurs because the population can respond differently to these influences in different regions (because of differing population response curves). As an example, if one region is critically impacted by a negative influence, there are still other regions to contribute to the overall population size. Furthermore, because the eastern monarch population is such a large component of the monarch species, the robustness of this population could lead to over- or underestimating the viability of monarch butterfly.

Literature Cited

Abatzoglou JT, Brown TJ. 2012. A comparison of statistical downscaling methods suited for wildfire applications. International Journal of Climatology 32:772-780.

Alonso-Mejía A, Rendón-Salinas E, Montesinos-Patiño E, Brower LP. 1997. Use of lipid reserves by monarch butterflies overwintering in Mexico: implications for conservation. Ecological Applications 7:934-947.

Altizer SM. 2001. Migratory behaviour and host-parasite co-evolution in natural populations of monarch butterflies infected with a protozoan parasite. Evolutionary Ecology Research 3:611-632.

Altizer S, Davis AK. 2010. Populations of monarch butterflies with different migratory behaviors show divergence in wing morphology. Evolution 64:1018-1028.

Altizer SM, Oberhauser KS. 1999. Effects of the protozoan parasite *Ophryocystis elektroscirrha* on the fitness of monarch butterflies (*Danaus plexippus*). Journal of Invertebrate Pathology 74:76-88.

Altizer S, de Roode JC. 2015. Monarchs and their debilitating parasites: Immunity, migration, and medicinal plant use. Pp. 83-93 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Altizer SM, Oberhauser KS, Brower LP. 2000. Associations between host migration and the prevalence of a protozoan parasite in natural populations of adult monarch butterflies. Ecological Entomology 25:125-139.

Altizer S, Brower L, Howard E, James D, Jepsen S, Lewandowski E, Morris G, Nail K, Oberhauser K, Pyle R. 2015. Captive breeding and releasing monarchs [online]. https://journeynorth.org/tm/monarch/conservation_action_release.pdf [Accessed 6 July 2020]

Anderson JB, Brower LP. 1996. Freeze-protection of overwintering monarch butterflies in Mexico: critical role of the forest as a blanket and an umbrella. Ecological Entomology 21:107-116.

Arnott HJ, Smith KM, Fullilove SL. 1968. Ultrastructure of a cytoplasmic polyhedrosis virus affecting monarch butterfly *Danaus plexippus*: I. Development of virus and normal polyhedra in larva. Journal of Ultrastructure Research 24:479-507.

Bargar TA, Hladik ML, Daniels JC. 2020. Uptake and toxicity of clothianidin to monarch butterflies from milkweed consumption. PeerJ 8:e8669 DOI 10.7717/peerj.8669

Barriga PA, Sternberg ED, Lefèvre T, de Roode JC, and Altizer S. 2016. Occurrence and host specificity of a neogregarine protozoan in four milkweed butterfly hosts (*Danaus* spp.) Journal of Invertebrate Pathology 140:75-82.

Bartel RA, Oberhauser KS, De Roode JC, Altizer SM. 2011. Monarch butterfly migration and parasite transmission in eastern North America. Ecology 92:342-351.

Batalden RV, Oberhauser KS. 2015. Potential changes in eastern North American monarch migration in response to an introduced milkweed, *Asclepias curassavica*. Pp. 215-224 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Batalden RV, Oberhauser K, Peterson AT. 2007. Ecological niches in sequential generations of eastern North American monarch butterflies (Lepidoptera: Danaidae): The ecology of migration and likely climate change implications. Environmental Entomology 36:1365-1373

Baxter-Gilbert JH, Riley JL, Neufeld CJH, Litzgus JD, Lesbarrères D. 2015. Road mortality potentially responsible for billions of pollinating. Journal of Insect Conservation 19:1029-1035.

Beever EA, O'Leary J, Mengelt C, West JM, Julius S, Green N, Magness D, Petes L, Stein B, Nicotra AB, Hellmann JJ. 2016. Improving conservation outcomes with a new paradigm for

137.

Bell E. 1998. Multitrophic level influences on the population ecology of the monarch butterfly (*Danaus plexippus* L.). Doctoral Thesis. University of California at Santa Cruz, Santa Cruz.

Berec L, Angulo E, Courchamp F. 2007. Multiple Allee effects and population management. Trends in Ecology and Evolution 22:185-91.

Blakley NR, Dingle H. 1978. Butterflies eliminate milkweed bugs from a Caribbean Island. Oecologia 37:133-136.

Borders B, Lee-Mäder E. 2015. Project Milkweed: A strategy for monarch habitat conservation. Pp. 190-196 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Borders B, Casey A, Row JM, Wynia R, King R, Jacobs A, Taylor C, Lee-Mäder E. 2013. Pollinator plants of the central United States: Native milkweeds (*Asclepias* spp.). The Xerces Society for Invertebrate Conservation, Portland OR, in collaboration with USDA-NRCS. 21 pp.

Bouseman JK, Sternburg JG. 2001. Field guide to butterflies of Illinois. Illinois Natural History Survey. Champaign, IL.

Braak N, Neve R, Jones AK, Gibbs M, Breuker CJ. 2018. The effects of insecticides on butterflies – A review. Environmental Pollution 242a:507-518.

Bradshaw CJA, Leroy B, Bellard C, Roiz D, Albert C, Fournier A, Barbet-Massin M, Salles J-M, Simard F, Courchamp F. 2016. Massive yet grossly underestimated global costs of invasive insects. Nature Communications 7:12986 (8 pp.).

Brower LP. 1995. Understanding and misunderstanding the migration of the monarch butterfly (Nymphalidae) in North America: 1857-1995. Journal of the Lepidopterists Society 49:304-85.

Brower LP. 1996. Monarch butterfly orientation: Missing pieces of a magnificent puzzle. Journal of Experimental Biology 199:93-103.

Brower LP, and Calvert WH. 1985. Foraging dynamics of bird predators on overwintering monarch butterflies in Mexico. Evolution 39:852-868.

Brower LP, Pyle RM. 2004. The interchange of migratory monarchs between Mexico and the western United States, and the importance of floral corridors to the fall and spring migrations. Pp. 144-166 in Nabhan GP, ed. Conserving Migratory Pollinators and Nectar Corridors in Western North America. Tucson: University of Arizona Press.

Brower LP, Calvert WH, Hedrick LE, Christian J. 1977. Biological observations on an overwintering colony of monarch butterflies (*Danaus plexippus*, Danaidae) in Mexico. Journal of the Lepidopterists Society 31:232-242.

Brower LP, Fink LS, Leong K, Oberhauser K, Altizer S, Taylor O, Vickerman D, Calvert WH, Van Hook T, Alonso-Mejia, A, Malcolm, SB. 1995. On the dangers of interpopulational transfers of monarch butterflies – discussion. Bioscience 45:540-544.

Brower LP, Kust DR, Rendón-Salinas E, García Serrano E, Kust KR, Miller J, Fernandez del Rey C, Pape K. 2004. Catastrophic winter storm mortality of monarch butterflies in Mexico during January 2002. Pp. 151-166 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Brower LP, Fink LS, Ramírez LR, Weiss SB, Slayback D, Perault DR. 2005. The physiological ecology of overwintering monarch butterflies. Integrative and Comparative Biology 45:970.

Brower LP, Williams EH, Fink LS, Slayback DA, Ramírez MI, Garcia MV, Zubieta RR, Weiss SB, Calvert WH, Zuchowski W. 2011. Overwintering clusters of the monarch butterfly coincide with the least hazardous vertical temperatures in the oyamel forest. Journal of the Lepidopterists Society 65:27-46.

Brower LP, Taylor OR, Williams EH, Slayback DA, Zubieta RR, Ramírez MI. 2012. Decline of monarch butterflies overwintering in Mexico: Is the migratory phenomenon at risk? Insect Conservation and Diversity 5:95-100.

Brower LP, Fink LS, Kiphart RJ, Pocius V, Zubieta RR, Ramírez MI. 2015. Effect of the 2010-2011 drought on the lipid content of monarch butterflies migrating thorough Texas to their overwintering sites in Mexico. Pp. 117-129 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Brower LP, Slayback DA, Jaramillo-López P, Ramírez I, Oberhauser KS, Williams EH, Fink LS. 2016. Illegal logging of 10 hectares of forest in the Sierra Chincua monarch butterfly overwintering area in Mexico. American Entomologist 62:92-97.

Buden DW, Miller JY. 2003. Butterflies of Pohnpei, Eastern Caroline Islands, Micronesia. Pacific Science 57:1-8.

California Department of Finance (Demographic Research Unit). 2019. Demographics (Projections and Population Estimates). http://www.dof.ca.gov/Forecasting/Demographics/ [Accessed 06 May 2020].

[CDPR] California Department of Pesticide Regulation. 2018. California's Managed Pollinator Protection Plan (MP3). 24 pp.

Calvert WH. 2004. Two methods for estimating overwintering monarch population size in Mexico. Pp. 121-127 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Calvert WH, and Brower LP. 1986. The location of monarch butterfly (*Danaus plexippus* L.) overwintering colonies in Mexico in relation to topography and climate. Journal of Lepidopterists' Society 40:164-187.

Calvert WH, Hedrick LE, Brower LP. 1979. Mortality of the monarch butterfly (*Danaus plexippus* L.): Avian predation at five overwintering sites in Mexico. Science 204:847-851.

Caminade C, McIntyre KM, Jones AE. 2019. Impact of recent and future climate change on vector-borne diseases. Annals of the New York Academy of Sciences 1436:157-173.

Cardno, Inc. 2020. Nationwide Candidate Conservation Agreement for Monarch Butterfly on Energy and Transportation Lands: An integrated Candidate Conservation Agreement with Assurances (CCAA) and Candidate Conservation Agreement (CCA). Energy Resources Center at The University of Illinois at Chicago. 95 pp.

[CDC] Centers for Disease Control. 2018. Illnesses on the rise. https://www.cdc.gov/vitalsigns/vector-borne/index.html [Accessed 10 April 2020].

[CEC] Secretariat of the Commission for Environmental Cooperation. 2008. North American monarch conservation plan. Communications Department of the Center for Environmental Cooperation Secretariat.

Center for Biological Diversity, Center for Food Safety, Xerces Society, Brower L. 2014. Petition to protect the monarch butterfly (*Danaus plexippus plexippus*) under the Endangered Species Act. 159 pp.

Chevin LM, Lande R, Mace GM. 2010. Adaptation, plasticity, and extinction in a changing environment: Towards a predictive theory. PLOS Biology 8(4):e1000357.

Cinquini L, Crichton D, Mattmann C, Harney J, Shipman G, Wang F, Ananthakrishnan R, Miller

Pascoe S, Gonzalez E, Fiore S, Schweitzer R. 2014. The Earth System Grid Federation: An open infrastructure for access to distributed geospatial data. Future Generation Computer Systems 36:400-417.

Cockrell BJ, Malcolm SB, Brower LP. 1993. Time, temperature, and latitudinal constraints on the annual recolonization of eastern North America by the monarch butterfly. Pp. 233-251 in Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Cook BI, Ault TR, Smerdon JE. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances 1:e1400082 (8 pp.).

[CORDEX] Coordinated Regional Climate Downscaling Experiment. 2018. WCRP CORDEX [online]. http://www.cordex.org/ [Accessed 26 November 2018]

COSEWIC. 2010. COSEWIC assessment and status report on the monarch *Danaus plexippus* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. vii + 43 pp.

Courchamp F, Clutton-Brock T, Grenfell B. 1999. Inverse density dependence and the Allee effect. Trends in Ecology and Evolution 14:405-410.

Couture JJ, Serbin SP, Townsend PA. 2015. Elevated temperature and periodic water stress alter growth and quality of common milkweed (*Asclepias syriaca*) and monarch (*Danaus plexippus*) larval performance. Arthropod-Plant Interactions 9:149-161.

Crandall KA, Bininda-Emonds OR, Mace GM, Wayne RK. 2000. Considering evolutionary processes in conservation biology. Trends in Ecology & Evolution 15:290-295.

Crone EE, Pelton EM, Brown LM, Thomas CC, Schultz CB. 2019. Why are monarch butterflies declining in the West? Understanding the importance of multiple correlated drivers. Ecological Applications 29:e01975 (13 pp.).

Davis AK. 2009. Wing color of monarch butterflies (*Danaus plexippus*) in eastern North America across life stages: Migrants are redder than breeding and overwintering stages. Psyche 2009:1-5.

Davis AK, Chi J, Bradley C, Altizer S. 2012. The redder the better: Wing color predicts flight performance in monarch butterflies. PLOS ONE 7:e41323.

Decker LE, de Roode JC, Hunter MD. 2018. Elevated atmospheric concentrations of carbon dioxide reduce monarch tolerance and increase parasite virulence by altering the medicinal properties of milkweeds. Ecology Letters 21:1353-1363.

Deutsch CA, Tewksbury JJ, Tigchelaar M, Battisti DS, Merrill SC, Huey RB, Naylor RL. 2018. Increase in crop losses to insect pests in a warming climate. Science 361:916-919.

Diffenbaugh, NS, Swain DL, Touma D. 2015. Anthropogenic warming has increased drought risk in California. Proceedings of the National Academy of Sciences of the United States of America 112:3931-3936.

Dilts TE, Steele M, Black S, Craver D, Cruz E, Engler J, Jepsen S, Jones A, McKnight S, Pelton E, Taylor A, and Forister M. 2018. Western Monarch and Milkweed Habitat Suitability Modeling Project Version 2 – Maxent Model Outputs. Xerces Society/US Fish and Wildlife Service/University of Nevada Reno. Retrieved from: www.monarchmilkweedmapper.org/

Dilts TE, Steele MO, Engler JD, Pelton EM, Jepsen SJ, McKnight SJ, Taylor AR, Fallon CE, Black SH, Cruz EE, Craver DR, and Forister ML. 2019. Host plants and climate structure habitat associations of the western monarch butterfly. Frontiers in Ecology and Evolution 7:188 (17 pp.).

Dingle H, Zalucki MP, Rochester WA, Armijo-Prewitt T. 2005. Distribution of the monarch butterfly, *Danaus plexippus* (L.) (Lepidoptera: Nymphalidae), in western North America. Biological Journal of the Linnean Society 85:491-500.

DiTommaso A, Losey JE. 2003. Oviposition preference and larval performance of monarch butterflies (*Danaus plexippus*) on two invasive swallow-wort species. Entomologia Experimentalis et Applicata 108:205-209.

Douglas MR, Tooker JF. 2015. Large-scale deployment of seed treatments has driven rapid increase in use of neonicotinoid insecticides and preemptive pest management in U.S. field crops. Environmental Science and Technology 49:5088-5097.

Espeset AE, Harrison JG, Shapiro AM, Nice CC, Thorne JH, Waetjen DP, Fordyce JA, Forister ML. 2016. Understanding a migratory species in a changing world: Climatic effects and demographic declines in the western monarch revealed by four decades of intensive monitoring. Oecologia 181:819-830.

[FAO] Food and Agricultural Organization of the United Nations. 2017. The future of food and agriculture: Trends and challenges. Rome. 180 pp. Retrieved from: www.fao.org/publications/fofa.

Fernández-Haeger J, Jordano D, Zalucki MP. 2015. Monarchs across the Atlantic Ocean: What's happening on the other shore? Pp. 247-256 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Fisher A, Saniee K, van der Heide C, Griffiths J, Meade D, Villablanca F. 2018. Climatic niche model for overwintering monarch butterflies in a topographically complex region of California. Insects 9(4):167.

Flockhart DTT, Wassenaar LI, Martin TG, Hobson KA, Wunder MB, Norris DR. 2013. Tracking multi-generational colonization of the breeding grounds by monarch butterflies in eastern North America. Proceedings of the Royal Society B 280:20131087.

Flockhart DTT, Pichancourt J-B, Norris DR, Martin TG. 2015. Unravelling the annual cycle in a migratory animal: Breeding-season habitat loss drives population declines of monarch butterflies. Journal of Animal Ecology 84:155-165.

Flockhart DTT, Brower LP, Ramírez MI, Hobson KA, Wassenaar LI, Altizer S, Norris DR. 2017. Regional climate on the breeding grounds predicts variation in the natal origin of monarch butterflies overwintering in Mexico over 38 years. Global Change Biology 23:2565-2576.

Flores-Martínez JJ, Martínez-Pacheco A, Rendón-Salinas E, Rickards J, Sarkar S, Sánchez-Cordero V. 2019. Recent forest cover loss in the core zones of the Monarch Butterfly Biosphere Reserve in Mexico. Frontiers in Environmental Science 7:167 (10 pp.).

Forister ML, Cousens B, Harrison JG, Anderson K, Thorne JH, Waetjen D, Nice CC, De Parsia M, Hladik ML, Meese R, van Vliet H, Shapiro AM. 2016. Increasing neonicotinoid use and the declining butterfly fauna of lowland California. Biology Letters 12:20160475 (5 pp.).

Forsman A. 2014. Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion, and infection biology. Proceedings of the National Academy of Sciences of the United States of America 111:302-307.

Freedman MG, Dingle H, Tabuloc CA, Chiu JC, Yang LH, Zalucki MY. 2017. Non-migratory monarch butterflies, *Danaus plexippus* (L.), retain developmental plasticity and a navigational mechanism associated with migration. Biological Journal of the Linnean Society 123:265-278.

Freedman MG, Dingle H. 2018. Wing morphology in migratory North American monarchs: Characterizing sources of variation and understanding changes through time. Animal Migration 5:61-73.

Frey D, Schaffner A. 2004. Spatial and temporal pattern of monarch overwintering abundance in western North America. Pp. 167-176 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Funk WC, Forester BR, Converse SJ, Darst C, and Morey S. 2019. Improving conservation policy with genomics: A guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics 20:115-134.

Ghalambor CK, McKay JK, Carroll SP, Reznick DN. 2007. Adaptive versus non-adaptive phenotypic plasticity and the potential for contemporary adaptation in environments. Functional Ecology 21:394-407.

Grant TJ, Perry HR, Zalucki MP, Bradbury SP. 2018. Predicting monarch butterfly (*Danaus plexippus*) movement and egg-laying with a spatially-explicit agent-based model: The role of monarch perceptual range and spatial memory. Ecological Modelling 374:37-50.

Griffiths JL. 2014. Monarch butterflies overwintering in coastal California: Low site fidelity and high intersite movement. Master's Thesis. California Polytechnic State University, San Luis Obispo.

Griffiths J, Villablanca F. 2015. Managing monarch butterfly overwintering groves: Making room among the eucalyptus. California Fish and Game 101:40-50.

Groves CR, Jensen DB, Valutis LL, Redford KH, Shaffer ML, Scott JM, Baumgartner JV, Higgins JV, Beck MW, Anderson MG. 2002. Planning for biodiversity conservation: putting conservation science into practice. BioScience 526:499-512.

Guerra PA, Reppert SM. 2013. Coldness triggers northward flight in remigrant monarch butterflies. Current Biology 23:419-423.

Halsch CA, Code A, Hoyle SM, Fordyce JA, Baert N, Forister ML. 2020. Contamination of milkweeds across the agricultural, urban, and open spaces of low elevation Northern California. Frontiers in Ecology and Evolution 8:162.

Hanski I, Gilpin M, eds. 1997. Metapopulation Biology: Ecology, Genetics, and Evolution. London, UK: Academic Press. 512 pp.

Hay-Roe MM, Lamas G, Nation JL. 2007. Pre- and post-zygotic isolation and Haldane rule effects in reciprocal crosses of *Danaus erippus* and *Danaus plexippus* (Lepidoptera: Danainae), supported by differentiation of cuticular hydrocarbons, establish their status as separate species. Biological Journal of the Linnean Society 91:445-453.

Hendry AP, Kinnison MT, Heino M, Day T, Smith TB, Fitt G, Bergstrom CT, Oakeshott J,

application. Evolutionary Applications 4:159-183.

Herman WS, Tatar M. 2001. Juvenile hormone regulation of longevity in the migratory monarch butterfly. Proceedings of the Royal Society B-Biological Sciences 268:2509-2514.

Herman WS, Brower LP, Calvert WH. 1989. Reproductive tract development in monarch butterflies overwintering in California and Mexico. Journal of the Lepidopterists Society 43:50-58.

Hijmans RJ. 2017. raster: Geographic Data Analysis and Modeling. R package version 2.6-7. https://CRAN.R-project.org/package=raster [Accessed 24 April 2020].

Hoffmann AA, Sgro CM. 2011. Climate change and evolutionary adaptation. Nature 470:479-485.

Holderegger R, Kamm U, Gugerli F. 2006. Adaptive vs neutral genetic diversity: Implications for landscape genetics. Landscape Ecology 21:797-807.

Honey-Rosés J, Maurer M, Ramírez MI, Corbera E. 2018. Quantifying active and passive restoration in Central Mexico from 1986-2012: Assessing the evidence of a forest transition. Restoration Ecology 26:1180-1189.

Howard E, Davis AK. 2012. Mortality of migrating monarch butterflies from a wind storm on the shore of Lake Michigan, USA. Journal of Research on the Lepidoptera 45:49-54.

Howard E, Aschen H, Davis AK. 2010. Citizen science observations of monarch butterfly overwintering in the southern United States. Psyche: A Journal of Entomology 1-6.

Hutchings JA. 2015. Thresholds for impaired species recovery. Proceedings of the Royal Society B 282:20150654 (11 pp.)

Inamine H, Ellner SP, Springer JP, Agrawal AA. 2016. Linking the continental migratory cycle of the monarch butterfly to understand its population decline. Oikos 125:1081-1091.

[IELP] International Environmental Law Project and the Xerces Society. 2012. The Legal Status of Monarch Butterflies in California. 104 pp. IELP Report on Monarch Legal Status. Portland, OR: International Environmental Law Project and the Xerces Society.

[IPCC] Intergovernmental Panel on Climate Change. 2001. Climate change 2001: impacts, adaptation, and vulnerability. In McCarthy JJ, Canziani OF, Lear NA, Dokken DJ, White KS, eds. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press.

[IPCC] Intergovernmental Panel on Climate Change. 2000. Special report on emission scenarios. Nakicenovic N and Swart R, eds. Cambridge, UK: Cambridge University Press. 570 pp.

[IUCN] The IUCN Red List of Threatened Species. 2018. 2001 categories & criteria (version 3.1). http://www.iucnredlist.org/static/categories_criteria_3_1 [Accessed 4 October 2018]

James DG. 1993. Migration biology of the monarch butterfly in Australia. Pp. 189-200 in Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

James DG. 2019. A neonicotinoid insecticide at a rate found in nectar reduces longevity but not oogenesis in monarch butterflies, *Danaus plexippus* (L.). (Lepidoptera: Nymphalidae). Insects 10:276 (10 pp.).

Jepsen S, Black SH. 2015. Understanding and conserving the western North American monarch population. Pp. 147-156 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Jepsen S, Schweitzer DF, Young B, Sears N, Ormes M, Black SH. 2015. Conservation status and ecology of monarchs in the United States. NatureServe, Arlington, VA, and the Xerces Society for Invertebrate Conservation, Portland, OR. 36 pp.

Jepsen S, Pelton P, Black SH, Fallon C, Voight C. 2017. Protecting California's butterfly groves: Management guidelines for monarch butterfly overwintering habitat. vi + 32 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Jeschke P, Nauen R, Schindler M, Elbert A. 2011. Overview of the Status and Global Strategy for Neonicotinoids. Journal of Agricultural and Food Chemistry 59:2897-2908.

Journey North. 2008. Hurricanes: a fall migration hazard. https://journeynorth.org/tm/monarch/FallHazards_HurricaneIke.html [Accessed 25 June 2020].

Journey North. 2018. The Extraordinary Travels of a Tagged Monarch. https://journeynorth.org/tm/monarch/TagRecoveryVAa.html [Accessed 12 October 2018].

Kantola T, Tracy JL, Baum KA, Quinn MA, Coulson RN. 2019. Spatial risk assessment of eastern monarch butterfly road mortality during autumn migration within the southern corridor. Biological Conservation 231:150-160.

Karloski JM, Evans C. 2016. Seasonal influences upon and long-term trends in the length of the Atlantic hurricane season. Journal of Climate 29:273-292.

Kasten K, Stenoien C, Caldwell W, Oberhauser KS. 2016. Can roadside habitat lead monarchs on a route to recovery? Journal of Insect Conservation 20:1047-1057.

Keilsohn W, Narango DL, Tallamy DW. 2018. Roadside habitat impacts insect traffic mortality. Journal of Insect Conservation 22:183-188.

Kharin VV, Zwiers FW, Zhang X, Wehner M. 2013. Changes in temperature and precipitation extremes in the CMIP5 ensemble. Climatic Change 119:345-357.

Kindvall O. 1996. Habitat heterogeneity and survival in a bush cricket metapopulation. Ecology 77(1):207-214.

Knight A, Brower LP. 2009. The influence of eastern North American autumnal migrant monarch butterflies (*Danaus plexippus* L.) on continuously breeding resident monarch populations in southern Florida. Journal of Chemical Ecology 35:816-823.

Krischik V, Rogers M, Gupta G, Varshney A. 2015. Soil-applied imidacloprid translocates to ornamental flowers and reduces survival of adult *Coleomegilla maculata*, *Harmonia axyridis*, and *Hippodamia convergens* lady beetles, and larval *Danaus plexippus* and *Vanessa cardui* butterflies. PLOS One 10:e0119133 (22 pp.).

Krishnan N, Zhang Y, Bidne KG, Hellmich RL, Coats JR, Bradbury SP. 2020. Assessing field-scale risks of foliar insecticide applications to monarch butterfly (*Danaus plexippus*) larvae. Environmental Toxicology and Chemistry 39:923-941.

Lankau R, Jørgensen, PS, Harris DJ, Sih A. 2011. Incorporating evolutionary principles into environmental management and policy. Evolutionary Applications 4:315-325.

Lark TJ, Salmon JM, Gibbs HK. 2015. Cropland expansion outpaces agricultural and biofuel policies in the United States. Environmental Research Letters 10:044003.

Larsen KJ, Lee RE. 1994. Cold tolerance including rapid cold-hardening and inoculative freezing of fall migrant monarch butterflies in Ohio. Journal of Insect Physiology 40:859-864.

Lemoine, NP. 2015. Climate change may alter breeding ground distributions of eastern migratory monarchs (*Danaus plexippus*) via range expansion of *Asclepias* host plants. PLOS ONE 10:e0118614.

Leong KLH. 1990. Microenvironmental factors associated with the winter habitat of the monarch butterfly (Lepidoptera, Danaidae) in central California. Annals of the Entomological Society of America 83:906-910.

Leong KLH. 1999. Restoration of an overwintering grove in Los Osos, San Luis County, California. Pp. 211-18 in Hoth J, Merina L, Oberhauser K, Pisantry I, Price A, Wilkinson T, eds. The 1997 North American Conference on the Monarch Butterfly, Montreal, Canada: Commission for Environmental Cooperation.

Leong KLH, O'Brien E, Lowerisen K, Colleran M. 1995. Mating activity and status of overwintering monarch butterflies (Lepidoptera, Danaidae) in central California. Annals of the Entomological Society of America 88:45-50.

Leong KLH, Sakai WH, Bremer W, Feuerstein D, Yoshimura G. 2004. Analysis of the pattern of distribution and abundance of monarch overwintering sites along the California coastline. Pp. 177-185 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Littel JS, McKenzie D, Peterson DL, Westerling AL. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications 19:1003-1021.

López García J. 2011. Deforestation and forest degradation in the Monarch Butterfly Biosphere Reserve, Mexico, 2003-2009. Journal of Maps 7(1):626-633.

Lotts K, Naberhaus T (coordinators). 2017. Butterflies and moths of North America. https://www.butterfliesandmoths.org/species/Danaus-plexippus [Accessed 19 October 2018].

Lyons JI, Pierce AA, Barribeau SM, Sternberg ED, Mongue AJ, de Roode JC. 2012. Lack of genetic differentiation between monarch butterflies with divergent migration destinations. Molecular Ecology 21:3433-3444.

[MAFWA] Midwest Association of Fish and Wildlife Agencies. 2018. Mid-America Monarch Conservation Strategy 2018-2038. http://www.mafwa.org/wp-content/uploads/2018/07/MAMCS_June2018_Final.pdf [Accessed 1 November 2018].

Malcolm SB, Zalucki MP. 1993. The monarch butterfly: Research and conservation. Pp. 3-8 in Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Malcolm SB, Cockrell BJ, Brower LP. 1987. Monarch butterfly voltinism: Effects of temperature constraints at different latitudes. Oikos 49:77-82.

Malcolm SB, Cockrell BJ, Brower LP. 1993. Spring recolonization of eastern North America by the monarch butterfly: Successive brood or single sweep migration? Pp. 253-267 in Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Mangal M, Tier C. 1993. A simple direct method for finding persistence times of populations and application to conservation problems. Proceedings of the National Academy of Sciences of the United States of America 90:1083-1086.

Marcar N, Crawford D, Leppert P, Jovanovic T, Floyd R, Farrow R. 1995. Trees for saltland: A guide to selecting native species for Australia. Victoria, Australia: CSIRO Press.

Masters AR, Malcolm SB, Brower LP. 1988. Monarch butterfly (*Danaus plexippus*) thermoregulatory behavior and adaptations for overwintering in Mexico. Ecology 69:458-467.

McCord JW, Davis AK. 2010. Biological observations of monarch butterfly behavior at a migratory stopover site: Results from a long-term tagging study in coastal South Carolina. Journal of Insect Behavior 23:405-418.

McCoshum SM, Andreoli SL, Stenoien CM, Oberhauser KS, Baum KA. 2016. Species distribution models for natural enemies of monarch butterfly (*Danaus plexippus*) larvae and pupae: Distribution patterns and implications for conservation. Journal of Insect Conservation 20:223–237.

Miller NG, Wassenaar LI, Hobson KA, Norris DR. 2012. Migratory connectivity of the monarch butterfly (*Danaus plexippus*): Patterns of spring re-colonization in eastern North America. PLOS One 7:e31891.

Monarch Watch. 2020. Monarch Population Status [online]. https://monarchwatch.org/blog/2020/03/13/monarch-population-status-42/ [Accessed 9 September 2020].

Mora Alvarez BX, Carrera-Treviño R, Hobson KA. 2019. Mortality of monarch butterflies (*Danaus plexippus*) at two highway crossing "hotspots" during autumn migration in Northeast Mexico. Frontiers in Ecology and Evolution 7:273 (10 pp.).

Moritz, C. 2002. Strategies to protect biological diversity and the evolutionary processes that sustain it. Systematic Biology 51:238-254.

Morris GM, Kline C, Morris SM. 2015. Status of *Danaus plexippus* population in Arizona. Journal of the Lepidopterists Society 69(2):91-107.

Moskowitz D, Moskowitz J, Moskowitz S, Moskowitz H. 2001. Notes on a large dragonfly and butterfly migration in New Jersey. Northeastern Naturalist 8:483-490.

Nagano CD, Sakai WH, Malcolm SB, Cockrell BJ, Donahue JP, and Brower LP. 1993. Spring migration of monarch butterflies in California. Pp. 219–232 in Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Nail KR, Oberhauser KS. 2015. Monarchs in a changing climate: An overview. Pp. 95-98 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Nail KR, Stenoien C, Oberhauser KS. 2015a. Immature monarch survival: Effects of site characteristics, density, and time. Annals of the Entomological Society of America 108:680-690.

Nail KR, Batalden RV, Oberhauser KS. 2015b. What's too hot and what's too cold?: Lethal and sublethal effects of extreme temperatures on developing monarchs. Pp. 99-108 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

NatureServe. 2013. Rank a species EO. http://help.natureserve.org/biotics/Content/Record_Management/Element_Occurrence/EO_Rank_a_species_EO.htm [Accessed 4 October 2018]

Nicotra, AB, Beever EA, Robertson AL, Hofmann GE, O'Leary J. 2015. Assessing the

change. Conservation Biology 29:1268-1278.

[NOAA] National Oceanic and Atmospheric Administration. 2010. Total number of hurricane strikes by counties/parishes/boroughs, 1900-2010 [online]. https://www.nhc.noaa.gov/climo/images/strikes_us.jpg [Accessed 25 June 2020].

[NOAA] National Oceanic and Atmospheric Administration. 2018. U.S. Climate Extremes Index (CEI): Graph [online]. https://www.ncdc.noaa.gov/extremes/cei/graph [Accessed 14 November 2018].

Oberhauser, KS. 1997. Fecundity, lifespan and egg mass in butterflies: Effects of male-derived nutrients and female size. Functional Ecology 11:166-175.

Oberhauser K, Peterson AT. 2003. Modeling current and future potential wintering distributions of eastern North American monarch butterflies. Proceedings of the National Academy of Sciences of the United States of America 100:14063-14068.

Oberhauser KS, Manweiler SA, Lelich R, Blank M, Batalden RV, de Anda A. 2009. Impacts of ultra-low volume resmethrin applications on non-target insects. Journal of the American Mosquito Control Association 25:83-93.

Oberhauser KS, Anderson M, Anderson S, Caldwell W, De Anda A, Hunter M, Kaiser MC, Solensky MJ. 2015. Lacewings, wasps, and flies—oh my: Insect enemies take a bite out of monarchs. Pp. 71-82 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Oberhauser K, Wiederholt R, Diffendorfer JE, Semmens D, Ries L, Thogmartin WE, López-Hoffman L, Semmens B. 2017a. A trans-national monarch butterfly population model and implications for regional conservation priorities. Ecological Entomology 42:51-60.

Oberhauser K, Elmquist D, Manuel Perilla-Lopez J, Gebhard I, Lukens L, Stireman J. 2017b. Tachinid fly (Diptera: Tachinidae) parasitoids of *Danaus plexippus* (Lepidoptera: Nymphalidae). Annals of the Entomological Society of America 1-9. DOI: 10.1093/aesa/sax048

Ofori BY, Stow AJ, Baumgartner JB, Beaumont LJ. 2017. Influence of adaptive capacity on the outcome of climate change vulnerability assessment. Scientific Reports 7:12979. DOI: 10.1038/s41598-017-13245-y

Olaya-Arenas P, Kaplan I. 2019. Quantifying pesticide exposure risk for monarch caterpillars on milkweeds bordering agricultural land. Frontiers in Ecology and Evolution 7:223.

Oliver T, Roy DB, Hill JK, Brereton T, Thomas CD. 2010. Heterogeneous landscapes promote population stability. Ecology Letters 13:473-484.

Paine TD, Millar JG. 2002. Insect pests of eucalyptus in California: Implications of managing invasive species. Bulletin of Entomological Research 92:147-151.

Paini DR, Sheppard AW, Cook DC, De Barro PJ, Worner SP, Thomas MB. 2016. Global threat to agriculture from invasive species. Proceedings of the National Academy of Sciences of the United States of America 113:7575-7579.

Parsons JA. 1965. A digitalis-like toxin in monarch butterfly *Danaus plexippus* L. Journal of Physiology-London 178:290-304.

Pelton E, Jepsen S, Schultz C, Fallon C, Black SH. 2016. State of the monarch butterfly overwintering sites in California. Portland, OR: The Xerces Society for Invertebrate Conservation. vi + 40 pp.

Pelton E, McKnight S, Fallon C, Code A, Hopwood J, Hoyle S, Jepsen S, Black SH. 2018. Managing for monarchs in the West: Best management practices for conserving the monarch butterfly and its habitat. Portland, OR: The Xerces Society for Invertebrate Conservation. vi + 106 pp.

Pelton EM, Schultz CB, Jepsen SJ, Black SH and Crone EE. 2019. Western monarch population plummets: Status, probable causes, and recommended conservation actions. Frontiers in Ecology and Evolution 7:258.

Petersen LR, Nasci RS, Beard CB, Massung RF. 2016. Emerging vector-borne diseases in the United States: What is next, and are we prepared? Pp. 258-284 in Mack A (rapporteur). Global Health Impacts of Vector-Borne Diseases: Workshop Summary. Washington, DC: The National Academies Press.

Pfeiler E, Nazario-Yepiz NO, Pérez-Gálvez F, Chávez-Mora CA, Laclette MRL, Rendón-Salinas E, Markow TA. 2016. Population genetics of overwintering monarch butterflies, *Danaus plexippus* (Linnaeus), from central Mexico inferred from mitochondrial DNA and microsatellite markers. Journal of Heredity 108:163-175.

Pierce AA, Zalucki MP, Bangura M, Udawatta M., Kronforst MR, Altizer S, Haeger JF, de Roode JC. 2014a. Serial founder effects and genetic differentiation during worldwide range expansion of monarch butterflies. Proceedings of the Royal Society B 281:20142230.

Pierce AA, de Roode JC, Altizer S, Bartel RA. 2014b. Extreme heterogeneity in parasitism despite low population genetic structure among monarch butterflies inhabiting the Hawaiian Islands. PLOS ONE 9:e100061.

Pierce AA, Altizer S, Chamberlain NL, Kronforst MR, de Roode JC. 2015. Unraveling the mysteries of monarch migration and global dispersal through molecular genetic techniques. Pp. 257-267 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Pierce DW, Das T, Cayan DR, Maurer EP, Miller NL, Bao Y, Kanamitsu M, Yoshimura K, Snyder MA, Sloan LC, Franco G, Tyree M. 2013. Probabilistic estimates of future changes in California temperature and precipitation using statistical and dynamical downscaling. Climate Dynamics 40:839-856.

Pierce DW, Kalansky JF, Cayan DR. 2018. Climate, drought, and sea level rise scenarios for California's fourth climate change assessment. California's Fourth Climate Change Assessment, California Energy Commission. Publication Number: CNRA-CEC-2018-006.

Pleasants JM. 2017. Milkweed restoration in the Midwest for monarch butterfly recovery: Estimates of milkweeds lost, milkweeds remaining and milkweeds that must be added to increase the monarch population. Insect Conservation and Diversity 10:42-53.

Pleasants JM, Oberhauser KS. 2013. Milkweed loss in agricultural fields because of herbicide use: Effect on the monarch butterfly population. Insect Conservation and Diversity 6:135-144.

Popp J, Pető K, Nagy J. 2013. Pesticide productivity and food security. A review. Agronomy for Sustainable Development 33:243-255.

Project Monarch Health. 2016. Monarch Health News: 2016 Monitoring Summary. 2 pp.

Puget Sound Stream Benthos. 2018. Benthic index of biotic integrity (B-IBI, BIBI) variations. https://www.pugetsoundstreambenthos.org/BIBI-Scoring-Types.aspx [Accessed 4 October 2018].

Ramírez MI, Sáenz-Romero C, Rehfeldt G, Salas-Canela L. 2015. Threats to the availability of overwintering habitat in the Monarch Butterfly Biosphere Reserve: Land use and climate change. Pp. 157-168 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Ramsay GW. 1964. Overwintering swarms of the monarch butterfly (*Danaus plexippus* (L.)) in New Zealand. New Zealand Entomologist 3:10-16.

Redford KH, Amato G, Baillie J, Beldomenico P, Bennett EL, Clum N, Cook R, Fonseca G, Hedges S, Launay F, Lieberman S, Mace GM, Murayama A, Putnam A, Robinson JG, Rosenbaum H, Sanderson EW, Stuart SN, Thomas P, Thorbjarnarson J. 2011. What does it mean to conserve a (vertebrate) species? BioScience 61:39-48.

Rendón-Salinas E, Fajardo-Arroyo A, Tavera-Alonso G. 2015. Forest surface occupied by monarch butterfly hibernation colonies in December 2014. World Wildlife Fund. 4 pp.

Ries L, Neupane N, Baum KA, Kipkin EF. 2018. Flying through hurricane central: Impacts of hurricanes on migrants with a focus on monarch butterflies. Animal Migration 5:94-103.

Rohweder JJ, Thogmartin WE. 2016. Monarch Conservation Planning Tools. U.S. Geological Survey, Upper Midwest Environmental Sciences Center, La Crosse, Wisconsin, USA.

RStudio Team. 2015. RStudio: Integrated Development for R. RStudio, Inc., Boston, MA http://www.rstudio.com/ [Accessed 24 April 2020].

Ruckelhaus M, McElhany P, Ford MJ. 2002. Recovering species of conservation concern: Are populations expendable? Pp. 305-329 in Kareiva P, Levin S., eds. The Importance of Species: Perspectives on Expendability and Triage. Princeton, NJ: Princeton University Press.

Sáenz-Romero C, Rehfeldt GE, Duval P, Lindig-Cisneros RA. 2012. *Abies religiosa* habitat prediction in climatic change scenarios and implications for monarch butterfly conservation in Mexico. Forest Ecology and Management 275:98-106.

Sakai WH, Calvert WC. 1991. Statewide monarch butterfly management plan for the state of

Department of Parks and Recreation, Sacramento, CA.

Sangle PM, Satpute SB, Khan FS, Rode NS. 2015. Impact of climate change on insects. Trends in Bioscience 8:3579-3582.

Satterfield DA, Villablanca FX, Maerz JC, Altizer S. 2016. Migratory monarchs wintering in California experience low infection risk compared to monarchs breeding year-round on non-native milkweed. Integrative and Comparative Biology 56:343-52.

Satterfield DA, Maerz JC, Hunter MD, Flockhart DTT, Hobson KA, Norris DR, Streit H, de Roode JC, Altizer S. 2018. Migratory monarchs that encounter resident monarchs show life-history differences and higher rates of parasite infection. Ecology Letters 21:1670-1680.

Saunders SP, Ries L, Neupane N, Ramírez MI, García-Serrano E, Rendón-Salinas E, Zipkin EF. 2019. Multiscale seasonal factors drive the size of winter monarch colonies. Proceedings of the National Academy of Sciences of the United States of America 116:8609-8614.

Schmidt BC. 2003. *Danaus plexippus* [online]. Entomology Collection. University of Alberta EH Strickland Entomological Museum. http://entomology.museums.ualberta.ca/searching_species_details.php?s=2854 [Accessed 06 June 2018].

Schultz CB, Brown LM, Pelton E, Crone EE. 2017. Citizen science monitoring demonstrates dramatic declines of monarch butterflies in western North America. Biological Conservation 214:343-346.

Scott JA. 1986. The butterflies of North America: A natural history and field guide. Stanford, CA: Stanford University Press.

Semmens BX, Semmens DJ, Thogmartin WE, Wiederholt R, López-Hoffman L, Diffendorfer JE, Pleasants JM, Oberhauser KS, Taylor OR. 2016. Quasi-extinction risk and population targets for the eastern, migratory population of monarch butterflies (*Danaus plexippus*). Scientific Reports 6:23265.

Sgro CM, Lowe AJ, Hoffmann AA. 2011. Building evolutionary resilience for conserving biodiversity under climate change. Evolutionary Applications 4:326-337.

Shaffer ML, Stein BA. 2000. Safeguarding our precious heritage. Pp. 301-321 in Stein BA,

New York: Oxford University Press.

Sharma HC, Prabhakar CS. 2014. Impact of climate change on pest management and food security. Pp. 23-36 in Abrol DP, ed. Integrated Pest Management: Current Concepts and Ecological Perspective. London, UK: Academic Press, Elsevier.

Shephard JM, Hughes JM, Zalucki MP. 2002. Genetic differentiation between Australian and North American populations of the monarch butterfly *Danaus plexippus* (L.) (Lepidoptera: Nymphalidae): An exploration using allozyme electrophoresis. Biological Journal of the Linnean Society 75:437-452.

Sillman J, Kharim VV, Zwiers FW, Zhang X, Bronaugh D. 2013. Climate extremes indices in the CMIP5 multimodel ensemble: Part 2. Future climate projections. Journal of Geophysical Research: Atmospheres 118:2473-2493.

Slayback DA, Brower LP. 2007. Further aerial surveys confirm the extreme localization of overwintering monarch butterfly colonies in Mexico. American Entomologist 53:146-149.

Sleeter BM. 2016. Land cover trends project: Central Valley report [online]. U.S. Geological Survey, Western Geographic Science Center, Menlo Park, CA. https://landcovertrends.usgs.gov/west/eco7Report.html [Accessed 18 June 2018].

Sleeter BM, Wilson TS, Sharygin E, Sherba JT. 2017. Future scenarios of land change based on empirical data and demographic trends. Earth's Future 5:1068-1083.

Smith DAS, Lushai G, Allen JA. 2005. A classification of *Danaus* butterflies (Lepidoptera: Nymphalidae) based upon data from morphology and DNA. Zoological Journal of the Linnean Society 144:191-212.

Smithers CN. 1973. A note on natural enemies of *Danaus plexippus* (L.) (Lepidoptera: Nymphalidae) in Australia. Australian Entomological Magazine 1:37-40.

Sohl TL, Sayler KL, Bouchard MA, Reker RR, Freisz AM, Bennett SL, Sleeter BM, Sleeter RR, Wilson T, Soulard C, Knuppe M, Van Hofwegen T. 2018. Conterminous United States Land Cover Projections - 1992 to 2100: U.S. Geological Survey data release, https://doi.org/10.5066/P95AK9HP.

Solensky MJ. 2004. Overview of monarch migration. Pp. 79-83 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Solensky MJ, Oberhauser KS. 2009. Sperm precedence in monarch butterflies (*Danaus plexippus*). Behavioral Ecology 20:328-334.

Southwest Monarch Study Inc. 2018. Fall migration south [online]. https://www.swmonarchs.org/migration-map-south.php [Accessed 16 August 2018].

Spielman D, Brook BW, Frankham R. 2004. Most species are not driven to extinction before genetic factors impact them. Proceedings of the National Academy of Sciences of the United States of America 101(42):15261-15264.

State of California. 2018. AB-2421 Wildlife Conservation Board: Monarch Butterfly and Pollinator Rescue Program. https://leginfo.legislature.ca.gov/faces/billTextClient.xhtml?bill_id=201720180AB2421 [Accessed 2 July 2020].

Stenoien C, Nail KR, Zalucki JM, Parry H, Oberhauser KS, Zalucki MP. 2016. Monarchs in decline: A collateral landscape level effect of modern agriculture. Insect Science 25:528–541.

Sternberg ED, Li H, Wang R, Gowler C, de Roode JC. 2013. Patterns of host-parasite adaptation in three populations of monarch butterflies infected with a naturally occurring protozoan disease: Virulence, resistance, and tolerance. American Naturalist 182:E235-E248.

Stevens SR, Frey DF. 2010. Host plant pattern and variation in climate predict the location of natal grounds for migratory monarch butterflies in western North America. Journal of Insect Conservation 14:731-744.

Svancara LK, Abatzoglou JT, Waterbury B. 2019. Modeling current and future potential distributions of milkweeds and the monarch butterfly in Idaho. Frontiers in Ecology and Evolution 7:168 (21 pp.).

Swain DL, Langenbrunner B, Neelin JD, Hall A. 2018. Increasing precipitation volatility in twenty-first century California. Nature Climate Change 8:427-433

Talla V, Pierce AA, Adams KL, de Man TJB, Nallu S, Villablanca FX, Kronforst MR, and de Roode JC. 2020. Genomic evidence for gene flow between monarchs with divergent migratory phenotypes and flight performance. Molecular Ecology 29:2567-2582.

Tampa Bay Climate Science Advisory Panel. 2015. Recommended projection of sea level rise in the Tampa Bay region. 11 pp. https://www.wucaonline.org/assets/pdf/pubs-csap-sea-level-rise.pdf [Accessed 19 June 2018].

Taylor KE, Stouffer RJ, Meehl GA. 2012. An overview of CMIP5 and the experiment design. Bulletin of the American Meteorological Society 93:485-498.

Tenger-Trolander A, Lu W, Noyes M, Kronforst MR. 2019. Contemporary loss of migration in monarch butterflies. Proceedings of the National Academy of Sciences of the United States of America 116(29):14671-14676.

Teplitsky C, Robinson MR, Merilä J. 2014. Evolutionary potential and constraints in wild populations. Pp. 190-208 in Charmantier A, Garant D, and Kruuk LEB, eds. Quantitative Genetics in the Wild. Oxford, UK: Oxford University Press.

Thogmartin WE, Diffendorfer JE, López-Hoffman L, Oberhauser K, Pleasants J, Semmens BX, Semmens D, Taylor OR, Wiederholt R. 2017a. Density estimates of monarch butterflies overwintering in central Mexico. PeerJ 5:e3221.

Thogmartin W, Wiederholt R, Oberhauser K, Drum R, Diffendorfer J, Altizer S, Taylor O, Pleasants J, Semmens D, Semmens B, Erickson R, Libby K, López-Hoffman L. 2017b. Monarch butterfly population decline in North America: Identifying the threatening processes. Royal Society Open Science 4:170760.

Thogmartin WE, López-Hoffman L, Rohweder J, Diffendorfer J, Drum R, Semmens D, Black S, Caldwell I, Cotter D, Drobney P, Jackson L, Gale M, Helmers D, Hilburger S, Howard E, Oberhauser K, Pleasants J, Semmens B, Taylor O, Wiederholt R. 2017c. Restoring monarch butterfly habitat in the Midwestern US: 'All hands on deck.' Environmental Research Letters 12: 074005.

Tuskes PM, Brower LP. 1978. Overwintering ecology of monarch butterfly, *Danaus plexippus* L., in California. Ecological Entomology 3:141-153.

Urquhart FA, Urquhart NR. 1977. Overwintering areas and migratory routes of monarch butterfly (*Danaus p. plexippus*, Lepidoptera Danaidae) in North America, with special reference to western population. Canadian Entomologist 109:1583-1589.

Urquhart FA, Urquhart NR. 1978. Autumnal migration routes of the eastern population of monarch butterfly (*Danaus p. plexippus* L.; Danaidae; Lepidoptera) in North America to the overwintering site in the Neovolcanic Plateau of Mexico. Canadian Journal of Zoology 56:1759-1764.

[USDA] U.S. Department of Agriculture (Fernandez-Cornejo J, Nehring R, Osteen C, Wechsler S, Martin A, Vialou A). 2014. Pesticide Use in U.S. Agriculture: 21 Selected Crops, 1960-2008. EIB-124, U.S. Department of Agriculture, Economic Research Service. 86 pp.

[USDA] U.S. Department of Agriculture. 2016. America's Diverse Family Farms: 2016 Edition. EIB-164, U.S. Department of Agriculture, Economic Research Service. 16 pp.

[USDA] U.S. Department of Agriculture. 2018. Recent Trends in GE Adoption. [online]. https://www.ers.usda.gov/data-products/adoption-of-genetically-engineered-crops-in-the-us/recent-trends-in-ge-adoption.aspx [Accessed 28 June 2018].

[USDA] U.S. Department of Agriculture. 2020. USDA Agricultural Projections to 2029: Interagency Agricultural Projections Committee. Long-term Projections Report OCE-2020-1. 114 pp.

[USDOT] U.S. Department of Transportation. 2020a. System Mileage Within the United States. Bureau of Transportation Statistics. https://www.bts.gov/content/system-mileage-within-united-states [Accessed 24 April 2020].

[USDOT] U.S. Department of Transportation. 2020b. Class 1 Railroad System Mileage and Ton-miles of Freight: 1960, 1965, 1970-2015. Bureau of Transportation Statistics. https://www.bts.gov/content/class-i-railroad-system-mileage-and-ton-miles-freight-1960-1965-1970-2015 [Accessed 24 April 2020].

[USEPA] U.S. Environmental Protection Agency (Authors: Atwood D, and Paisley-Jones C). 2017 Pesticides Industry Sales and Usage: 2008–2012 Market Estimates. 32 pp.

[USGS NAWQA] U.S. Geological Survey National Water-Quality Assessment. 2017. Pesticide national synthesis project [online]. https://water.usgs.gov/nawqa/pnsp/ [Accessed 7 February 2018].

[USFWS and NOAA] U.S. Fish & Wildlife Service and National Oceanic and Atmospheric

Decisions. 68 FR 15100-15115.

Van den Berg H, Zaim M, Yadav RS, Soares A, Ameneshewa B, Mnzava A, Hii J, Dash AP, Ejov M. 2012. Global trends in the use of insecticides to control vector-borne diseases. Environmental Health Perspectives 120:577-582.

Van Hook T. 1996. Monarch butterfly mating ecology at a Mexican overwintering site: Proximate causes of non-random mating. Dissertation. University of Florida. 259 pp.

Vane-Wright RI. 1993. The Columbus hypothesis: An explanation for the dramatic 19th century range expansion of the monarch butterfly. Pp. 179-187 in Malcolm SB, Zalucki MP, eds.

Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Vidal O, Rendón-Salinas E. 2014. Dynamics and trends of overwintering colonies of the monarch butterfly in Mexico. Biological Conservation 180:165-175

Vidal O, López-García J, Rendón-Salinas E. 2014. Trends in deforestation and forest degradation after a decade of monitoring in the Monarch Butterfly Biosphere Reserve in Mexico. Conservation Biology 28:177-186.

Villareal TA. 2015. An assessment of the size of the monarch butterfly release industry in 2014. PowerPoint presentation. 10 pp.

Voorhies KJ, Szymanski J, Nail KR, Fidino M. 2019. A method to project future impacts from threats and conservation on the probability of extinction for North American migratory monarch (*Danaus plexippus*) populations. Frontiers in Ecology and Evolution 7:384 (13 pp.).

[WAFWA] Western Association of Fish and Wildlife Agencies. 2019. Western monarch butterfly conservation plan 2019-2069. https://www.wafwa.org/Documents%20and%20Settings/37/Site%20Documents/Committees/Monarch/WAFWA%20Monarch%20Conservation%20Plan.pdf [Accessed 22 April 2020]. 118 pp.

Walker JJ. 1914. The geographical distribution of *Danaida plexippus*, L. (*Danais Archippus*, F.) with especial reference to its recent migration. The Entomologist's Monthly Magazine 50:181-193, 224-237, 249-250.

Warren AD, Davis KJ, Stangeland EM, Pelham JP, Grishin NV. 2013. Illustrated lists of American butterflies [online]. http://www.butterfliesofamerica.com/L/t/Danaus_plexippus_a.htm [Accessed 29 May 2018].

Wassenaar LI, Hobson KA. 1998. Natal origins of migratory monarch butterflies at wintering colonies in Mexico: New isotopic evidence. Proceedings of the National Academy of Sciences of the United States of America 95:15436-15439.

Waterbury B, Potter A. 2018. Integrating strategic conservation approaches for the monarch butterfly in the State Wildlife Action Plans of Idaho and Washington. Final report prepared for the U.S. Fish & Wildlife Service. 79 pp.

Weiss SB, Rich PM, Murphy DD, Calvert WH, Ehrlich PR. 1991. Forest canopy structure at overwintering monarch butterfly sites: Measurements with hemispherical photography. Conservation Biology 5:165-175.

Wells H, Wells PH, Cook P. 1990. The importance of overwinter aggregation reproductive success of monarch butterflies (*Danaus plexippus* L.). Journal of Theoretical Biology 147:115-131.

Wells H, Wells PH, and Rogers SH. 1993. Is multiple mating an adaptive feature of monarch butterfly overwintering aggregation? Malcolm SB, Zalucki MP, eds. Biology and Conservation of the Monarch Butterfly. Natural History Museum of Los Angeles County, Science Series 38.

Williams AP, Seager R, Abatzoglou JT, Cook BI, Smerdon JE, Cook ER. 2015. Contribution of anthropogenic warming to California drought during 2012–2014. Geophysical Research Letters 42:6819-6828.

Williams AP, Cook ER, Smerdon JE, Cook BI, Abatzoglou JT, Bolles K, Baek SH, Badger AM, Livneh B. 2020. Large contribution from anthropogenic warming to an emerging North American megadrought. Science 368:314-318.

Williams EH, Brower LP. 2015. Microclimatic protection of monarch butterflies provided by Mexico's high elevation Oyamel fir forests: A review. Pp. 109-116 in Oberhauser KS, Nail KR, Altizer SM, eds. Monarchs in a Changing World: Biology and Conservation of an Iconic Insect. Ithaca, NY: Cornell University Press.

Wolfe DW, Ziska L, Petzoldt C, Seaman A, Chase L, Hayhoe K. 2008. Projected change in climate thresholds in the Northeastern U.S.: implications for crops, pests, livestock, and farmers. Mitigation and Adaptation Strategies for Global Change 13:555–575.

Xerces Society for Invertebrate Conservation. 2020. The Xerces Society Western Monarch Thanksgiving Count 1997-2019 [online]. https://www.westernmonarchcount.org/wp-content/uploads/2020/01/WMTC-Data-1997-2019_1.14.2020_v1.pdf [Accessed 22 April 2020].

Yang LH, Ostrovsky D, Rogers MC, Welker JM. 2016. Intra-population variation in the natal origins and wing morphology of overwintering western monarch butterflies *Danaus plexippus*. Ecography 39:998-1007.

York HA, Oberhauser KS. 2002. Effects of duration and timing of heat stress on monarch butterfly (*Danaus plexippus*) (Lepidoptera: Nymphalidae) development. Journal of the Kansas Entomological Society 75:290-298.

Zackay A. 2007. Random genetic drift and gene fixation. [online] https://www.metabolic-

21 June 2018].

Zalucki MP. 1982. Temperature and rate of development in *Danaus-plexippus* L and *D. chrysippus* L (Lepidoptera, Nymphalidae). Journal of the Australian Entomological Society 21:241-246.

Zalucki MP, Lammers JH. 2010. Dispersal and egg shortfall in monarch butterflies: What happens when the matrix is cleaned up? Ecological Entomology 35:84-91.

Zalucki MP, Rochester WA. 1999. Estimating the effect of climate on the distribution and abundance of the monarch butterfly, *Danaus plexippus* (L.): A tale of two continents. Pp. 150-63 in Hoth J, Merino L, Oberhauser K, Pisantry I, Price S, and Wilkinson T, eds. The 1997 North American Conference on the Monarch Butterfly. Montreal: Commission for Environmental Cooperation.

Zalucki MP, Rochester WA. 2004. Spatial and temporal population dynamics of monarch down under: Lessons for North America. Pp. 219-228 in Oberhauser KS, Solensky MJ, eds. The Monarch Butterfly: Biology and Conservation. Ithaca, NY: Cornell University Press.

Zagorski ME. 2016. Long live the king? A GIS analysis of climate change's impact on the future wintering range and economy of the monarch butterfly (*Danaus plexippus*) in Mexico. Latin American, Caribbean, and Latino Studies Senior Thesis, Gettysburg College. 55 pp.

Zhan S, Zhang W, Niitepold K, Hsu J, Haeger JF, Zalucki MP, Altizer S, de Roode JC, Reppert SM, Kronforst MR. 2014. The genetics of monarch butterfly migration and warning colouration. Nature 514:317-321.

Appendices

Appendix 1: Taxonomy

Appendix 2: Methods

1. **Updates to Voorhies et al. 2019 model**
2. **Inputs to model**
3. **Other threats and catastrophic events considered**
4. **Future scenarios**
5. **Climate change projections**

Appendix 3: Additional results

1. **Percent change in area and average number of days above 38°C and 42°C**
2. **Projected area and average number of days >38˚C and 42°C under RCP 8.5**
3. **pE over time under current and future scenarios**

Appendix 1. Taxonomy

At the time that the monarch butterfly (*Danaus plexippus plexippus*) was petitioned to be listed under the Endangered Species Act of 1973 (Center for Biological Diversity et al. 2014), the petition noted that there were six recognized subspecies of *Danaus plexippus* (*plexippus*, *megalippe*, *nigrippus*, *tobagi*, *portoricensis*, and *leucogyne*; Warren et al. 2013). However, examination of the literature and contact with a butterfly taxonomist, suggest there are only 2 or 3 subspecies, and that the subspecies concept for monarch butterflies is not currently rigorously defined.

In 2005, Smith and colleagues published their findings on *Danaus* taxonomy. They classified *Danaus plexippus* as having only two subspecies: *plexippus* and *megalippe*. *Danaus plexippus plexippus* is the subspecies that resides throughout most of North America, and throughout islands in the Pacific Ocean. *Danaus plexippus megalippe* is non-migratory and resides in parts of the southern U.S., the Caribbean, and Central and South America. They suggest that *tobagi*, *portoricensis*, and *leucogyne* may be color variants of *Danaus plexippus megalippe*, rather than separate subspecies. However, they do not comment on *Danaus plexippus nigrippus* (a potential subspecies that is non-migratory and found in parts of South America). In communications with butterfly taxonomy expert, Jonathan Pelham (Curatorial Associate [Lepidoptera] at the University of Washington Burke Museum), he agrees with the Smith et al. (2005) findings.

The potential third subspecies, *nigrippus*, was mentioned in a study where it was shown to be a different species than the South American-residing southern monarch (*Danaus erippus*; Hay-Roe et al. 2007). However, it is unclear whether any work has defined *nigrippus* as separate from either *megalippe* or *plexippus* subspecies. It is also uncertain whether monarchs in the northern and northwest portions of South America are subspecies *plexippus*, *megalippe*, or *nigrippus*.

J. Pelham stated that "*plexippus* represents the 'Monarch' as we have known it, *megalippe* represents the Caribbean fraction, which is typical of many widespread Neotropical butterfly species and *nigrippus* represents the southernmost entity" (J. Pelham, pers. comm. 2017). This classification depends on *Danaus plexippus plexippus* being migratory, and the other subspecies being non-migratory. However, non-migratory *Danaus plexippus plexippus* exist throughout the range (both within North America and throughout the Pacific). There are many unknowns about the precise borders of the monarch range, and there is even more difficulty in precisely determining where potential subspecies might interface. Most of scientific papers on *D. plexippus* examined do not specify subspecies, further complicating any determination of where potential subspecies might exist.

Given the complexity and uncertainty of monarch subspecies, as well as the petitioners' request to determine "whether any newly identified North American subspecies may warrant federal protection" (Center for Biological Diversity et al. 2014, p. 16), we are considering monarchs (*Danaus plexippus plexippus*) throughout the known range of the species.

Appendix 2. Methods

[1] Updates to Voorhies et al. 2019 model

Since the publication of the Voorhies et al. model, we made several changes to the model:

1) The input values have been updated: lambda values, epsilon values, and starting population sizes.
2) The time-frames for the influences are now "influence-specific." We allow each influence to reach its full magnitude of impact within the time-frame specific to that influence; in the published paper, the magnitude of change was incrementally distributed over 50 years (see paper methods section or p. 4).
3) The influence of climate is modeled differently in a couple ways. [1] The effects of climate change continue to be incorporated via availability of milkweed. In this version, climate change effects are combined with milkweed over the milkweed specific time-frame (20 years) and on its own for an additional 40 years (to reach the full duration of the climate change effect). [2] In this version, climate change is also combined with migration nectar influence in the south subregion. It is combined in the same way it is combined with milkweed and is used as an input to the migration nectar population response curve for the southern sub-region of the eastern population).
4) Future scenarios for milkweed and nectar in breeding habitat in the eastern population now include subregion specific values to be fed to subregion specific population response curves. Previously, we had one future scenario for milkweed and nectar in the breeding range and it was applied to all three subregions using their subregion specific population response curves. Now both inputs and response curves are subregion specific).

[2] Inputs to model

Table 2A1. Initial starting values for the population model. Inds= individuals

Population	Model Parameter	Value	Source
Both	Years	60	SSA Team
	Simulations	1,000,000	SSA Team
Eastern	Ninit	3.656 ha	5-year average
	λ	-0.0408	Semmens et al. 2016*
	ε	0.721	Semmens et al. 2016*
	Extinction threshold low	0.05 ha	Expert-elicited, Voorhies et al. 2019
	Extinction threshold high	0.61 ha	Expert-elicited, Voorhies et al. 2019
	Density/ha	2.11E+07 inds/ha	Thogmartin et al. 2017a
	Cap	36 ha	SSA Team
	Regional Importance_NC	0.68	Expert-elicited, Voorhies et al. 2019
	Regional Importance_NE	0.20	Expert-elicited, Voorhies et al. 2019
	Regional Importance_S	0.12	Expert-elicited, Voorhies et al. 2019
Western	Ninit	168,365 inds	5-year average
	λ	-0.13	Schultz et al. 2017
	ε	0.99	Schultz et al. 2017*
	Extinction threshold low	20,000 inds	Schultz et al. 2017*
	Extinction threshold high	50,000 inds	Wells et al. 1990
	Cap	2,400,000 inds	SSA Team

*Parameter values differ slightly from Schultz et al. 2017 and Semmens et al. 2016 because the population datasets have been updated with values through winter 2019-2020.

[3] Other threats and catastrophic events considered

In addition to the primary influences considered above, we also looked at many other factors that may be impacting monarchs. These included but were not limited to natural enemies (disease/parasitism), captive rearing, collection, impacts of tourism at overwintering sites, invasive swallow-wort plants, vehicle mortality, and natural catastrophes. We also considered other potential positive impacts, such as positive impacts of research and monitoring.

Other Stressors

Monarchs are impacted by a number of diseases and natural enemies. One of the most well-known and well-studied natural enemies of monarchs, *OE* (a monarch parasite), impacts worldwide populations at different rates (see *Representation* section in Chapter 2; Altizer and de Roode 2015, p. 84), with non-migratory populations typically having higher rates of infection (Bartel et al. 2011. p. 348). This protozoan parasite impacts monarchs (*OE*'s only known host), leading to decreased survival and fitness in the monarch (Altizer and Oberhauser 1999, p. 85). While infection rates can be high, we have not seen a large and continuous increase in proportion of monarchs that are heavily infected over time in eastern North America (Project Monarch Health 2016, p. 1). Other diseases can infect monarchs, including nuclear polyhedrosis virus, but most reports of these are anecdotal and no reports to our knowledge indicate increasing rates of disease (Arnott et al. 1968).

In addition to disease and parasites, immature monarchs are heavily preyed upon by natural enemies (upwards of 90% of monarchs are killed in immature stages; Nail et al. 2015a), but there is not any conclusive evidence available that suggests predation rates are currently increasing. These immature monarch predators range from ants, tachinid fly parasitoids, and various other insects for eggs and larvae, and wasps (*Pteromalus cassotis* and *Polites dominulus*) for pupae (Oberhauser et al. 2015, p. 72). The most studied larval natural enemy, the tachinid fly parasitoid, does not show a significant trend in proportion of monarchs parasitized over the years studied (Oberhauser et al. 2017b, p. 6). Adult monarchs also have predators, many of which have been documented at the Mexican overwintering sites (including birds, mice, and wasps; Oberhauser et al. 2015, p. 72). There is thought to be an approximate bird predation rate of 9% (Brower and Calvert 1985, p. 864), with potentially higher rates at smaller sites (Calvert et al. 1979, p. 850). However, these higher rates of predation have not been measured since the recorded decline in the eastern North American population began.

Captive rearing of monarchs was considered, as there are potential negative impacts of this practice on a large scale (Altizer et al. 2015, pp. 1-3). However, the number of monarchs being raised in mass-rearing operations is unclear (Villareal 2015, p. 9-10), and the impacts were difficult to quantify; thus, we did not consider this a primary influence. There is some information on vehicle mortality on insects (Baxter-Gilbert et al. 2015, Keilsohn et al. 2018), and some research on monarch vehicle mortality specifically (McKenna et al. 2001, Mora Alvarez et al. 2019, Kantola et al. 2019), and while this warrants future attention, we did not feel we had enough information to show that this was increasing or one of the current primary drivers of changes in monarch populations, nor was it identified as a primary driver in our expert elicitation. We did not find strong evidence of tourism at overwintering sites or insect collection impacting monarchs at the population level; hence, we did not currently consider them as primary influences. This is not to say that these or other threats could not become primary influences going forward, and thus should continue to be evaluated in the future.

The impact of invasive swallow-wort plants on monarchs was another influence that was considered. Black swallow-wort (*Cynanchum louiseae*) and pale swallow-wort (*C. rossicum*) are two European plants that are invasive in North America. They are in the milkweed family, but monarch caterpillars are unable to feed on these plants. However, there has been observed oviposition on these plants by adult monarchs, leading to speculation that these plants could serve as ecological sinks. However, the evidence for this is limited, with one study showing no oviposition on these species in the laboratory (DiTommaso and Losey 2003, p. 207) and another study showing limited oviposition in the field when common milkweed is scarce (Casagrande and Dacey 2007, p. 633). Given this evidence, we did not think invasive swallow-wort plants were a primary influence for driving the monarch decline.

We also considered the direct impacts of herbicides to monarchs. Results of herbicide toxicity studies suggest that various types of herbicides may result in direct effects to lepidopterans if exposed at recommended field application rates for the labeled land use/cover type. However, the direct effects of most herbicides to monarchs are unknown, and likely to be highly variable. In several studies, the simulated application site was some type of conservation area where chemical control of invasive plants was presumed, resulting in maximum exposure of herbicide to lepidopteran. It is important to note that we found no studies evaluating the effects of herbicides to lepidopterans at concentrations representative of exposure due to drift from an application site to nearby habitat (i.e., exposure concentrations at less than a labeled rate) for this risk assessment. While we acknowledge the potential for toxic effects of herbicides to monarchs under certain exposure conditions, we consider the effects of insecticides to be the primary driver in monarch population impacts due to pesticides (insecticides, herbicides, fungicides, rodenticides, etc.). See our *Supplemental Materials 1b* for a detailed description of the direct impact of herbicides on monarchs, including data, references, and supporting information.

We also considered positive influences, such as research and monitoring (e.g., the information that might be gained from the national integrated monitoring strategy). While these future impacts are difficult to determine or quantify, we note the importance of these efforts and their potential future influence on monarch populations.

Other Catastrophic Events

<u>Fire</u>

The frequency, size, and intensity of wildfire in the western U.S. has increased over time (Littel et al. 2009, p. 1003; Waterbury and Potter 2018, p. 43). The three largest fires in California history occurred in 2017 and 2018. Wildfire pose risks to both breeding and overwintering habitat as well as causing direct mortality of butterflies. Given the broad distribution of breeding habitat throughout the West, it is unlikely, however, that any single fire or series of fires would destroy a sufficient amount of habitat such that catastrophic losses occur. Additionally, monarchs are highly mobile and may be able to escape slow-moving fires and thus, direct mortality is unlikely. Similarly, during the winter, monarchs occupy numerous sites along broad areas of coastal California. Coupled with the close proximity of many of these sites to residential areas (where fire is more likely to be quickly contained), the likelihood of a catastrophic fire is low (Pelton et al. 2016, p. 28). However, if population numbers continue to decline, the impact of losing some portion of breeding habitat or one or two of the largest overwintering sites will increase the risk of extinction for the migratory population. Thus, there is insufficient information indicating that the magnitude and severity of fire poses a catastrophic risk to the western monarch population.

<u>Hurricanes</u>

Much of the coastline of the eastern U.S. has sustained impact by multiple hurricanes in recorded history (NOAA 2010). The states hit hardest by hurricanes are occupied by the eastern migratory population throughout much of the year. Hurricanes have the potential to kill some individual monarchs but only a hurricane in Texas or Mexico during peak migration to Mexico could have catastrophic effects on the eastern population. In an analysis by Ries et al. (2018, pp. 98-101), the authors determined that hurricanes and large masses of migrating monarchs are unlikely to cross paths in time and space because most major hurricanes happen in September or earlier and migrating monarchs funnel through Texas in October and November. Although hurricanes also have the potential to indirectly affect monarchs (Ries et al 2018, pp. 99-101), there is no evidence indicating that indirect effects (e.g., increased fall plant growth) would be catastrophic to the eastern migratory population. Currently, there is no evidence that major storms have directly killed masses of individual monarchs, and there are anecdotal accounts of monarchs surviving or flying in the opposite direction of severe storms (Journey North 2008; Moskowitz et al. 2001, p. 488). Should the timing and duration of hurricane season change in the future, as has been suggested by news outlets but not supported by research (see Karloski and Evan 2016, p. 273), migrating monarchs could be at an increased risk. Thus, there is currently insufficient information indicating that hurricanes pose a catastrophic risk to the eastern monarch population.

[4] Future scenarios

Eastern North American Population – Milkweed and nectar projections for Eastern North America were driven by milkweed stem changes from conservation efforts, Conservation Reserve Program acres, and land cover change.

Conservation Efforts

To calculate milkweed stem estimates, we began by establishing a baseline for the year 2014 using a "seamless" land cover dataset developed by Rohweder and Thogmartin (2016) that combined data from the National Land Cover Dataset (NLCD), Cropland Data Layer (CDL), Topologically Integrated Geographic Encoding and Referencing, and Homeland Security Infrastructure Program. We used the seamless dataset to calculate the number of acres of each land cover type in eastern subregions. We then multiplied the acres of each land cover type by the corresponding milkweed stem density in stems per acre from Thogmartin et al. (2017c), which were derived from literature and expert input. The result was an estimate of the total number of milkweed stems on the landscape in the Northcentral, Northeast, and South subregions. We assumed milkweed density is a reliable proxy for habitat quality, including nectar resources. Further, we assumed that the milkweed density estimates in the upper Midwest can be reasonably applied to Northeast and South subregions.

Using land cover type and acreage information in the Monarch Conservation Database (MCD), we calculated the current amount of habitat due to conservation efforts by adding milkweed from completed and implemented conservation efforts to the 2014 baseline number of milkweed. We calculated the number of milkweed from conservation efforts by tallying the number of acres of each land cover type that have been improved due to completed and implemented conservation efforts, and multiplying those acres by the net change in milkweed. We calculated the net change in milkweed by subtracting baseline milkweed stem density from the user provided data or "potential" milkweed density for the land cover type in question when user provided data was not available (Table 2A2). Milkweed density values in Table 2A2 for each land cover type are generally based on Thogmartin et al. (2017c; further clarified via pers. comms with Thogmartin), and represents the average estimate of biologically reasonable milkweed density for a given land cover type (derived from a combination of literature review and expert input). Potential milkweed density was not available for all land cover types due to discrepancies between land cover types used in Thogmartin et al. (2017c) and the seamless dataset (Rohweder and Thogmartin 2016). The estimated baseline and potential milkweed densities represent the current state of knowledge and can be updated when additional information becomes available.

We then derived a level of future conservation effort, relative to the current amount of habitat with upper and lower bound projections of Conservation Reserve Program acreage and land cover change. Our future scenarios (upper and lower bounds) included formalized, but not yet implemented (i.e. planned) conservation efforts submitted to the MCD. We assume the conservation efforts completed to-date will be maintained and continue to provide monarch milkweed and nectar resources for both scenarios.

For the Northcentral subregion, we assumed implementation of the Mid-America Conservation Strategy, which will result in an estimated 1.3 billion additional milkweed stems by 2038 from monarch conservation efforts. To account for net change since 2014, we calculated the gain in milkweed from completed and implemented efforts in the MCD as described above and subtracted this figure from the 1.3 billion stem goal. The result is the remaining total number of additional milkweed stems needed to meet the 1.3 billion stem goal from all potential sources and sectors. Next, we subtracted the projected gains under the upper bound scenario from Conservation Reserve Program and land cover projections (see below) to calculate the number of additional milkweed stems specifically from non-CRP conservation efforts needed to achieve the 1.3 billion stem goal relative to 2014 levels. For the lower bound in the Northcentral subregion, we assumed that additional conservation effort would occur to offset a portion of projected CRP losses; in this case, conservation effort equated to the same level of effort associated with the upper bound scenarios plus the equivalent gains that we had projected due to CRP increases under the upper bound scenario. In essence, the same level of habitat would be added to the landscape under the lower bound scenario as was assumed under the upper bound scenario (minus the additional benefits that were attributed to projected land cover change); however, additional losses would simultaneously occur due to broader CRP declines at that resulted in losses greater than the CRP gains under the upper bound scenario (also see *Conservation Reserve Program*). For the Northeast and South subregions, given the lack of an overarching monarch conservation strategy analogous the Mid-America Monarch Conservation Strategy, we simply calculated the change in milkweed from future formalized conservation efforts in the MCD using the methodology described above and similarly added the upper and lower projections CRP and land cover.

Table 2A2. Baseline and potential milkweed densities for land cover types. Values from Thogmartin et al. 2017.

Classification	**Estimated Baseline Milkweed Density**	**Potential Density**
22 - Developed Low Intensity (NLCD) (Inside Urban Areas)	1.00	50.00
23 - Developed Med Intensity (NLCD)	0.50	25.00
24 - Developed High Intensity (NLCD)	0.10	10.00
26 - Developed Low Intensity (NLCD) (Outside Urban Areas)	19.74	84.50
21 - Developed Open Space (NLCD) Linear	0.00	16.31
25 - Developed Open Space (NLCD) Core	0.00	3.09
120 - TIGER Secondary Roads	57.15	175.00
110 - TIGER Primary Roads and Ramps	57.15	150.00
140 - TIGER Local Roads	57.15	100.00

Classification	Estimated Baseline Milkweed Density	Potential Density
174 - TIGER Private Roads	3.09	3.09
180 - All TIGER Roads (Inside Urban Areas)	0.00	0.00
31 - Barren (NLCD)	0.00	0.00
41 - Deciduous Forest (NLCD)	0.00	0.00
42 - Evergreen Forest (NLCD)	0.00	0.00
43 - Mixed Forest (NLCD)	0.00	0.00
76 - Grassland (NLCD)	3.09	40.00
77 - Grassland (NLCD) PADUS Protected	3.09	250.00
100 - HSIP Transmission Line (Outside Urban Areas)	3.09	150.00
101 - HSIP Transmission Line (Inside Urban Areas)	0.00	0.00
200 - TIGER Rails (Outside Urban Areas)	3.09	200.00
201 - TIGER Rails (Inside Urban Areas)	0.00	0.00
52 - Shrubland (NLCD)	3.09	3.09
1 - Corn LOW	0.05	4.04
14 - Soybeans LOW	0.05	4.04
3 - Other Crops (CDL) LOW	3.09	5.56
4 - Other Crops (CDL) MEDIUM	5.30	7.74
5 - Other Crops (CDL) HIGH	7.50	9.93
6 - Fallow Idle (CDL) HIGH	3.09	4.05
7 - Fruit Xmas Trees Vines (CDL) LOW	3.09	5.56
8 - Fruit Xmas Trees Vines (CDL) MEDIUM	5.30	7.74
9 - Fruit Xmas Trees Vines (CDL) HIGH	7.50	9.93
2 - Corn LOW (Marginal)	0.05	200.00
15 - Soybeans LOW (Marginal)	0.05	200.00
10 - Hay Alfalfa (CDL) LOW	3.09	40.00
78 - Pasture (NLCD)	3.09	40.00
79 - Pasture (NLCD) PADUS Protected	3.09	126.55
95 - Herbaceous Wetlands (NLCD)	61.37	68.16
90 - Woody Wetlands (NLCD)	61.37	68.16
Unclassified (Weighted average of all land cover types)	7.03	28.63

Conservation Reserve Program

To calculate the net change in Conservation Reserve Program acres from 2014 and 2018 and current amount of CRP acreage, we began by requesting county-level information from the Farm

Service Agency (FSA) for acres of CRP conservation practices that Thogmartin et al. (2017c) determined to be beneficial for monarchs. We shared with an FSA economist a "non-sensitive" version of the seamless dataset for consistency and the economist was able to extract from their system and the seamless dataset a breakdown of CRP acres for conservation practices benefitting monarchs by land cover type in each county for 2014 and 2018. We then applied the baseline and potential milkweed stem density for each land cover type per Thogmartin et al. (2017c) (see *Conservation Efforts*) to calculate the total number of milkweed from CRP acres and subtracted 2014 county totals from 2018 county totals to get the net change. We added the net change in CRP milkweed to milkweed from completed and implemented conservation efforts to calculate the current habitat due to CRP. For the milkweed and nectar future scenarios with respect to CRP, we assumed a 22% increase relative to 2018 CRP milkweed in the upper bound, and a 35% loss in the lower bound, respective to each subregion, based on USDA projections, recent trends in CRP acreage, and expert opinion (USDA 2020; Skip Hyberg, retired Senior Economist, pers. comm.).

Land Cover Change

We used the FORE-SCE (FOREcasting SCEnarios) land cover change model developed by the USGS Earth Resourses Observation Science (EROS) Center to develop future scenarios with respect to background changes in land cover under a range of emissions scenarios between 2010 and 2050 (Sohl et al. 2018). Unfortunately, the land cover types used in the FORE-SCE model did not all match the land cover types from Rohweder and Thogmartin (2016) or Thogmartin et al. (2017c) despite being based largely upon the same underlying dataset (the 2011 National Land Cover Dataset, NLCD). We matched any mismatched land cover types used in the FORE-SCE model with seamless dataset land cover types using overarching themes (e.g. developed, agriculture, grassland, wetland, etc.; Table 2A3). Additionally, there were land cover sub-types for which the FORE-SCE model did not predict future change but were crucial components of the seamless rater dataset, such as roads and rail lines. For roads and rail lines, we estimated the change based on mile statistics over the past decade from the U.S. Department of Transportation (USDOT 2020a, 2020b). Due to a lack of available data, we assumed no change in acreages of transmission lines. For Conservation Reserve Program, see methodology described above. For seamless dataset land cover types grouped into a single FORE-SCE land cover type (e.g. cropland), we assumed the percent change projected in the FORE-SCE model or other datasets applied evenly to all grouped land cover types. Projected changes in the "Mechanically Disturbed" and "Mining" land cover types used in the FORE-SCE model were not accounted for, as there is no analogous land cover type defined in the seamless dataset. While the projected percent change in some conservation units are significant, they generally accounted for a relatively small proportion of the landscape.

Once we calculated the percent change for each land cover type using the FORE-SCE model, we applied that percent change to the seamless dataset using the Table 1 to calculate projected acres of seamless dataset land cover types and applied the milkweed stem densities per Thogmartin et al. (2017c) to calculate future milkweed. We assumed linear change from 2010 and 2050 and divided the change over the 40-year period to calculate annual change and projected acres of each land cover type in 2018. We subtracted the 2014 baseline milkweed from projected milkweed due to land cover change in 2018 to calculate the net change in milkweed due to background land cover change. Under all scenarios, we project an increase in milkweed due to background land cover change. This projected increase in milkweed stems initially seems counterintuitive given that the FORE-SCE model and other sources of information (i.e., USDOT road mile statistics) generally predict an increase in more "developed" land cover types and a slight decrease in more land cover types such as grassland and shrubland. The numerical increase in milkweed due to land use change is largely a factor of differences in the estimated milkweed stem density for each land cover type. For example, certain types of roadway corridors are estimated to have much higher baseline milkweed stem densities than grassland or shrubland. While land use change appears to result in an increase in milkweed stems numerically, what is not factored in is the overall quality of habitat. As such, we used the projected increase in milkweed stems from the FORE-SCE for the upper bound scenario with respect to milkweed and nectar from land cover change. For the lower bound, we assumed no net change due to land cover change.

Table 2A3. Groupings of land cover type from the USGS EROS FORE-SCE model and Rohweder and Thogmartin 2016.

Classification (FORE-SCE Model)	**Classification (Rohweder and Thogmartin 2016)**
Developed	Developed – Low/ Medium/High Intensity, Exurban, Open Space
NA	Roads – Secondary, Primary & Ramps, Small, Private, Inside Urban Areas
Mechanically Disturbed National Forest, Other Public Lands, Disturbed Private	NA
Mining	NA
Barren	Barren
Deciduous Forest	Deciduous Forest
Evergreen Forest	Evergreen Forest
Mixed Forest	Mixed Forest
NA	CRP - Non-wet, Wet
Grassland	Grassland, Protected Grassland
NA	Transmission Line
NA	Rails
Shrubland	Shrubland
Cropland	Corn, Soy, Other Crops, Fallow Idle, Fruit/Christmas Trees & Vines
Hay/Pasture	Hay Alfalfa, Pasture, Protected Pasture
Herbaceous Wetland	Herbaceous Wetland
Woody Wetland	Woody Wetland

[5] Climate change projections

To calculate the percent change from 2012 to 2069 in the average number of days and spatial extent of which temperatures are above 38˚C between April and May and 42˚C between April and May of 2012 and 2069 in the continental United States, we downloaded climate projections from the Multivariate Adaptive Constructed Analogs (MACA) Climatology Lab (Abatzoglou and Brown 2012). The MACA Climatology Lab provides downscaled climate data from a number of Coupled Model Intercomparison Project 5 (CMIP5) climate models (Taylor 2012). For simplicity, we use the period between April and May in 2012 and 2069 and a threshold of 38˚C to describe our methodology. We downloaded projected daily maximum surface temperature for the continental United States for the 2006-2025 and 2066-2070 timeframes, and to account for variation between models and uncertainty, we downloaded projections under Representative Concentration Pathway (RCP) scenarios 4.5 and 8.5 and averaged outputs from 5 models. Each dataset came in the form of a NetCDF file, which consists of "stacked" raster datasets (Figures 2A1 and 2A2). Each approximately 4.6km x 4.6km grid cell of the dataset contains the daily "tasmax", or maximum air temperature in degrees Celsius 2 meters above the surface of the Earth for one day (Figure 1). We used the raster package in RStudio to import the data as a raster brick, or a stack of the 61 rasters, with each raster representing one day between April 1st and May 31st (Figure 2; Hijmans 2017; RStudio Team 2015). To calculate the total number of cells in a raster with tasmax values above 38˚C between April 1st and May 31st, we reclassified each raster, assigning all cells with tasmax values 38˚C or below a value of 0, and all remaining cells (i.e. cells with tasmax values above 38˚C) a value of 1. The result was a stack of 61 reclassified rasters, each containing cells with values of 0 or 1 indicating whether the tasmax was above 38˚C at that location. We refer to the reclassified value of each cell as a "cell day". Since each individual raster represents a single day, the maximum cell day value for any given cell is 1. We summed the rasters together to get the total number of cell days above 38˚C between April 1 and May 31 (Figure 2A3). The final combined raster is the sum of all 61 individual rasters and therefore, the maximum cell day value for any given cell in the resulting raster is 61, which would mean that every day between April 1st and May 31st has a tasmax above 38˚C at that cell location. We then plotted the final combined raster to get a map that indicates the number of cell days for each cell between April 1st and May 31st that had a tasmax of 38˚C degrees or above for a single model.

We then averaged the number of cell days above 38˚C at each cell across 5 models to capture the range of projections and plotted the average combined raster (Figure 2A4). We followed this process for the year 2012 and 2069. From these data, we were able to calculate the change in the spatial extent of temperatures above 38˚C spatially by calculating the change in the percent of land area occupied by cells with cell day values of at least one, and tabularly by calculating the change in the percent of total number of cell days with tasmax values above 38˚C.

We calculated the number of cells with at least one cell day above 38˚C by summarizing the raster table and summing the number of cells with tasmax values greater than one. The result was a binary dataset with either cells with no days with tasmax values above 38˚C or cells with one or more days with tasmax values above 38˚C. By dividing the number of cells with cell day values greater than one by the total number of cells in the raster, we calculated the change in the spatial extent of cell days with tasmax values above 38˚C. Using Figure 2A4 as an example, there are 6 cells with at least one cell day above 38˚C and thus the spatial extent of temperatures above 38˚C is 67% (6÷9 total cells).

To calculate the change in the percentage of total cell days with values above 38˚C, we first calculated the total number of cell days for each final combined raster by multiplying the total number of cells in the raster by 61, or the total number of days between April 1st and May 31st. Since the final combined rasters represent averaged days above 38˚C across 5 models, we rounded the day values to the nearest whole number to avoid having fractions of days (Figure 5). Next, we used the raster table to calculate the number of cell days with tasmax values above 38˚C. Raster tables generally consist of a "value" column and a "count" column. The "value" in the tasmax rasters used in this analysis represents the number of days above 38˚C and the count is the number of cells in the dataset with that number of days above 38˚C. We multiplied each value by the corresponding count and summed the products to get total number of cell days above 38˚C. Using Figure 2A4 as an example, there are 3 cells with no days above 38˚C, 1 cell with 1 day above 38˚C, 4 cells with 2 cell days above 38˚C, and 1 cell with 3 days above 38˚C. Multiplying each value with its count (0x3, 1x1, 2x4, and 3x1) and summing gives 12 total number of cell days above 38˚C. The total number of cell days in the example is 27 (9 total cells in each raster multiplied by 3 days), and thus the percent of cell days with tasmax values of 38˚C is 44% (12÷27).

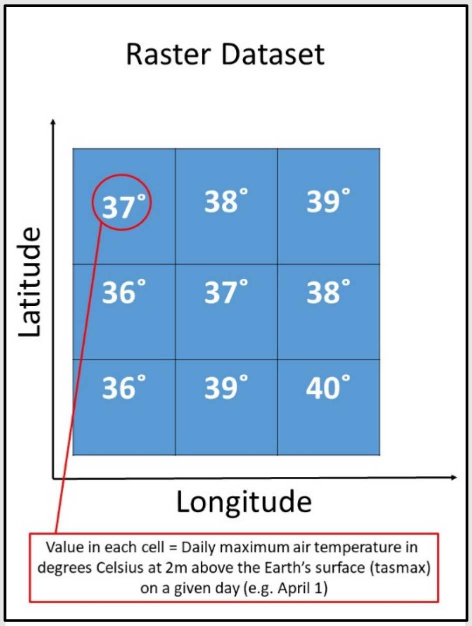

Figure 2A1. A raster dataset is composed of a spatially referenced grid with each grid cell containing data. For this analysis, the data in each cell represents the daily maximum air temperature 2 meters above the surface of the earth.

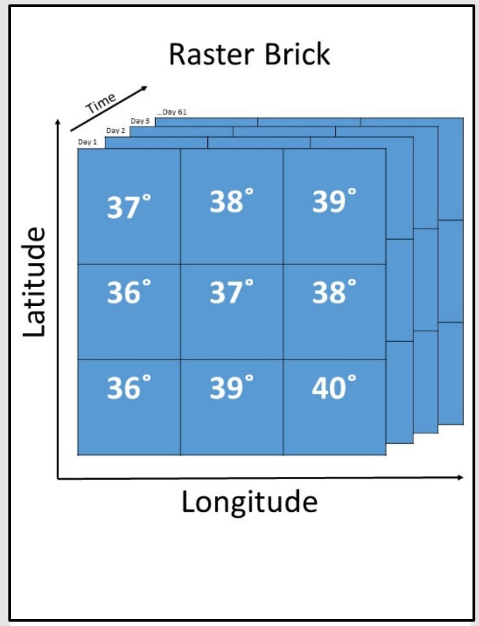

Figure 2A2. A raster brick consists of stacked individual raster datasets. For this analysis, each raster represents a single day between April 1st and May 31st.

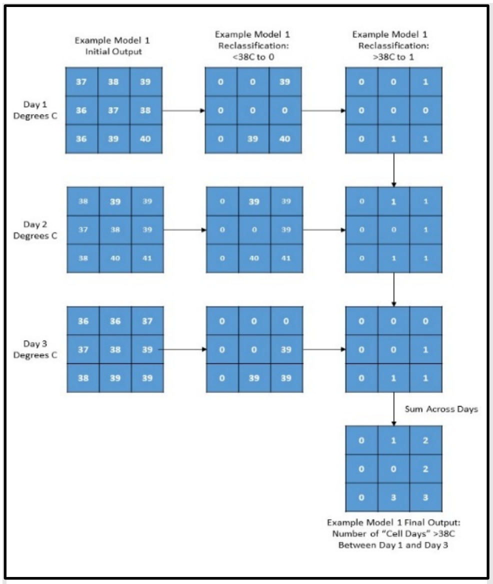

Figure 2A3. Reclassifying the raster dataset and assigning a value of "0" to all cells with tasmax value of 38˚C or lower and a value of "1" to all cells that have a tasmax value of above 38˚C. This resulting value is referred to in this analysis as a "cell day". Summing through the days of a raster brick provides the total number of cell days above 38˚C.

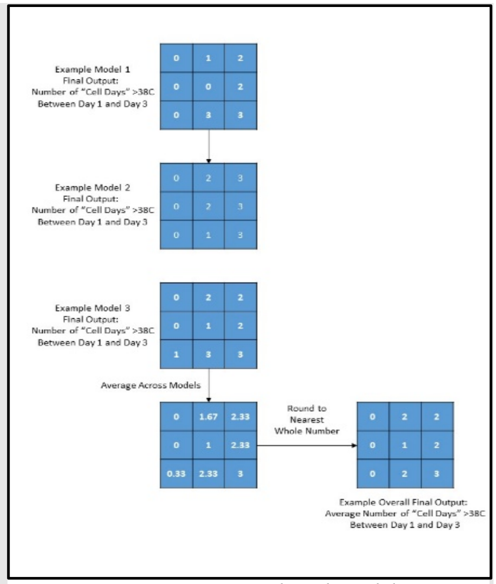

Figure 2A4. Averaging combined model outputs to get the average number of cell days above 38˚C.

Worldwide

We evaluated projections from downscaled General Circulation Models produced by the Coordinated Regional Climate Downscaling Experiment under Representative Concentration Pathways RCP scenarios 4.5 and 8.5 from the Earth System Grid Federation to visually determine if the spatial extent and number of days above the lethal threshold (42˚C) is projected to increase (CORDEX 2018; Cinquini 2014). Where possible, we used bias-adjusted outputs averaged across at least one iteration of each model available to account for variation across models and scenarios. To capture the warmest period for each population, we focused on the July and August timeframe in the northern hemisphere and January and February for Australia and Central America and April and May for Southeast Asia in the southern hemisphere. For Australia and Central America, we were able to average the results over three models; however, downscaled data was only available for scenario RCP 8.5. For the populations in Southeast Asia, we averaged over three models, but only one model output was available for RCP 4.5. We obtained five downscaled and bias-corrected datasets for both RCP 4.5 and 8.5 scenarios for Europe. We also obtained global climate projections from General Circulation Models developed under the Climate Model Intercomparison Project 5 (CMIP5) so we could evaluate projections for all populations more consistently (Taylor 2012). We note that because a population is in the "No Known Risk" risk category does not necessarily mean it has no risk overall (it could be at risk due to one of the influences we were unable to evaluate); rather, it is at no known risk for the two influences that were evaluated.

We acknowledge the World Climate Research Programme's Working Group on Coupled Modelling, which is responsible for CMIP, and we thank the climate modeling groups (Met Office Hadley Centre, Max Planck Institute *for Meteorology, Norwegian climate Centre, Centre National de Recherches Meteorologiques / Centre Europeen de Recherche et Formation Avancees en Calcul Scientifique, European EC-EARTH Consortium, Institut Pierre-Simon Laplace, Canadian Centre for Climate Modelling and Analysis, Centro Euro-Mediterraneo per I Cambiamenti Climatici, Atmosphere and Ocean Research Institute (The University of Tokyo), National Institute for Environmental Studies, and Japan Agency for Marine-Earth Science and Technology, National Center for Atmospheric Research) for producing and making available their model output. For CMIP the U.S. Department of Energy's Program for Climate Model Diagnosis and Intercomparison provides coordinating support and led development of software infrastructure in partnership with the Global Organization for Earth System Science Portals.*

Appendix 3. Additional Results

[1] Percent change in area and average number of days above 38°C and 42°C

Table 3A1. Projected 2012 (May and April) baseline total number of 4.6km x 4.6km grid cells and average number of cells with at least one day above 38˚C and 42˚C under RCP 4.5 and 8.5.

Population Unit	Total Number of 4.6km^2 Raster Cells	Average Number of Cells w/at Least 1 Day			
		>38˚C Under RCP 4.5 (Apr-May 2012)	>38˚C Under RCP 8.5 (Apr-May 2012)	>42˚C Under RCP 4.5 (Apr-May 2012)	>42˚C Under RCP 8.5 (Apr-May 2012)
Eastern					
Northcentral	134,563	3,845	0	67	0
Northeast	23,445	0	0	0	0
South	147,796	32,573	35,446	100	2,777
West	161,501	29,085	24,983	10,452	7,403

Table 3A2. Percent change in the area and average cell days above 38° C for each conservation unit under RCP 4.5 and 8.5 from April and May of 2012 to 2069.

Population Unit	% Change in Area RCP 4.5	% Change in Area RCP 8.5	% Change in Cell Days RCP 4.5	% Change in Cell Days RCP 8.5
Eastern				
Northcentral	-99	1,008,000	-99	1,008,800
Northeast	28,400	16,900	28,400	16,900
South	94	200	331	438
Western	-23	109	38	114

Table 3A3. Percent change in the area and average cell days above 42° C for each conservation unit under RCP 4.5 and 8.5 from April and May of 2012 to April and May of 2069.

Population Unit	% Change in Area RCP 4.5	% Change in Area RCP 8.5	% Change in Cell Days RCP 4.5	% Change in Cell Days RCP 8.5
Eastern				
Northcentral	-99	30,000	-99	30,000
Northeast	0	0	0	0
South	6,630	1,637	8,147	3,575
Western	-11	148	11	182

[2] Projected area and average number of days >38˚C and 42°C under RCP 8.5

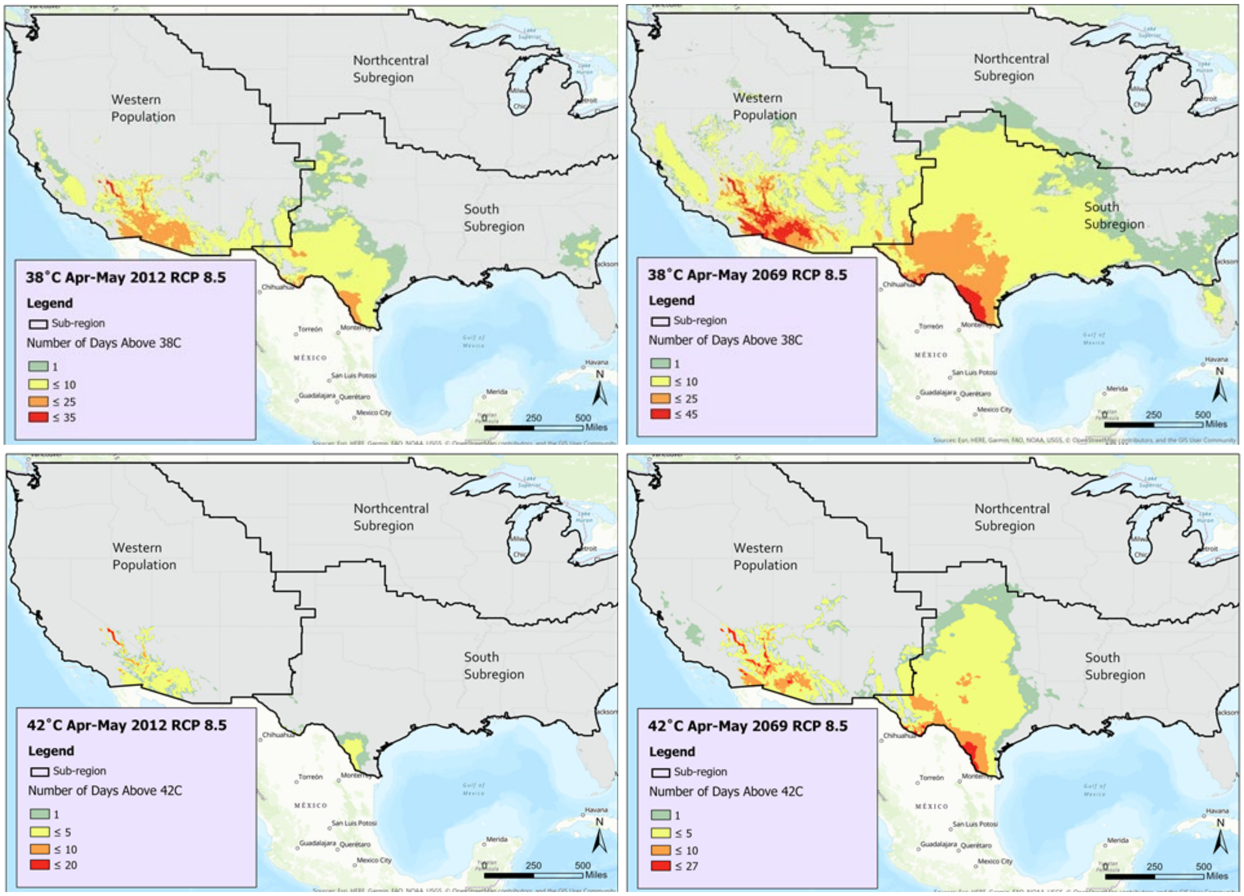

Figure 3A1. The spatial extent and average number of days >38˚C (top) and 42°C (bottom) in April and May 2012 (left) and 2069 (right) under RCP 8.5.

[3] pE over time under current and future state conditions

Table 3A4. pE values for the western and eastern North American populations. pE predictions under current state conditions represent the 50% confidence interval.

	10 Year	20 Year	30 Year	40 Year	50 Year	60 Year
Western Pop						
Current - 25%	0.60	0.80	0.90	0.95	0.97	0.99
Current - 75%	0.68	0.85	0.93	0.97	0.98	0.99
Future - Worst case	0.71	0.88	0.95	0.97	0.99	0.99
Future - Best case	0.66	0.84	0.92	0.96	0.98	0.99
Eastern Pop						
Current - 25%	0.02	0.09	0.18	0.29	0.39	0.48
Current - 75%	0.08	0.22	0.36	0.49	0.60	0.69
Future - Worst case	0.09	0.29	0.46	0.58	0.67	0.75
Future - Best case	0.04	0.13	0.24	0.35	0.46	0.56
p(both pops persist)						
Current - 25%	0.39	0.18	0.08	0.04	0.02	0.01
Current - 75%	0.29	0.12	0.04	0.02	0.01	0.00
Future - Worst case	0.27	0.08	0.03	0.01	0.00	0.00
Future - Best case	0.33	0.14	0.06	0.02	0.01	0.00

Supplemental Materials 1a for the Monarch (*Danaus plexippus plexippus*) Species Status Assessment Report, Revised July 2020

U.S. Fish and Wildlife Service: Sarah Warner, Nancy Golden, Dave Warburton, Lisa Williams

The Risk of Insecticides to the Monarch Butterfly

The risk of insecticide impacts to monarchs is primarily influenced by the extent to which monarchs are exposed to insecticides throughout their range. This assessment presents an overview of: (1) the use of insecticides within monarch habitat, (2) pathways of monarch exposure to insecticides, (3) toxicity of insecticides to monarchs, and (4) a summary evaluation of insecticide risk. Factors influencing insecticide exposure and the uncertainties inherent in these factors are also presented to guide future research/monitoring and monarch conservation strategies.

Insecticides in Monarch Habitat

The monarch butterfly is widely distributed across the United States, occurring in a variety of urban and rural habitat types that include milkweed plants and other flowering forbs. Monarch habitat includes gardens and yards, urban parks, farmlands and other agricultural production areas, rights of way, and protected natural areas. Though pesticide use is most often associated with agricultural production, any habitat where monarchs are found may be subject to insecticide use or exposure. Insecticides can be used for insect pest control anywhere there is a pest outbreak or for general pest prevention. Homeowners may treat yards and gardens to protect plants from pests or purchase plants from nurseries that sell insecticide-treated plants (often from the neonicotinoid class of pesticides) as ornamentals. Natural areas, such as forests and parks, may be treated to control for insects that defoliate, bore into wood, or otherwise damage trees. Outbreaks of pests such as gypsy moths, Mormon crickets, or grasshoppers may trigger insecticide treatments over larger areas to control populations. Use of insecticides in vector control, especially pyrethroids and organophosphates, may be significant in areas of the country where mosquitoes pose a public health threat or reach nuisance levels.

Expenditures on insecticides in 2012 topped $5 billion in the United States, with 60 million pounds being used for agriculture (57%), home and garden (23%), and in the industrial/commercial/governmental sector (20%; EPA 2017). Chemical classes of the most commonly used insecticides during the time of the report (2008 - 2012) were organophosphates and carbamates, and pyrethroids (EPA 2017). In addition, neonicotinoid insecticides (a class of insecticides first registered in the 1990s) accounted for 80% of global seed treatment sales by 2008 (Jeschke et al. 2011). Treated seeds are used for nearly all of the corn and soybean crop acreage in the U.S. (Douglas and Tooker 2015), and neonicotinoid-treated plants are commonly sold as ornamentals for yards and gardens.

Given this extent of insecticide use over the wide distribution of monarch habitat across a variety of land use sectors, there is significant potential for monarchs to be exposed to insecticides in the United States.

Monarch Insecticide Exposure Pathways

Insecticide exposure pathways to both adults and larvae of the monarch include: (1) *dietary exposure* (ingestion of an insecticide on or within plant tissue that the monarch is feeding upon), and/or (2) *contact exposure* (direct contact with airborne insecticides that land on the monarch or are deposited on plants that the monarch comes in contact with). Figure 1 illustrates these potential insecticide exposure pathways to each life stage of the monarch. While the monarch may be exposed to insecticides throughout all life stages, this evaluation is limited to larval and adult stages, as these are considered to be the most significant from a biological perspective, and the most likely in actual environmental settings. Further, there are insufficient data to evaluate exposure and effects to the other life stages beyond a conceptual analysis. Due to overlapping generations of monarchs through the spring-fall months, both larvae and adults may be exposed to insecticides in any given geographic location the species may occur outside of its overwintering areas.

Figure 1. Insecticide exposure pathways to monarch life stages.

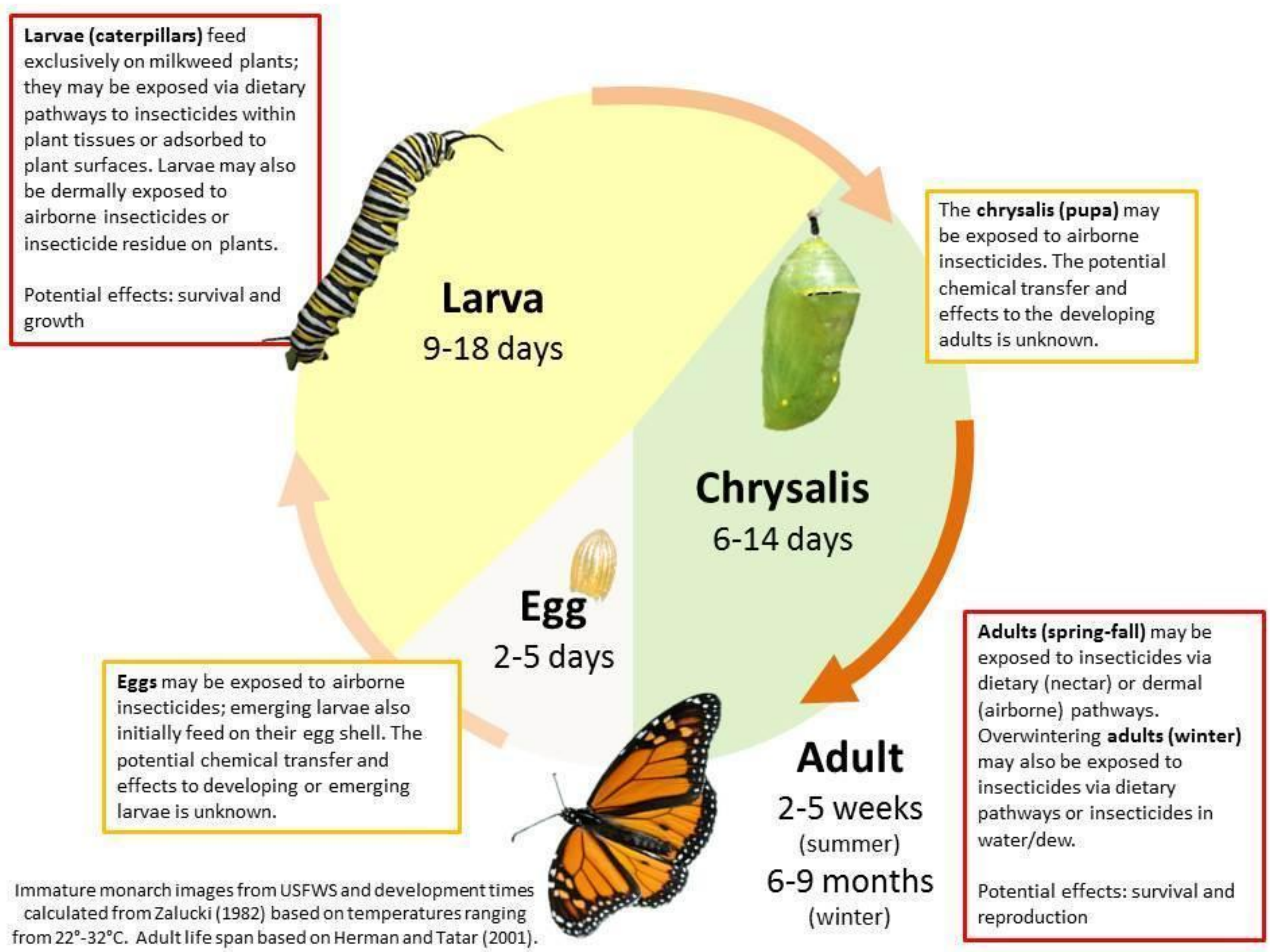

Figure produced by Kelly Nail and Dave Warburton, USFWS.

Insecticides can move through the environment and expose monarchs by the following routes:

1) **Direct Spray**: Monarchs that inhabit the same area as insect pests are susceptible to insecticide exposure (through either dietary or contact pathways) via direct spray of insecticides. One significant scenario for this occurrence is in areas subject to mosquito control with pyrethroid and organophosphate insecticides (used as mosquito adulticides).
2) **Pesticide Drift**: Monarchs may be exposed to pesticides via dietary or contact pathways in any area adjacent to a treatment location where the pesticide leaves the site of application ("drifts") via droplets, vapor, or dust. Whether a pesticide will drift, and how far from the treatment area that drift occurs, are influenced by numerous factors including method of application, height of spraying equipment, wind speed, weather conditions, nozzle size, terrain, and the use of best management practices by applicators to control for these factors and limit drift occurrences.
3) **Systemic**: Monarchs may be exposed via dietary pathways to insecticides that become

incorporated into plant tissues (e.g., leaves, pollen, nectar). Although numerous insecticides may be systemic to some degree, neonicotinoids in particular are known for this characteristic, and are expressed throughout the plant including nectar and pollen of treated crops and plants (Goulson 2013).

The degree to which an insecticide persists and moves through the environment can influence its availability, and thus exposure to monarchs. Pesticides can differ widely in these characteristics, even within the same class of chemicals; those which persist longer or are more mobile can result in greater exposure to monarchs.

For example, chemical characteristics of many neonicotinoids include high water solubility and relatively long persistence in the environment. These characteristics contribute to the propensity of neonicotinoid insecticides to transport long distances beyond use areas. Neonicotinoids have been found in well-water (Starner and Goh 2012, Huseth and Groves 2014), and can also drift off-site when incorporated into pollen (Bonmatin et al. 2015), suggesting far-reaching effects and potential landscape-scale mobility. When used as seed treatments, over 90% of the active ingredient can enter the soil and remain available (reported half-lives range from 200 to over 1000 days, Goulsen 2013). During seed sowing, less than 2% is lost in dust-off; more can be lost and deposited in the field margin areas if talcum powder or graphite is added to the seeds (Krupke et al. 2012).

For a monarch to be exposed to an insecticide through its diet, residues must be deposited on or incorporated within the dietary item associated with the relevant life stage, specifically milkweed leaves for larvae and nectar from flowers for adults. How the plant metabolizes or stores insecticides in its tissues and how it is expressed in leaves or nectar can influence exposure potential and the degree of risk to monarchs and needs to be studied. While insecticide residues have been documented in both of these media, few studies exist to help estimate concentrations (i.e., the magnitude of exposure) in the variety of areas where monarchs may be exposed, including agricultural and adjacent lands, residential areas, and parks or other presumed natural areas.

Exposure to pesticides in pollen and nectar

While monarchs are not expected to feed on pollen, reports of its widespread contamination in crop areas illustrates the ability of flowering plants to serve as sources of exposure, at least in areas in and around crops. Presence in pollen is likely indicative of presence in nectar and with further investigation into the relative accumulation of residues, concentrations measured in pollen may be used to estimate concentrations in nectar. There is some evidence that residues in nectar may be lower than those in pollen, though factors such as application method, application timing, and environmental conditions are likely to affect concentrations available to monarchs from this source. There are few North American studies measuring concentrations occurring in plants following exposure based on typical or labeled application methods, and a lack of field sampling from active crops and non-crop areas.

Investigations of contaminants in honeybee colonies illustrate that insecticides used in crops are available to pollinating insects. In a large-scale study of colonies in 23 states and one Canadian province, representing several agricultural cropping systems, concentrations of 98 different pesticides were detected in collected bee pollen (Mullin et al. 2010). Bee pollen, which aggregates pollen collected from different individuals and flowers, contained an average of 7 pesticides per sample. Chlorpyrifos was the most frequently detected insecticide in 44% of samples.

Residues of insecticides were regularly detected in pollen and nectar following two studies of experimental pesticide applications in field conditions, though concentrations varied. Average concentrations of neonicotinoids in pollen from pumpkins following various methods of application ranged up to 80.2 ng/g imidacloprid (plus an additional 19.1 ng/g metabolites), 88.3 (10.3) ng/g dinotefuran, and 95.2 (26.8) ng/g thiamethoxam (Dively and Kamel 2012). Concentrations were lower in the second year of the study, presumably due to extreme environmental conditions resulting in heat and moisture stress. Neonicotinoid metabolites accounted for 15 - 27% of total residues across years. Residues in nectar were consistently 74 - 88% lower than pollen residues, and residues in leaves were generally higher, though only correlated with values in pollen and nectar for imidacloprid. At-planting applications resulted in the lowest concentrations, and those applications occurring closer to flowering resulted in higher residues. In another study, concentrations of imidacloprid or thiamethoxam in nectar and pollen of squash treated via soil application or drip irrigation (a subset of the application methods tested in the above study) resulted in similar concentrations in pollen (5-35 ng/g) and nectar (5-20 ng/g) regardless of application method, insecticide, or study year (Stoner and Eitzer 2012). Average concentrations were 14 ng/g imidacloprid and 12 ng/g thiamethoxam in pollen, and 10 ng/g imidacloprid and 11 ng/g thiamethoxam in nectar. Residues were similar across two study years despite rainfall totals in the second year about half of those in the first. Data for metabolites were not presented.

In a study simulating greenhouse application, residues of imidacloprid and its metabolites (hydroxy and olefin), were measured in Mexican milkweed (*Asclepias curassavica*) flowers following soil applications at labeled rates for greenhouse use (Krischik et al. 2015). Whole flowers contained a mean of 6,030 ng/g midacloprid and 980 ng/g metabolites 21-51 days post-application. A second soil application 7 months after the first resulted in mean concentrations of 21,670 ng/g imidacloprid and 6,440 ng/g metabolites in whole flowers. The authors speculated that the higher residues from this application may be due to concentration in flowers during a time of slower vegetative growth. Metabolites accounted for 14% and 23% of total residues for each year, respectively, similar to the percentages measured in nectar and pollen described above. The authors acknowledge that residues in pollen and nectar may be different than residues in whole flowers and that the correlation needs to be scientifically determined.

Exposure to insecticides in milkweed leaves

Larval monarchs can be exposed to insecticides by ingesting residues that are expressed in the leaf tissue of milkweeds. Insecticides have been detected in milkweed leaves near agricultural fields in at three two studies. Variation in frequency of detection and concentration levels across years or seasons was common to both studies. While the two studies below measure concentrations in common milkweed, it is worthwhile to note that in the toxicity studies reviewed below, monarchs are exposed using four different species of milkweed plants. At present, it is not known whether the pharmacokinetics (i.e., how the plant metabolizes, stores, and expresses systemic insecticides in its tissues) is comparable across milkweed species and how this may affect the exposure and bioavailability to monarchs using these plants.

Clothianidin was measured in common milkweed (*Asclepias syriaca*) leaves that were adjacent to fields (mean distance of 1.47 m) at eight sites in South Dakota shortly after maize planting in 2014 using an ELISA method[1] (Pecenka and Lundgren 2015). Mean clothianidin concentration per plant was reported as 0.58 ppb overall and 1.14 ppb in plants with detectable residues, with a maximum 4.02 ppb in one plant. Clothianidin was detected in about half of the samples, with twice the proportion having detectable residues in July (65%) compared to June (37%). Monitoring of plants during sampling revealed that monarchs were actively using these sites, with an average of 1.3 eggs and 0.6 larvae per plant in June, and 1.4 eggs and 0.3 larvae in July.

Olaya-Arenas and Kaplan (2019) analyzed pesticides in soil and leaves of common milkweed (*Asclepias syriac*a) within 100 m of crop fields in northwest Indiana to determine if areas adjacent to fields provide greater exposure to monarchs. Three neonicotinoids were detected in leaves with variation in percent detection and concentrations by year. Clothianidin was detected in 15-25% of samples in June, but rarely detected in July or August. Concentrations varied between 2015 (0.71 ng/g mean, 56.5 ng/g maximum) and 2016 (0.48 ng/g mean, 28.5 ng/g max). Thiamethoxam was detected in just 2% of samples in 2015 (0.19 ng/g mean, 94.8 ng/g max), yet found in 75-99% in 2016 (1.87 ng/g mean, 151.3 ng/g max). Imidacloprid was only detected in 0.2% of samples in 2015 (up to 3.7 ng/g) and was not detected in 2016. The pyrethroid deltamethrin was detected in 98.9% of samples in 2016 (37.0 ng/g mean, 1,352.9 ng/g max). Distance from the edge of a crop field or the amount of crop was generally a poor predictor of pesticide detection, with only thiamethoxam demonstrating this relationship. Clothianidin was the only insecticide detected in soil, with concentrations consistent throughout the summer and correlated with those in milkweed leaves. In general, higher concentrations of insecticides were found earlier in the season with year to year variation.

Halsch et al. 2020 investigated insecticide exposure to milkweed plants across three land-use sectors that included agriculture, wildlife refuges, urban parks and gardens in northern California. The field study determined what pesticides are available to monarch during a one-time sampling event in late June - when monarch larvae are likely to be present. In this field study, 227 leaf samples of narrowleaf milkweed (*Asclepias fascicularis,*161 samples), common milkweed (*A. speciosa*, 50), woolly pod milkweed (*A. eriocarpa*, 4) and tropical milkweed (*A. curassavica*, 12) were collected from 19 sites across the Central Valley. The sites were located in conventional farms, an organic farm, a milkweed restoration site, a roadside location adjacent to an agriculture field, five in wildlife refuges, four in urban areas, and two from retail nurseries. In addition to the milkweed samples that were collected in the field, milkweed plants were purchased from home and garden stores and leaves were analyzed for pesticides. A total of 64 pesticides were detected across samples: 25 insecticides, 27 fungicides, 11 herbicides, and 1 adjuvant. A greater number of pesticides were detected in plants sampled from agricultural and retail locations compared to samples from refuge and urban sites. Chlorantraniliprole (registered for use in urban areas) was detected in 91% of the samples and methoxyfenozide (registered for

[1] In reviewing the methods as described in this paper and correspondence with one of the authors who stated that he did not think that leaf disks were weighed, it is not clear whether the reported concentrations in ppb are on a ng/g basis in the leaves or a ug/L basis in the leaf extracts, so these concentrations should be considered to be less certain than those from other publications cited in this document.

use on a variety of crops) was measured in 96% of samples. The authors compared the concentrations detected in milkweed leaves to honeybee and monarch toxicity levels. Sixteen percent (36 out of 227) of the milkweed leaves sampled had concentrations over an LD50 value for honeybee toxicity with exceedances from 7 of the 19 sampled sites. Three other pesticides (cyantraniliprole, fipronil, and methoxyfenozide) exceeded a honeybee LD50 and these were sampled from retail and urban sites. In 25% of the samples, chlorantraniliprole concentrations exceed a tested LD50 for monarchs. Clothianidin was detected above a monarch LD50 from one agriculture site. Authors indicate that for the vast majority of the pesticides detected in the milkweed leaves it is unknown what the biological effects are on monarch caterpillars.

Effects of Insecticides to Monarchs

Insecticides are pesticides with chemical properties that are designed to kill insects. Their main uses are to control insect pests in agricultural production, natural habitats, lawns and gardens, and in and around households and buildings. The U.S. Environmental Protection Agency (USEPA), under the authority of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), regulates and registers pesticides for use in the United States. To evaluate the environmental risk of proposed pesticide use as part of the registration process, the USEPA requires laboratory studies of toxicity to select non-target species. The non-native honeybee (*Apis mellifera*) is currently the primary invertebrate surrogate used in testing to evaluate risks to non-target terrestrial insects. If negative effects to non-target species are anticipated from the proposed use of a pesticide, the USEPA may choose to not approve the pesticide for registration, or to require restrictions on pesticide labels to help minimize anticipated impacts. However, under FIFRA risk management, a degree of non-target risk may be deemed acceptable if the risks are outweighed by the potential benefits of use of a pesticide. Therefore, risk to non-target species, including monarchs, cannot be ruled out simply because a pesticide has undergone the registration process and is used according to the label.

Most insecticides considered herein are non-specific and broad-spectrum in nature. That is, insects exposed to insecticides are broadly susceptible to mortality and sublethal effects. Furthermore, the larvae of many insects in the Order Lepidoptera are considered major pest species, especially in agricultural and forested areas, and insecticides are tested specifically on this taxon to ensure that they will effectively kill individuals at labeled application rates. Therefore, it is reasonable to presume that monarchs exposed to insecticides within areas of use are likely to be killed or otherwise affected following an application. Monarchs exposed in areas outside insecticide use where drift occurs may also be affected depending on the concentration of the pesticide to which they are exposed.

Scientific data documenting insecticide effects to lepidopterans are largely limited to: (1) laboratory dosing studies on larvae to investigate the toxicity of an insecticide with various endpoints measured, (2) modeling studies predicting the extent of insecticide threat to individuals or populations, and (3) field-based studies that investigate insecticide concentrations in plant tissues (as described above) and/or attempt to measure effects to populations in treated and untreated areas. All three types of studies have their limitations. For example, standardized methods of laboratory toxicity testing have not yet been adopted for lepidopteran species, resulting in inconsistencies in exposure regimes (e.g., duration, contact vs ingestion, life stage) and reporting of toxicity values (e.g., units of measurement). Lack of accepted testing protocols confound the ability to make comparisons across studies and species. Given such variability, this section presents a brief summary of select information from published literature on the effects and toxicity of three widely-used classes of insecticides to monarchs or other lepidopteran species: organophosphates, pyrethroids, and neonicotinoids. Conclusions are noted where possible. Other classes of insecticides and other types of pesticides can be similarly investigated.

Organophosphates and Pyrethroids

Information on direct toxicity of organophosphate and pyrethroid insecticides to lepidopteran species is available from efficacy studies on target pest species (particularly *Pieris brassicae* and related species, reviewed in Braak et al. 2018). In this assessment, we generally focus on toxicological effects to non-target species, with data available within the families Nymphalidae, Lycaenidae, Papilionidae, Hesperiidae, and Pieridae (Salvato 2001, Hoang et al. 2011, Eliazar and Emmel 1991, Hoang and Rand 2015, Bargar 2012a, Davis et al. 1991). Most studies measured the acute toxicity of insecticides to various lepidopteran species and report median lethal dose values (LD50s) for dietary or contact exposure pathways. Methods varied across studies in relation to length of exposure, life stage, chemical form (active ingredient vs formulated product), and exposure regime. In general, while toxicity was exhibited across all species and chemicals, no consistent patterns emerged either within or across studies that demonstrated sensitivity was related to species (or species group), life stage, or size of adults, though inconsistency in testing regimes may limit the ability to detect patterns that exist. Of the organophosphates tested (dichlorvos, malathion, naled, and dimethoate) species tended to exhibit the greatest sensitivity to naled and the least to malathion, though these results were not always consistent across species and methods. For pyrethroids, toxicity values were reported for two insecticides, permethrin and resmethrin. However, resmethrin testing was performed in formulation with piperonyl butoxide, a synergist that is combined with pesticides to enhance toxicity and comparisons cannot be made between relative toxicity of these two insecticides. Based on the available data from these insecticide studies, there is no evidence to imply that a particular species or family of lepidopterans is expected to exhibit more or less sensitivity to a particular organophosphate or pyrethroid than others, including targeted pest species.

Only two studies looked specifically at effects to monarchs within these classes of insecticides. Both studies found that monarchs exposed to pyrethroids at concentrations expected following field applications could experience mortality. Oberhauser et al. (2006) found that larvae that consumed milkweed leaves treated with permethrin in dilutions of field operable solutions (dilutions 0.5 and 0.1%) had significantly reduced rates of survival. Of the 60 larvae exposed to the two treatments, 37 died (33 as larvae and 4 as pupae) and larval stage development time was significantly delayed. Survival rates were lower for first instar larvae compared to later instar larvae. In the same study, effects to female oviposition choice, the number of eggs laid, and survival 1, 8, and 15 days after the initial spray event. Females were placed in enclosures that contained milkweeds exposed across three treatment groups: (1) milkweed plants sprayed with operational solutions of permethrin, (2) milkweed sprayed with operational solutions of permethrin, treated with oil solution, and untreated, and (3) milkweed plants that were untreated. Overall female survival was low for the two permethrin treatments (8-16 %) compared to 92% survival for the untreated treatment; with the lowest survival rate 1 day after the initial spray event. In addition, the studies found that ovipositing females did not discriminate amongst treatment groups, but fewer eggs were laid on permethrin treated plants

1 day after initial spray date compared to treated plants 8 and 15 days later.

Oberhauser et al. (2009) exposed adult and larval monarchs to ultra-low volume (ULV) applications of resmethrin (as the formulated product Scourge, which contains resmethrin plus the synergist piperonyl butoxide) to evaluate the effects of mosquito control on monarchs. Three experiments examined impacts to survival in adults and larvae subject to direct spray at varying locations upwind and downwind, and in larvae consuming previously exposed milkweed. Monarch mortality varied with conditions of experimental design, but significant increases over controls were found at distances up to 120 m downwind from the application site over the three experiments. Milkweed plants sprayed one day prior to monarch exposure resulted in significant mortality to larvae as compared to controls. In one of three experiments, adult mass was negatively affected by exposure to resmethrin. One experiment exposed house fly (*Musca domestica*) and milkweed bug (*Oncopeltus fasciatus*) larvae to resmethrin under conditions that caused monarch mortality and found no effects to survival of either species.

Neonicotinoids

There are few published studies examining the toxicity of neonicotinoids to monarchs (described herein). A summation of toxicity values of neonicotinoids across taxa (insects, birds, fish, molluscs, mammals, annelids) found insects to be the most sensitive taxa when exposed via contact or the dietary/ingestion pathway with LD50s ranging from 0.82 to 88 ng per insect (Goulson 2013). The variation in LD50 values is attributed to size of the insect, with the most sensitive insect being the brown planthopper (*Nilaparvata lugens*; a native species) weighing 1 mg, and the least sensitive insect being the Colorado potato beetle (*Leptinotarsa decemlineata*; a crop pest and non-native species) weighing 130 mg.

Three studies looked specifically at neonicotinoid effects to monarchs. While reduced survival was detected in most treatments, results of each study were influenced by differences in pesticide tested, life stage, exposure regime, and experimental methods. Pecenka and Lundgren (2015) attempted to mimic a pulsed exposure in the field by feeding swamp milkweed leaves dosed with clothianidin to larvae for 36 hours during the first stadium, and then observing effects up to the third instar. Each larva was fed a single 1 cm milkweed disk with an aqueous solution of clothianidin on agarose gel on the leaf. Once that disk was consumed, the larvae were then fed clean milkweed leaves until the end of the experiment in the third instar. Increasing mortality was observed with increasing dose, measured in μg/L (ng/g) clothianidin in the 10 μL of solution applied to each leaf disk: the LC10, LC20, LC50, and LC90 concentrations were found to be 7.72, 9.89, 15.63, and 30.70 ng/g, respectively. Significant effects to development time, body length, and weight for newly eclosed second instars were observed at doses as low as 0.5 ng/g. This study reveals effects to monarchs at seemingly low environmental concentrations of clothianidin; however, concentrations as reported (ug/L of solution per leaf disk) are not easily extrapolated to typical concentration units for a dietary testing exposure scenario (gram per leaf or ng/g ww of leaf). Therefore, it is difficult to make a direct comparison to concentrations expected to be found on milkweed leaves in the environment.

Krischik et al. (2015) investigated imidacloprid rates for greenhouse/nursery use. The authors suggest that this particular use of the insecticide can result in higher concentrations of residues found in flowering plants compared to imidacloprid used as a seed treatment; therefore, it was selected for the study. Multiple experiments were conducted using Mexican milkweed (*Asclepias curassavica)* plants with imidacloprid applied to the soil to investigate dietary exposure pathways from whole flowers or plant tissues to insects. Mexican milkweed flowers grown in soils treated with imidacloprid at labeled rates reduced survival in 3 of 4 lady beetle species, in some cases as soon as two/three days after treatment. Adult monarch and painted lady butterflies either free-ranging or force-fed imidacloprid in solution showed no effects to survival, fecundity, or egg hatch at either labeled rates or twice labeled rates. However, larval survival of both species was reduced by day 7, with few monarchs surviving past this point. Authors hypothesized that adult butterflies may not metabolize the insecticide, instead excreting it unchanged.

James (2019) examined the effects of nectar dosed with imidacloprid on monarch longevity and egg production. For the 28 day study, adult monarchs (11 males, 11 females) were consistently fed a sugar-water solution containing 23.5 ng/g imidacloprid, a concentration within the range detected in nectar of crop plants. Mortality occurred in dosed monarchs and individuals exhibited behavioral effects by day 12 (uncoordinated flapping of wings and uncontrolled vibrating of body and wings). Sample sizes throughout the study were low: At 12 days post eclosion, 4 males and 4 females remained in the dosed group with 4 males and 4 females in the control. At 22 days post eclosion, 2 individuals remained in the dosed group with 3 males and 5 females in the control. No effects were detected in mass, forewing length, oocyte development, and growth. This study tested one scenario in which adult monarchs feed on the nectar of crop plants treated with imidacloprid under certain conditions. It is uncertain the degree and frequency to which monarchs nectar on crop plants, the full range of concentrations likely to be present in treated plants, and if the imidacloprid concentration tested is representative of what could be expressed in the nectar of native flowering plants.

To determine the residue level in the milkweed tissue that leads to an adverse effect to monarchs, Barger et al. (2020) conducted three experiments that estimated the dietary exposure level of clothianidin associated with adverse effects in monarch butterflies. Results showed transfer of clothianidin from soil to milkweed plant (swamp milkweed- *Asclepias incarnata*), to larvae and to adult – this is the first study to show life-stage transfer from soil to adult. In the experiments, swamp milkweed plants were dosed (via soil treatments) with five concentrations, each experiment increasing dose levels, and larvae were exposed via dietary exposure from the time they hatched from eggs until pupation. Endpoints measured included larval survival and growth, pupation success, and adult mass. Experiment 1 consisted of concentrations that included the label rate for application of a clothianidin product, while Experiments 2 and 3 included only concentrations greater than the label application rate. In Experiment 1, clothianidin was measured in the milkweed leaves and detected in the larvae only at the two dose levels greater than the label rate, with concentrations in leaves measured at 11 ng/g (SD = 3.6) and 54 ng/g (SD = 27) in the two dose groups and in larvae at 6.0 ng/g (SD 3.3) and 13 ng/g (SD 3.4). Two of the three surviving adult butterflies from the highest dose group had detectable concentrations of clothianidin (3.1 and 5.2 ng/g). At the label application rate, concentrations in leaves, larvae, and adults were all below the detection limit and no significant effects to larval growth and survival, adult mass, pupal were observed. For Experiments 2 and 3, dose levels were all greater than label application rates for several clothianidin products. The greater dose levels resulted in detectable concentrations in leaves and larvae from all treatments. Experiment 3 was conducted to eliminate the possible effect of aphids that infested plants during Experiment 2; therefore, only the results for Experiment 3 are reported herein; however, the elevated exposure in both experiments led to adverse effects on survival and growth. Clothianidin was detected in the milkweed leaves at measurable concentrations ranging from 54 (SD = 42) to 1,545 ng/g (SD = 481), and larval consumption of the contaminated leaves negatively affected larval growth and adult survival. Larval growth was affected at 1,154 ng/g leaf and no larvae in this highest dose level reached the pupal stage. Larval mortality ranged from 50% in the lowest dose level (54 ng/g leaf) to 100% in the highest dose level, and 33-50% of the monarch butterflies died at the pupal stage in both of the lowest dose levels tested. Four adult monarchs successfully eclosed, three in the control and one in the lowest dose level. Due to the results of the three consecutive experiments, the authors suggest that clothianidin concentrations expected from applications that follow the label in wild milkweed plants are generally not high enough to adversely affect monarch butterflies and that monarchs may be relatively insensitive to clothianidin at label application rates.

Krishnan et al. 2020 conducted contact (cuticular) and dietary toxicity tests on monarch butterfly larvae at each life-stage for five insecticides that are registered for use as foliar applications on maize and soybean: a pyrethroid (beta-cyfluthrin), an anthranilic diamide (chlorantraniliprole), an organophosphate (chlorpyrifos), and two neonicotinoids (imidacloprid and thiamethoxam). For the dietary assays, larvae were reared on insecticide-treated tropical milkweed (*Asclepias curassavica*) leaves for 48 or 24 hours. Contact and dietary LD50s differed among larval stages with first instars being the most sensitive followed by third and fifth instars. The LD50 concentrations for beta-cyfluthrin and chlorantraniliprole ranged from 9.2 to 480 ng/g larva and 12.0 to 190 ng/g larva. respectively, and were the most toxic insecticides across all instars. Chlorpyrifos was the least toxic to first instars (LD50 of 79,000 ng/g larva). For the neonicotinoids, clothianidin was more toxic to larvae than both imidacloprid and thiamethoxam.

Risk Evaluation

Ecological risk assessment to evaluate potential insecticide effects to monarchs can be assessed by (1) comparing laboratory-derived toxicity values to environmental concentrations of insecticides (based on either predictive modeling or post-application sampling), and/or (2) studying effects to individuals exposed to insecticide applications in the field. Effects (lethal and sublethal) are then characterized and a determination is made as to the extent of risk. Additional (unknown) risk in the field can be caused from indirect effects of insecticides, such as susceptibility to disease or predation, and the potential for additive or even synergistic effects from exposure to multiple pesticides in the field. The lack of standardized toxicity testing andlimited monarch-specific data limit a definitive risk assessment for monarchs. Accordingly, available assessments generally center on other lepidopteran species, from which risk to monarchs can be extrapolated.

Organophosphates and pyrethroids

Though organophosphate and pyrethroid insecticides are used in all facets of pest control, there has been particular interest in performing risk assessments based on exposure scenarios from mosquito control applications, as lepidopterans can be exposed within the site of application (i.e., they occur in areas where mosquitoes are treated). In particular, the need for mosquito control in southern Florida has led to concerns regarding the effects on native lepidopterans. The few studies described below indicate that mosquito adulticide applications may pose risk to lepidopteran species, but results differed across studies, pesticide type, and species. Mosquito adulticide treatment differs from other treatments in that application rates tend to be lower than other uses, and pesticide is applied in a ULV spray designed to maximum time before deposition so as to encounter airborne mosquitoes. For these reasons, factors such as application rate and environmental transport should be considered when relating the risk assessments and field studies of mosquito adulticides described below to other uses of these insecticides (e.g., cropland, natural areas, and residential settings).

In an assessment of the risk of naled, deposition was measured 50 minutes following a single pre-dawn ULV spray for mosquito control (applied as Trumpet EC at a rate of 70 g a.i./ha; Bargar 2012b). These results were combined with morphometric data for 22 species within 5 families to estimate deposition onto butterflies roosting in the application area during a pre-dawn spray. Using lepidopteran toxicity values from the literature (described above), a 67-80% chance of exceeding the mortality estimate for the butterflies was predicted following such a spray. Assuming equivalent sensitivity, the greatest risk was estimated for butterflies within the Lycaenidae family, and the lowest risk for those within the Hesperiidae family; relative risk to butterflies within Papilionidae, Nymphalidae, and Pieridae families was considered to be intermediate.

Another risk assessment examined potential effects to native Florida caterpillars from the mosquito control pesticides permethrin, naled, and dichlorvos (Hoang and Rand 2015). Exposure data for this analysis were taken from a report generated from a field monitoring program in Big Pine Key, Florida in 2007-08, though measured values on leaves were not presented directly in Hoang and Rand (2015). The joint probability analysis in the risk assessment revealed that permethrin concentrations on host plants had a 42% chance of exceeding the lowest observed adverse effects dose (LOAED) for native Florida caterpillars and a 0.02% chance of exceeding acute LD50 values. Probabilities of exceedance for diclorovos were 11% and 2.2% for its LOAED and LD50, respectively, and the probability of exceedance was 11% for the LD50 for naled. The authors indicated that these values may underestimate actual risk in the field as they are based solely on 24-hour dietary exposure and do not consider the influence of direct topical exposure from drift or chronic exposure from insecticide persistence on leaves.

Two other field studies also examined native butterfly populations in areas with mosquito control. Population surveys in the rock pinelands of south Florida (Long Pine Key) and the Lower Florida Keys (Big Pine Key) were conducted in areas that receive year-round application of pesticides (pyrethroids and organophosphates) for mosquito control and those without such treatment (Salvato 2001). Adult densities of Florida leafwing (*Anaea troglodyta floridalis,* family Nymphalidae) were significantly lower in treated areas than in control areas. Population counts of Bartram's scrub-hairstreak (*Strymon acis bartrami*, family Lycaenidae) and Meske's skipper (*Hesperia meskei*, family Hesperiidae) did not appear to be reduced following pesticide application. In a second study, insecticide residue deposition and butterfly survival were monitored following a spray of naled during routine mosquito control in North Key Largo, Monroe County, Florida (Zhong et al. 2010). Sampling stations were set up within the spray zone, drift zone, and control areas (>25 miles away). Survival rates of 5th instar Miami blue butterfly caterpillars (*Cyclargus thomasi bethunebakeri*, family Lycaenidae) were 52-98% at sampling stations within the spray zone, and did not differ between drift and control zones. Naled was recorded in a remote drift zone 12 miles from the application area causing mortality to test mosquitoes in sampling stations, but not to butterfly larvae similarly exposed. Naled concentrations greater than 1000 ug/m^2 were associated with dramatically reduced larvae survival rates, though larvae surviving to the pupal stage successfully emerged. Wind speed was associated with higher deposition and larval mortality.

Neonicotinoids

While no field studies exist to assess the population effects of neonicotinoids, modeling studies have attempted to relate monarch declines to this class of pesticides. Forister et al. (2016) investigated neonicotinoid use and butterfly declines at four sites in Northern California that have been monitored for four decades. The model indicated an association between declining butterfly numbers and increasing neonicotinoid use, suggesting that neonicotinoids could influence populations occurring close to application sites. Similarly, Thogmartin et al. (2017) analyzed multiple threats to monarchs including climate, habitat loss, disease, and insecticides in a time series analysis using partial least squares regression models. Glyphosate and neonicotinoid use in monarch breeding habitat were both correlated with the observed monarch population decline. Gilburn et al. (2015) modeled neonicotinoid usage on agricultural lands and population estimates for 17 species of butterflies in the UK from 1985 to 2012. A negative correlation was indicated for hectares of farmland that used neonicotinoid pesticides and butterfly population declines. The authors determined that more studies are needed to determine if there is a causative link between neonicotinoid usage and the decline of butterflies, or whether the negative correlation represents a proxy for other environmental factors associated with intensive agriculture practices.

In an assessment broadly examining insecticides, DiBartolomeis et al. (2019) incorporated existing toxicity data (honeybee LD50 data for contact and oral toxicity), persistence (soil half-life), and mass applied (estimated total pounds per acre used for foliar and seed treatments) to model pesticide loading (defined as acute insecticide toxicity loading, AITL) in agricultural land and surrounding areas. The model suggests that from 1992 to 2014, the AITL in the United States increased 4-fold based on contact toxicity and 48-fold based on oral toxicity. The authors attribute this change to an increase in pesticide loading from neonicotinoids beginning in 2004. Three neonicotinoids (imidacloprid, thiamethoxam, clothianidin) combined to contribute 91.8% of the total AITL for oral toxicity. As presented, the AITL is a measure of raw insecticide toxicity in the environment and does not take into account how non-target species such as monarchs may be exposed to these chemicals. As previously discussed, factors such as accumulation in exposure media (e.g., nectar, leaf, direct spray) and the location and timing of application can be highly influential in estimating effects to individuals and populations, and may differ across classes of insecticides. Environmental persistence, as measured by a chemical's half life in soil, appears to be a significant driver in results, yet its relationship to pesticide availability to nontarget target species in unclear. As such, it is difficult to translate the conclusions of this assessment to potential effects to monarchs.

In Krishnan et al. 2020, larval dose response curves generated from toxicity studies were used to model monarch mortality rates caused by insecticide drift exposure downwind from sprayed crop fields. Two scenarios were modeled: predicted spraying for (1) soybean aphid and (2) true armyworm - a pest of maize. The models took into account three application methods: aerial application, high ground boom, and low ground boom and predicted mortality rates (using both contact and dietary larval exposure data) between 0-60 meters from the edge of a sprayed field. Application rates based on the insecticide label were used in the models. Models for aerial applications using beta-cyfluthrin and chlorantraniliprole for the soybean aphid management scenario predicted larval mortality between 100 and 32% at distances 0-60 meters downwind from the agriculture field based on cuticular toxicity data. Based on dietary toxicity data, predicted larval mortality was between 100 and 10% for modeled distances downwind from the agriculture field. Larval mortality for chlorpyrifos, imidacloprid, and thiamethoxam, (using cuticular toxicity data) was 99, 91, and 67%. For the same insecticides, larval mortality was 96, 80, and 83% based on dietary toxicity data. Modeling for high ground boom applications produced similar predictions; however, lower mortality was predicted at distances 15, 30, and 60 meters downwind compared to aerial applications in which greater larval mortality was observed at 0 meters downwind. Across the scenarios, the mortality rates were generally highest for the first instars and lowest for fifth instars. The lowest percentage of monarch mortality was modeled at 60 meters downwind from the crop edge.

Summary and Conclusions

Despite inconsistencies in testing regimes (e.g., chemical concentrations, application methods and exposure routes, and life stage and species tested), studies presented here and in other reviews (Mule et al. 2017; Braak et al. 2018) demonstrate that insecticides can have negative effects on lepidopteran species. The majority of the studies evaluated for the Monarch Species Status Assessment are laboratory toxicity tests designed to identify the insecticide concentration that causes mortality or adverse effects. More recent laboratory toxicity studies have attempted to evaluate the effects at relevant environmental concentrations. Field studies are also available that measure insecticide concentrations in milkweeds or monitor effects to lepidopterans within and outside of an application site. Finally, modeling studies weigh the risk of insecticides amongst other threats to monarch populations. Many of these studies concluded that insecticide use may potentially have negative effects to lepidopterans, including monarchs. While these studies provide pieces of information to evaluate the risk of insecticides to monarchs, enough data gaps remain for the many variables involved to prevent a comprehensive analysis of effects.

As insecticides are generally likely to cause adverse effects to butterflies, exposure of monarchs (both adults and larvae) to these chemicals through diet and contact is the primary determinant of risk across a variety of land use sectors throughout the species' range. Monarch exposure to insecticides is not readily predictable, but dependent on individual monarchs encountering pesticide residues on or near the individual plants they use. In addition, exposure is influenced by factors such as the extent and frequency of insecticide use, timing of application, application rate and method, proximity of monarchs to the application site, contact with residues in the air or on plant surfaces, availability of residues in dietary items associated with lifestage present (leaves or nectar), and pesticide persistence.

The extent and manner of insecticide use itself is not regularly monitored or easily predicted in any given area. Insecticide use can vary both temporally and spatially, and is subject to regional or broad scale changes from disease and pest outbreaks, and emerging pest pressure. The toxicity of insecticides present on the landscape to lepidopterans may change based on the development and use of new insecticides, the regulation of older insecticides, the unknown effects of pesticide mixtures in the environment, and the advent of new technologies to prevent drift and reduce nontarget exposure.

Despite the challenges to determine a quantifiable extent to which insecticides impact the monarch population, and to determine a specific cause and effect relationship of insecticide effects to monarchs in environmental settings across various land use sectors, the substantial body of information available allows for a qualitative evaluation of the risk of insecticides to monarchs. Based on insecticide chemical characteristics and use; and the exposure potential, laboratory toxicity tests, field studies, and models presented herein, *insecticides are a threat to monarch populations*. This is primarily due to insecticides being used in areas on the landscape where monarchs occur; the fact that insecticides are designed to kill insects (and in many cases specifically target lepidopteran species); insecticides are likely to cause both lethal and nonlethal effects to non-target lepidopterans that are exposed in areas of application (such as crops fields, city parks, natural areas, residential areas, and yards and gardens); and may cause both lethal and nonlethal effects to non-target insects that are exposed from drift by droplet, vapor, and dust in areas outside of application sites and from systemic incorporation into non-target plant tissues.

Though many uncertainties (described throughout this assessment) regarding insecticide exposure and effects make it difficult to determine the *degree or extent of risk* to both individuals and at the population level, there are some factors that contribute to this uncertainty that are manageable and can be addressed through conservation actions, toxicity and exposure research and methodologies, and outreach/education programs. Manageable factors include:

- General awareness of insecticide use (e.g., ornamental plants and other consumer products that may contain neonicotinoids), and public policy affecting insecticide registration and use.
- Extent of development and adoption of best management practices for insecticide use, including Integrated Pest Management (e.g., establishing “acceptable levels” of pest pressure) and drift control measures.
- Extent of agricultural land uses with monoculture systems that increase the potential for, and frequency of, insect pest outbreaks and the economic need for chemical control.
- Societal expectations for widespread use of mosquito control insecticides.
- Technological capability to develop chemical insect pest controls which are more selective for the pest species, short-lived in the environment, less mobile, etc.
- Lack of standardized toxicity testing protocols to determine effects to the monarch and other non-target lepidopterans.
- Lack of standardized methods for field studies to determine the extent of exposure to the monarch population and other non-target lepidopterans.
- Field measurements of insecticide residues in select components of monarch habitat across a variety of land use sectors (i.e., quantified exposure).
- Lack of studies that clearly relate laboratory trials and field studies to realistic field exposure and effects to monarch butterflies.

Additional research and monitoring of aspects associated with these factors can provide the information necessary to reduce the uncertainties, and to determine which factors are the most important to manage risk. Most of these factors directly relate to insecticide exposure –

managing exposure manages risk. There are several guides and references available to manage insecticide exposure as part of broader monarch conservation strategies, including:

- Monarch Butterfly Conservation Report (see page 34) https://www.fws.gov/savethemonarch/pdfs/MonarchConferenceReport2016.pdf
- USFWS IPM for Lawns and Gardens https://www.fws.gov/pollinators/pdfs/FWS_IPM_Urban_Outreach_Final_April_26_2018_final_web_508.pdf
- USFWS IPM for Farmlands https://www.fws.gov/pollinators/pdfs/FWS_IPM_Farmland_Outreach_Final_April_26_2018_web_508.pdf

Literature Cited

Barger T, Hladik ML, Daniels JC. 2020. Uptake and toxicity of clothianidin to monarch butterflies from milkweed consumption. PeerJ 8:e8669 DOI 10.7717/peerj.8669.

Bargar T. 2012a. The relationship between total cholinesterase activity and mortality in four butterfly species. Environmental Toxicology and Chemistry 31: 2124-2129.

Bargar T. 2012b. Risk assessment for adult butterflies exposed to the mosquito control pesticide naled. Environmental Toxicology and Chemistry 31:885-891.

Bijleveld M, Lexmond V, Bonmatin J, Goulson D, Noome DA. 2014. Worldwide integrated assessment on systemic pesticides Global collapse of the entomofauna: exploring the role of systemic insecticides. Environmental Science and Pollution Research 22:1-4.

Bonmatin J, Giorio C, Girolami V, Goulson D, Kreutzweiser D, Krupke C, Liess M, Long E, Marzaro M, Mitchell E, Noome D, Simon-Delso N, Tapparo A. 2015. Environmental fate and exposure; neonicotinoids and fipronil. Environmental Science and Pollution Research. 22:35-67. DOI 10.1007/s11356-014-3332-7

Braak N, Neve R, Jones AK, Gibbs M, Breuker CJ. 2018. The effects of insecticides on butterflies - A review. Environmental Pollution. 242:507-518. doi: 10.1016/j.envpol.2018.06.100.

Davis BNK, Lakhani KH, Yates TJ. 1991. The hazards of insecticides to butterflies of field margins. Agr Ecosyst Environ 36:151-161.

DiBartolomeis M, Kegley S, Mineau P, Radford R, Klein K. 2019. An assessment of acute insecticide toxicity loading (AITL) of chemical pesticides used on agricultural land in the United States.PLoS ONE 14(8): e0220029. doi.org/10.1371/journal.pone.0220029.

Dively and Kamel. 2012. Insecticide residues in pollen and nectar of a cucurbit crop and their potential exposure to pollinators. J Agric Food Chem 4449-4456.

Douglas MR, Tooker JF .2015. Large-Scale deployment of seed treatments has driven rapid increase in use of neonicotinoid insecticides and preemptive pest management in U.S. field crops. Environmental Science and Technology 49:5088-5097.

Easton AH, Goulson D. 2013. The neonicotinoid insecticide imidacloprid repels pollinating flies and beetles at field-realistic concentrations. PLoS ONE 8(1):e54819.

Eliazar PJ, Emmel TC. 1991. Adverse impacts to non-target insects. Mosquito control pesticides: ecological impacts and management alternatives. Conference Proceedings. Scientific Publishers Inc., Gainesville, Florida.

[USEPA] U.S. Environmental Protection Agency. 2017. Pesticides industry sales and usage: 2008-2012 market estimates. Office of Chemical Safety and Pollution Prevention, Washington, DC.

Forister M, Cousens B, Harrison J, Anderson K, Thorne J, Waetjen D, Nice C, De Parsia M, Hladik M, Meese R, Van Vliet H, Shapiro A. 2016. Increasing neonicotinoid use and the declining butterfly fauna of lowland California. Biological Letters 12(8):1-5.

Gilburn AS, Bunnefeld N, Wilson JM, Botham MC, Brereton TM, Fox R, Goulson D. 2015. Are neonicotinoid insecticides driving declines of widespread butterflies? PeerJ 3:el402.

Gill RJ, Ramos-Rodriguez O, Raine NE. 2012. Combined pesticide exposure severely affects individual- and colony-level traits in bees. Nature 491:105-108.

Goulson D. 2013. An overview of the environmental risks posed by neonicotinoid insecticides. Journal of Applied Ecology Ecology 50(4):977-987.

Hallmann CA, Foppen RPB, van Turnhout CAM, Kroon H de, Jongejans E. 2014. Declines in insectivorous birds are associated with high neonicotinoid concentrations. Nature 511:1-3.

Halsch C, Code A, Hoyle SM, Fordyce JA, Baert N, Forister ML, 2020. Pesticide Contamination of Milkweeds Across the Agricultural, Urban, and Open Spaces of Low-Elevation Northern California. Frontiers in Ecology and Evolution. Doi: 10.3389/fevo.2020.00162.

Henry M, Béguin M, Requier F, Rollin O, Odoux JF, Aupinel P, Aptel J, Tchamitchian S, Decourtye A. 2012. A common pesticide decreases foraging success and survival in honey bees. Science 336(6079):348–350.

Hoang T, Rand G. 2015. Acute toxicity and risk assessment of permethrin, naled, and dichlorvos to larval butterflies via ingestion of contaminated foliage. Chemosphere 120:714-721.

Hoang T, Pryor R, Rand G, Frakes R. 2011. Use of butterflies as nontarget insect test species and the acute toxicity and hazard of mosquito control insecticides. Environmental Toxicology and Chemistry 30:997-1005.

Huseth AS, Groves RL. 2014. Environmental fate of soil applied neonicotinoid insecticides in an irrigated potato agroecosystem. PLoS ONE 9(5): e97081.

James DG. 2019. A neonicotinoid insecticide at a rate found in nectar reduces longevity but not oogenesis in monarch butterflies, Danaus plexippus (L.). (Lepidoptera: Nymphalidae). Insects 10:276. doi:10.3390/insects10090276.

Jeschke P, Nauen R, Schindler M, Elbert A. 2011. Overview of the status and global strategy for neonicotinoids. Journal of Agricultural and Food Chemistry 59:2897-2908.

Krischik V, Rogers M, Gupta G, Varshney A. 2015. Soil-applied imidacloprid translocates to ornamental flowers and reduces survival of adult *Coleomegilla maculata*, *Harmonia axyridis*, and *Hippodamia convergens* lady beetles, and larval *Danaus plexippus* and *Vanessa cardui* butterflies. PLoS ONE 1-22.

Krishnan N, Zhang Y, Bidne KG, Hellmich RL, Coats JR, Bradbury SP. 2020. Assessing Field-Scale Risks of Foliar Insecticide Applications to Monarch Butterfly (Danaus plexippus) Larvae. Environmental Toxicology and Chemistry. Volume 00, Number 00—pp. 1–19.

Krupke CH, Hunt GJ, Eitzer BD, Andino G, Given K. 2012. Multiple routes of pesticide exposure for honey bees living near agricultural fields. PLoS One 7:e29268.

Mule R, Sabella G, Robba R, Manachini B. 2017. Systematic review of the effects of chemical insecticides on four common butterfly families. Frontiers in Environmental Science 5: 1-5.

Mullin CA, Frazier M, Frazier JL, Ashcraft S, Simonds R, vanEngelsdorp D, Pettis JS. 2010. High levels of miticides and agrochemicals in North American apiaries: Implications for honey bee health. PLoS ONE 5(3): e9754. https://doi.org/10.1371/journal.pone.0009754.

Oberhauser K, Brinda S, Weaver S, Moon R, Manweiler S. 2006. Growth and survival of monarch butterflies (Lepidoptera: Danaidae) after exposure to permethrin barrier treatments. Environmental Entomology 35:1626-1634.

Oberhauser K, Manweiler S, Lelich R, Blank M, Batalden R, de Anda A. 2009. Impacts of ultra-low volume resmethrin applications on non-target insects. Journal of the American Mosquito Control Association 25:83-93.

Olaya-Arenas P, Kaplan I. 2019. Quantifying pesticide exposure risk for monarch caterpillars on milkweeds boarding agricultural land. Frontiers in Ecology and Evolution 7:223. doi: 10.3389/fevo.2019.00223.

Pecenka J, Lundgren J. 2015. Non-target effects of clothianidin on monarch butterflies. Science of Nature 102:19.

Roessink I, Merga LB, Zweers HJ, Van den Brink PJ. 2013. The neonicotinoid imidacloprid shows high chronic toxicity to mayfly nymphs. Environmental Toxicology and Chemistry 32:1096-1100.

Salvato 2001. Influence of mosquito control chemicals on butterflies (Nymphalidae, Lycaenidae, Hesperiidae) of the lower Florida Keys. Journal of the Lepidopterists' Society 55: 8-14.

Starner K, Goh KS. 2012. Detections of the neonicotinoid insecticide imidacloprid in surface waters of three agricultural regions of California, USA, 2010-2011. Bulletin of Environmental Contaminants and Toxicology 88:316-321.

Stoner KA, BD Eitzer. 2012. Movement of soil-applied imidacloprid and thiamethoxam into nectar and pollen of squash (Cucurbita pepo). PLoS ONE 7(6):e39114. doi: 10.1371/journal.pone.0039114. Epub 2012 Jun 27.

Thogmartin W, Wiederholt R, Oberhauser K, Drum R, Diffendorfer J, Altizer S, Taylor O, Pleasants J, Semmens D, Semmens B, Erickson R, Libby K, Lopez-Hoffman L. 2017. Monarch butterfly population decline in North America: identifying the threatening processes. Royal Society Open Science 1-16.

Whitehorn PR, O'Connor S, Wackers FL, Goulson D. 2012. Neonicotinoid pesticide reduces bumble bee colony growth and queen production. Science 336:351–352.

Zhong H, Hribar LJ, Daniels JC, Feken MA, Brock C, Trager MD. 2010. Aerial ultra-low-volume application of naled: Impact on nontarget imperiled butterfly larvae (*Cyclargus thomasi bethunebakeri*) and efficacy against adult mosquitoes (*Aedes taeniorhynchus*). Environmental Entomology 39:1961-1972.

Supplemental Materials 1b for the Monarch (*Danaus plexippus plexippus*) Species Status Assessment Report, Revised November 2019

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The Risk of Exposure and Direct Toxicity of Herbicides to the Monarch Butterfly

Herbicides are widely used throughout the range of the monarch, and can cause mortality and reduced vitality to milkweed host plants and nectar source plants. However, plants may survive exposure if the herbicide has no toxicity to the plant (e.g., it is selective only for certain plants) or if concentrations are not high enough to elicit an effect (e.g., exposure from drift). In these cases, monarch caterpillars may retain use of the host or forage plants, but may be directly exposed to herbicides through contact or diet.

As with insecticides, the potential for direct effects of herbicides on monarchs can vary by active ingredient, product additives used (e.g. surfactants), exposure pathway, life history phase exposed, timing of application, and the amount of chemical exposed to the monarch. Herbicides work by interacting with the cellular structure or biochemical pathway of the target plant, and by causing tissue damage and plant mortality. Some herbicides are enzyme inhibitors acting on the enzymes that are important for plant growth and development. Although the mode of action for herbicides is to target specific pathways for plants, there are similarities between some plant enzymes that herbicides target and insect enzymes. For example, some herbicides target acetyl CoA carboxylase in plants, an enzyme important for plant growth but also for protein synthesis in insects (Lou et al. 2001, Goldring and Read 1993). Other herbicides can target glutamine synthetase, an enzyme critical for photorespiration in plants and ammonia detoxification and reassimilation in insects (Kutlesa and Caveney 2001). For the vast majority of herbicides, the mode of action and influence on lepidopteran biological systems remains unknown.

We are unaware of published data testing the direct effects of herbicides to monarchs. This section provides an evaluation of the risks of herbicides to monarchs based on a brief summary of herbicide-lepidopteran toxicity studies; it does not include an exhaustive review of the available science.

Herbicide concentrations in milkweed leaves

As with insecticides, oral exposure of monarchs to herbicides is dependent on residues being present on or within dietary items. Olaya-Arenas and Kaplan (2019) detected herbicides in leaves of milkweed (*A. syriac*a) within 100 m of crop fields in northwest Indiana. Atrazine was the most frequently detected herbicide, in 80-87% of the samples and at the highest concentrations (2015: 6.84 ng/g mean, 0.52 ng/g median, 238.7 ng/g maximum; 2016: 37.0 ng/g mean, 4.73 ng/g median, 1352.9 ng/g maximum), followed by s-metolachlor (in greater concentrations early in season) and acetochlor.

Herbicide toxicity to lepidopterans

Studies suggest that the active ingredients in some herbicide formulations have the potential to cause lethal and sublethal effects in lepidopterans under certain exposure scenarios. Schultz et al. (2016) tested the direct effects of graminicides fluazifop-p-butyl, sethoxydim, clethodim mixed with the adjuvant NuFilm on three *Euphydryas* species in the 2nd instar larval phase under two different scenarios. In the first experiment, *E. colon* larvae were directly exposed to the treatments at labeled rates for habitat types which could be treated for invasive plants, placed in individual rearing containers, and fed until entering diapause. Control groups received a NuFilm only treatment and a water only treatment. This experiment found that contact treatment with sethoxydim reduced survivorship of pre-diapause *E. colon* larvae by 20% compared to the water only control, while there was no observed effect to larval survival from fluazifop-p-butyl, clethodim, and the NuFilm treatments. In the second experiment, all three *Euphydryas* species were exposed to fluazifop-p-butyl mixed with NuFilm; hostplants were also treated, with larvae and host plants placed within a mesocosm study design. Survival, larval development time, and feeding behavior were observed. This experiment found no effects of fluazifop-p-butyl on larval survival or development time; however, feeding group size (number of gregarious larvae) was reduced by exposure to the herbicide.

Stark et al. (2012) examined the individual effects of three formulated herbicide products containing triclopyr (Garlon 4 Ultra - a selective herbicide used to control woody plants and broad leaved plants), sethoxydim (Poast - a selective herbicide used to control grasses), and imazapyr (Stalker - a non-selective herbicide used to control grasses) directly applied to 1st instar Behr's metalmark (*Apodemia virgulti*) and their food source (buckwheat) at labeled field rates. Larvae were then fed treated plants and allowed to develop into adults. Triclopyr, sethoxydim, and imazapyr products each reduced the number of pupae (and consequently the number of adults) produced compared to the control by 24%, 27%, and 36%, respectively.

To investigate the most likely and worst case scenarios for herbicide exposure to lepidopterans, Russell and Schultz (2009) assessed the biological effects of two herbicides to the 3rd instar phase of the Puget blue (*Icaricia icarioides blackmorei*) and the cabbage white (*Pieris rapae*). The timing of the 3rd instar larval phase corresponds to when herbicides are most likely to be used in the field. Survival, development time, and growth were measured in the larvae after the exposure of two grass-specific herbicides and one surfactant (Preference) in mixtures: fluazifop-*p*-butyl and surfactant, sethoxydim and surfactant, fluazifop-*p*-butyl and water, sethoxydim and water, a water control, and an untreated control. A backpack sprayer was used to administer the treatments to simulate ground application; maximum labeled spot spraying recommended rates were applied. To test most likely scenarios, larvae were placed on host plants (*Lupinus albicaulis*) and the herbicide mixtures for each treatment were directly sprayed on the plants. Larvae were exposed to the residues via contact and dietary exposure. To test for the worst case scenario, larvae and the host plants were separately sprayed with the herbicide mixtures and the larvae were then placed on the plant to simulate maximum direct contact and dietary exposure. The study found that survival was reduced for *P. rapae* (but not for *I. i. blackmorei*) when exposed to fluazifop-*p*-butyl plus surfactant (21% reduction) and sethoxydim plus surfactant (32% reduction) compared to the control. Development time to eclosion for *I. i. blackmorei* occurred earlier in all treatment groups compared to the controls, but this was not observed for *P.*

rapae. Wing area was smaller for female *P. rapae* when exposed to fluazifop-*p*-butyl plus surfactant (10% reduction) and sethoxydim plus surfactant (14% reduction) compared to the controls. Males exhibited a 9% reduction in total wing area in the sethoxydim plus surfactant treatment.

Kutlesa and Caveney (2001) found the herbicide glufosinate-ammonium (GLA), a non-selective post-emergence contact herbicide that competitively inhibits the enzyme glutamine synthetase, to cause lethality to Brazilian skippers (*Calpodes ethlius*) from dietary exposure from concentrations calculated to be similar to field application rates. 5th instar caterpillars were placed in petri dishes on moistened filter paper and fed leaf discs from the plant species Canna lily that were treated with acute doses of GLA to determine an LD50. Each caterpillar received one treated leaf disc and were observed until it was completely consumed (approximately 24 hours) and then provisioned with untreated leaves until pupation or death. The LD50 for GLA was calculated to be slightly lower than expected residues on leaves after field application. For behavioral studies, caterpillars were fed leaves that had high and low concentrations of GLA and mass and general behaviors were recorded daily. A decline in normal activity was observed 2-3 days after treatment with a daily dose of 5 mmol and the caterpillars stopped feeding altogether after 3-4 days. Multiple normal behaviors were observed to be altered and the caterpillars died after 6-7 days after exposure.

Bohnenblust et al. (2013) did not detect toxic effects of dicamba via contact or dietary exposure to 2nd and 3rd instar larvae of the corn earworm (*Helicoverpa zea*) and the painted lady (*Vanessa cardui)*. In contact exposure studies, larvae were placed in treatments and topically dosed with dimethylamine (DMA) and diglycolamine (DGA) formulation of dicamba within a range of the field application rate and placed in individual 50-mm petri dishes. Larvae were not provisioned during the toxicity studies and mortality was assessed at 4, 8, 12, 24, and 48-hour exposure durations. Percent mortality was equal across all treatments indicating that dose had little effect on survival for both species. To assess dietary exposure on the growth and development of *H. zea* and *V. cardui* larvae, soybean (*Glycine max*) and nodding plume thistle (*Carduus nutans*) were exposed to DMA formulation using a research grade automated sprayer at four rates that represent a range of 0.0001-0.1 of the current label rate of dicamba. After spraying, plants were isolated by treatment in a greenhouse. After three days, starved larvae (24 hours with no food provisions) were placed on the treated plants (*H. zea* on soybean and *V. Cardui* on thistle) and monitored until pupation or death. No differences in *H. zea* larval survival were detected across treatments and there was no relationship detected between number of days to pupation and herbicide dose. In the tests using thistle and *V. cardui* larvae, reductions in larval and pupa mass were observed.

LaBar and Schultz (2012) did not observe lethal or sublethal effects in a field study in which the habitat of the Puget blue was sprayed with sethoxydim and a non-ionic surfactant. During observational data collections, there was little to no observed impact on larval performance in the field or on oviposition for adults in the sprayed fields compared to non-treated fields.

Summary: Risk of Herbicides to Monarchs

In the herbicide toxicity studies summarized above, results suggest that various types of herbicides may result in direct effects to lepidopterans if exposed at recommended field application rates for the labeled land use/cover type. In several studies, the simulated application site was some type of conservation area where chemical control of invasive plants was presumed, resulting in maximum exposure of herbicide to lepidopteran. It is important to note that we found no studies evaluating the effects of herbicides to lepidopterans at concentrations representative of exposure due to drift from an application site to nearby habitat (i.e., exposure concentrations at less than a maximum labeled rate) for this risk assessment.

For those herbicide-lepidopteran toxicity studies in which effects were observed, reductions in survival were generally between 20-40% of the exposed population. Effects were detected in a variety of herbicide types, including those that are non-selective, as well as those that are selective for monocots or dicots. However, results of these studies are mixed, and in a number of cases, no direct effects were found to lepidopterans from specific herbicides or particular exposure regimes.

In summary, herbicides have been detected in milkweed plants growing in proximity to agricultural fields and larval monarchs can be exposed by ingesting residues that are expressed in plant tissues; however, the direct effects of most herbicides to monarchs are unknown and likely to be highly variable. The toxicological information presented above represents a small percentage of all herbicide products used, and does not account for the most widely used herbicides such as glyphosate, atrazine, metolachlor, and 2-4 D. For those herbicides in which direct effects were detected, we are unable to elucidate the extent or specific circumstances of their use within the monarch range. While we acknowledge the potential for toxic effects of herbicides to monarchs under certain exposure conditions, we consider the effects of insecticides to be the primary driver in monarch impacts due to pesticides (insecticides, herbicides, fungicides, rodenticides, etc.).

Literature Cited

Bohnenblust E, Egan JF, Mortensen D, Tooker J. 2013. Direct and Indirect Effects of the Synthetic-Auxin Herbicide Dicamba on Two Lepidopteran Species. Environmental Entomology. 42: 586-894.

Goldring JP, Read JS. 1993. Insect acetyl-CoA Carboxylase activity during the larval, pupal, and adult stages of insect development. Comparative Biochemistry and Physiology Part B: Comparative Biochemistry 106: 855-858.

Kutlesa NJ, Caveney S. 2001. Insecticidal activity of glufosinate through glutamine depletion in a caterpillar. Pest Manag Sci 57:25–32

LaBar C, Schultz CB. 2012. Investigating the effects of grass-specific herbicides on non-target butterflies. Natural Areas Journal 32:177-189.

Lou X, Matsumoto H, Usui K. 2001. Comparison of physiological effects of fluazifop-butyl and sethoxydim on oat (*Avena sativa* L.). Weed Biology and Management 1:120-127.

Mullin, CA, Frazier M, Frazier JL, Ashcraft S, Simonds R, VanEngelsdorp D, Pettis JS. 2010. High levels of miticides and agrochemicals in North American apiaries: Implications for honey bee health. PLoS One 5, e9754.

Olaya-Arenas P, Kaplan I. 2019. Quantifying pesticide exposure risk for monarch caterpillars on milkweeds boarding agricultural land. Frontiers in Ecology and Evolution 7:223. doi: 10.3389/fevo.2019.00223.

Russell C, Schultz CB. 2009. Effects of grass-specific herbicides on butterflies: An experimental investigation to advance conservation efforts. Journal of Insect Conservation. DOI: 10.1007/s10841-009-9224-3

Schultz CB, Zemaitis JL, Cameron, Thomas M, Bowers D, Crone EE. 2016. Non-Target Effects of Grass-Specific Herbicides Differ among Species, Chemicals and Host Plants in *Euphydryas* Butterflies. Journal of Insect Conservation. October 2016, 20: 867–877.

Stark JD, Chen XD, Johnson CS. 2012. Effects of herbicides on Behr's metalmark butterfly, a surrogate species for the endangered butterfly, Lange's metalmark. Environmental Pollution 164: 24-27.

Pearl River Basin, Mississippi, Federal Flood Risk Management Project

Annex D4 - Project Descriptions

June 2024

CONTENTS

LIST OF TABLES

LIST OF FIGURES

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Section 1

ALTERNATIVE A1: USACE NONSTRUCTURAL PLAN

The nonstructural analysis was based on an inventory of residential and non-residential structures that was developed by USACE in 2023 using the National Structural Inventory version 2.0. An assessment of structures located in the 10 percent, 4 percent, 2 percent, and 1% AEP floodplains was performed for the portions of the study are subject to flooding from the main stem of the Pearl River and backwater flooding on the tributaries (Figures 3-1a through 3-1d and Table 3-2). Elevation and floodproofing was considered to determine the effectiveness of a nonstructural alternative. For the analysis, residential structures were to be elevated to the 1% AEP/BFE plus one foot, up to 13 feet above the ground, and nonresidential structures were to be floodproofed up to 3 feet above the ground. All nonstructural components would be implemented on a voluntary basis in cooperation with the property owner. The assumption is that there would be 100 percent participation rate; however, for socially vulnerable areas the participation rate based on similar USACE projects, such as Huntington District Section 202 program is that approximately a 50 percent participation rate is typically realized.

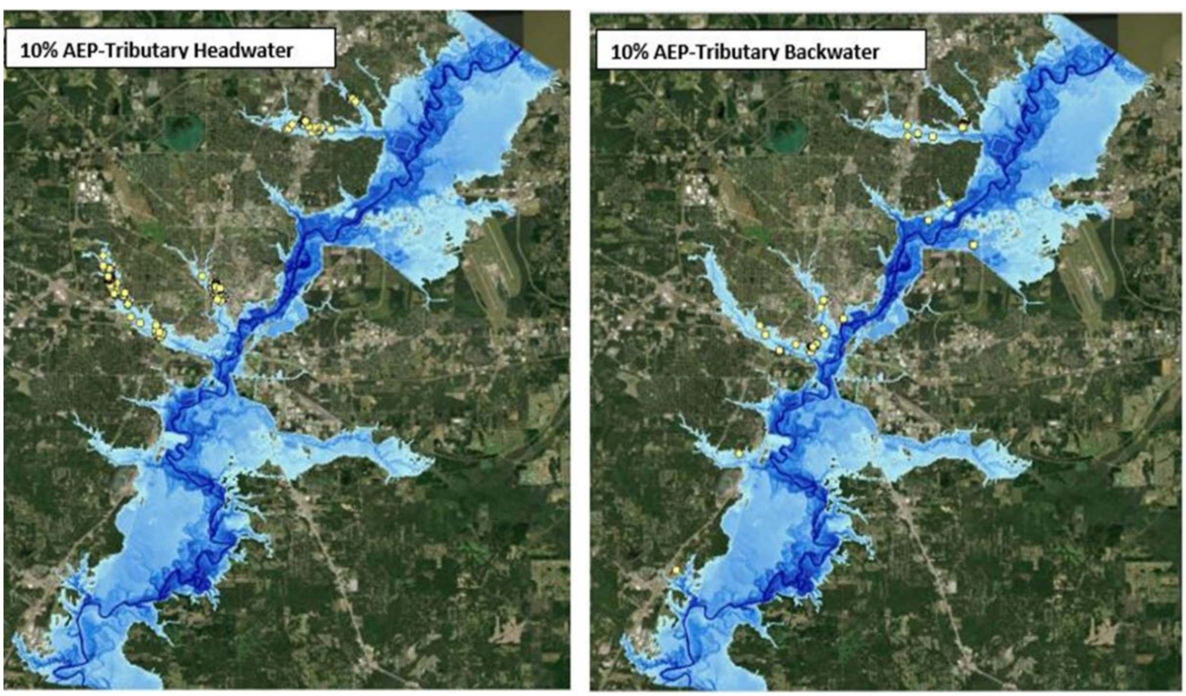

Figure I-1a. Structures inundated from a Cumulative 10% AEP Event separated by Headwater and Backwater Flooding

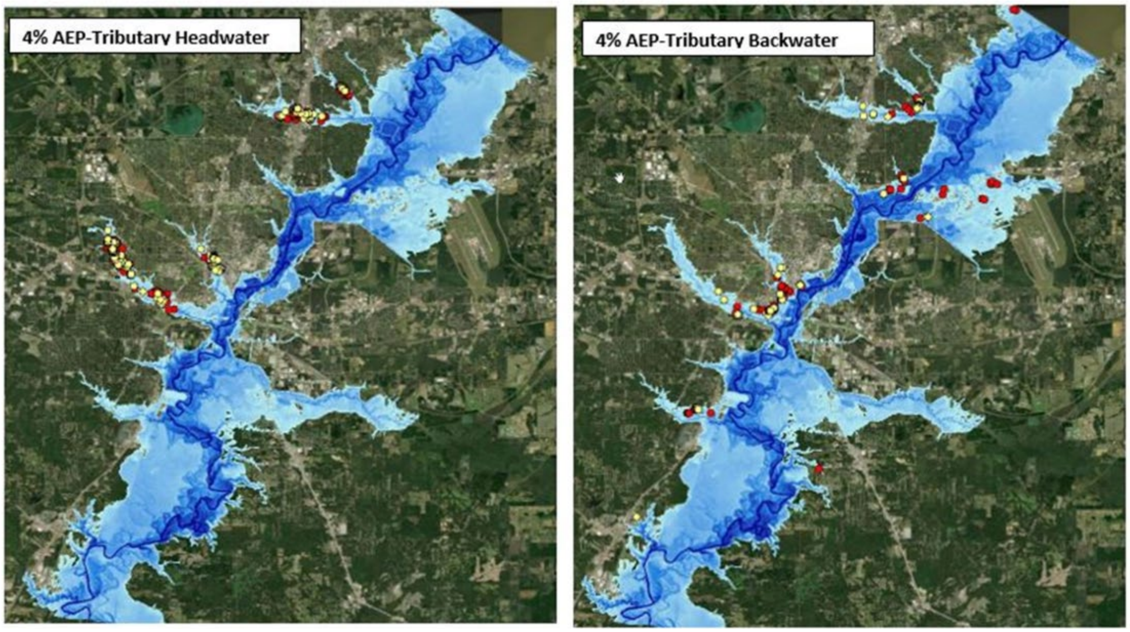

Figure I-1b. Structures Inundated from a Cumulative 4% AEP Event Separated by Headwater and Backwater Flooding

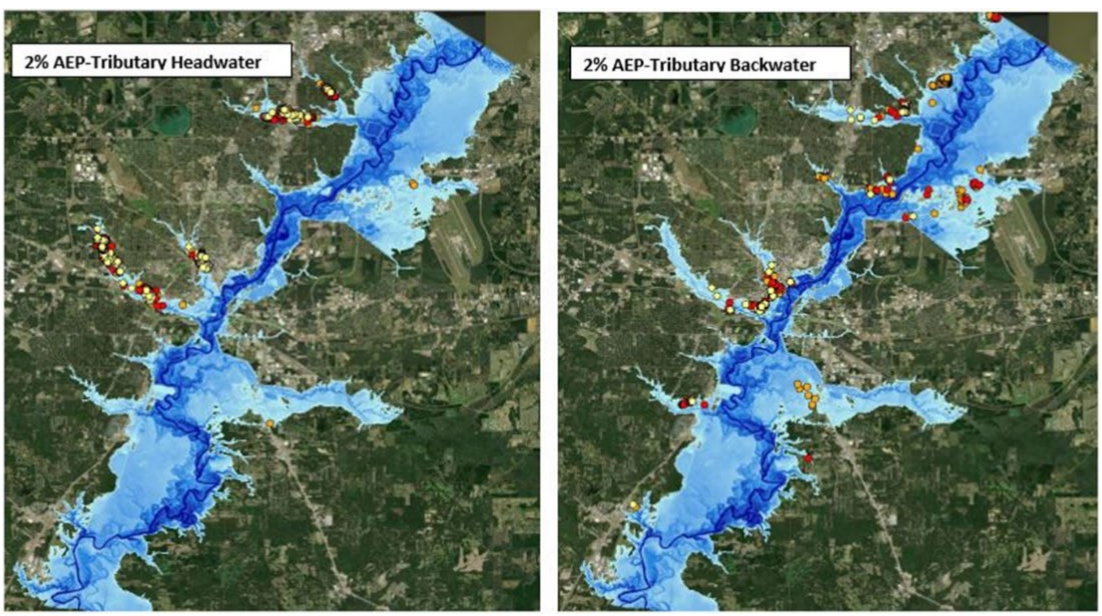

Figure I-1c. Structures Inundated from a Cumulative 2% AEP Event Separated by Headwater and Backwater Flooding

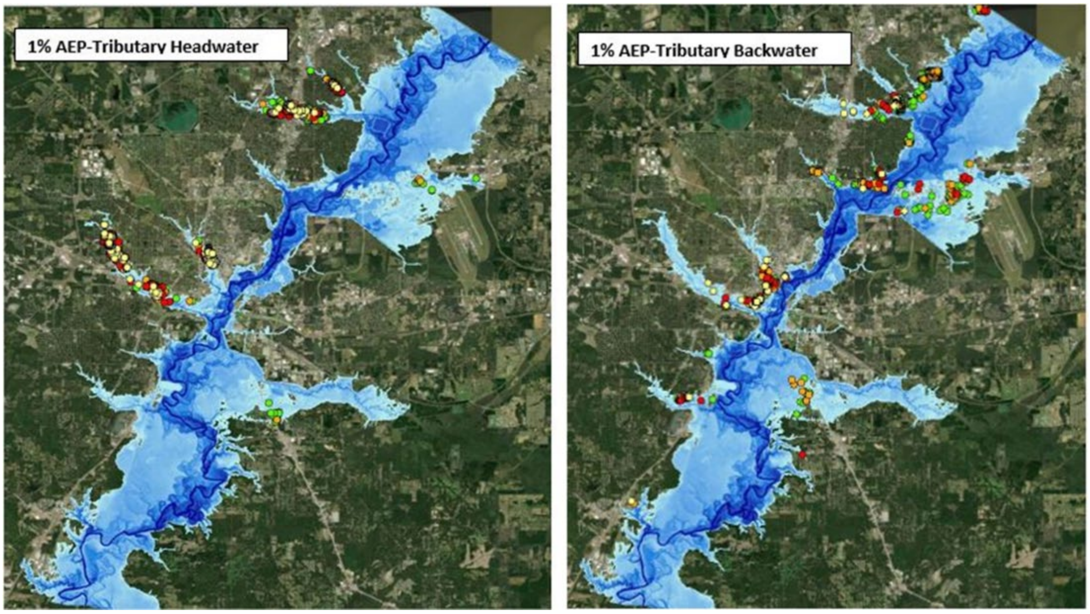

Figure I-1d. Structures Inundated from a Cumulative 1% AEP Event Separated by Headwater and Backwater Flooding (Colored dots represent structures in the following AEP floodplains: green dots are 1% (100 year), orange dots are in the 2% (50 year), red dots are within the 4% (25 year) and yellow dots are within the 10% (10 year)).

Table I-1. Noncumulative Nonstructural Benefits for Study Area for Elevating and Floodproofing, FY24 Price Level and Discount Rate

	(10%AEP)	(4% AEP)	(2% AEP)	(1% AEP)
Project First Cost	18,967,742	31,105,161	76,799,092	154,076,828
Interest During Construction	64,430	105,659	260,874	523,374
Total Investment Cost	19,032,173	31,210,821	77,059,967	154,600,203
AA Investment Cost	704,969	1,156,077	2,854,372	5,726,534
Benefits EAD Reduced	2,259,000	1,751,000	1,793,000	1,466,000
Net Benefits	1,554,031	594,923	(1,061,372)	(4,260,534)
B/C Ratio	3.2	1.5	0.6	0.3

Based on an incremental floodplain analysis, the 10 percent and 4 percent incremental AEP floodplains were both economically justified. Approximately 143 structures, 81 residential and 62 nonresidential, are included in this cumulative 4 percent AEP floodplain. The cumulative results of the 4 percent AEP floodplain are displayed in Table I-2.This nonstructural plan is referred to as Alternative A1.

Table I-2. Summary of Results for Alternative A1, the USACE modified Nonstructural Plan, FY24 Price Level and Discount Rate

Project First Cost	$50,072,903
Interest During Construction	$170,090
Total Investment Cost	$50,242,993
AA Investment Cost	$1,861,000
Total AA Cost	$1,861,000
Benefits EAD Reduced	$4,010,090
Net Benefits	$2,149,090
B/C Ratio	2.2

These structures have been identified to be preliminarily eligible for the nonstructural alternative. Due to feedback from public meetings in May and June 2023 requesting the option to have properties acquired, the option of nonstructural property acquisition (buyout) on a voluntary basis is included in the nonstructural implementation plan (Appendix K). In addition, 10 of the 600 structures are located within the FEMA Regulated Floodway and would only be eligible for demolition or relocation. Structures located within the FEMA Regulated Floodway, based on preliminary analysis, have relatively similar flood risk in comparison to structures located outside of the FEMA Regulated Floodway.

Table I-3. Nonstructural Plan A1 Structure Type Eligibility

Structure Type	**Public**	**Private-Non-Profit**	**Residential-Non-Historic**	**Residential-Historic**	**Nonresidential**
Property Acquisition & Structure Demolition	x	x	x	x	x
Property Acquisition & Structure Relocation	x	x	x	x	x
Structure Elevation			x	x	x
Structure Dry floodproofing				x	x
Structure Wet floodproofing			x	x	x

Retrofitting of Existing Buildings			x	x	x

NON-STRUCTURAL IMPLEMENTATION APPROACHES

Property Acquisition and Structure Demolition

Property acquisition and structure demolition consists of the acquiring the existing at-risk structure and, typically, the underlying land, and conversion of the land to open space through the demolition of the structure. The property must be deed-restricted in perpetuity to open space uses to restore and/or conserve the natural floodplain functions.

Property Acquisition and Structure Relocation

Property acquisition and structure relocation consists of the physical relocation of an existing structure to an area outside of a hazard-prone area and, typically, the acquisition of the underlying land. Relocation must conform to all applicable State and local regulations. The property must be deed-restricted in perpetuity to open space uses to restore and/or conserve the natural floodplain functions.

Elevation

Elevation is physically raising an existing structure to an elevation to the 1 percent AEP BFE based on year 2082 hydrology or higher if required by USACE or local ordinance. Foundations must be designed to properly address all loads and effects, be appropriately connected to the floor structure above, and utilities must be properly elevated.

Dry Floodproofing

Dry floodproofing is using techniques applied to keep non-residential structures dry by sealing the structure to keep floodwaters out. Dry flood proofing would be completed on eligible structures at or below 3 feet (0.9m) depth.

Wet Floodproofing

Techniques designed to permit floodwaters to enter a structure to prevent or provide resistance to damage from flooding. Wet Floodproofing of a structure interior is intended to counteract hydrostatic pressure on the walls, surface, and support systems of the structure by equalizing interior and exterior water levels during a flood.

Retrofitting of Existing Buildings

Modifications to the structural elements of a building to reduce or eliminate the risk of future flood damage and to protect inhabitants. The structural elements of a building that are essential to protect to prevent damage include foundations, load-bearing walls, beams, columns, structural floors and roofs, and the connections between these elements. Retrofitting also includes modifications to the nonstructural elements of a building or facility to reduce or eliminate the risk of future damage and to protect inhabitants. Retrofits are primarily defined as modifications to the elements of a building to reduce or eliminate the risk of future damage. Structural retrofits are designed to protect elements such as foundations, load-bearing walls, beams, columns, building envelopes, windows, structural floors, roofs, and the connections between these elements. Nonstructural retrofitting involves the modification of a building or facility's nonstructural elements and may include elevation of heating and ventilation systems to minimize or prevent flood damage.

Section 2

ALTERNATIVE C: NFI CHANNEL IMPROVEMENT/WEIR/LEVEE PLAN PROJECT DESCRIPTION

Flood risk management benefits are realized by removing areas that constrict the floodplain by deepening the channel and floodplain. By doing this, conveyance of water downstream is improved through the project area. The water surface elevation of the river would be lowered in some places by as much as 8 feet (2.4 m) within the project area. Normal river stages would be permanently elevated. Flood elevations would be reduced within the reach of excavation and upstream of the excavation. Alternative C (Figure I-2 and Table I-4) consists of the construction of channel improvements, demolition of the existing weir near the J. H. Fewell WTP site and construction of a new weir with a low-flow gate structure further downstream for water supply to be continued while simultaneously creating an area of surface water for recreational opportunities, Federal levee improvements (excavated material plan), and upgrading an existing non-Federal ring levee with slurry wall around the Savannah Street WWTP.

Construction of the project would require relocations and/or improvements to various public and private utilities and infrastructure, (Table I-5), avoidance and minimization features required under the ESA, and the creation of new habitat mitigation areas to offset losses within the project's construction footprint areas.

There are 9 transmission lines within the project area. All efforts would be made to avoid, monitor, maintain clearance requirements, and protect these structures. If avoidance is not possible, then utility relocation or raising of lines/protection of structures would be necessary. It is estimated that 5 to 6 of these lines will require additional utility relocation costs. Coordination with the operating entity to determine specific requirements of each transmission line will be conducted during PED.

USACE modeling of Alternative C considered a variety of upgrades to the NFI routing. These included calibration to the recent 2020 flood event, which had not occurred at the time of NFI modeling, incorporating more recent flow record data (1980s to 2022), updating all runs to unsteady state routing, inclusion of tributary coincident flow, and the inclusion of lateral structures to represent the levees (Figure I-3). Updated calibration has shown that the system response has changed since the 1979 event to be more efficient. as illustrated by the comparable events from 1983 and 2020. The two events had similar flows at Pearl River gage in Jackson, but the stage was reduced by approximately 2.9 ft for the 2020 event.

Table I-5. Alternative C Project Key Features

Feature	ALT C		Units
	Quantity NFI (211 report)	Quantity USACE	
NON-STRUCTURAL			
Non-structural plan	acquisition		structures
STRUCTURAL			
Lake Surface Water Area	1700	2562.25	acres
Clearing and Grubbing	2,600	2301.39	acres
Channel Improvements Excavation	1400	1443.25	acres (mcy)
Fill Area	870	858.14	acres (mcy)
Stabilization or armoring for bridge abutments	10	7	bridges
Hard Point in tributary channels to prevent incision/sediment into newly constructed lake		850	Feet (crossing river)
Newly Federalized Levee (inc. slurry wall	1.7	1.7	miles
Slurry Wall Savanna Street WWTP	1.7	1.7	miles
New Slurry wall for seepage of existing features	n/a	1,460 ft	miles
Weir and new gate	1	1	each
Pumps to address interior drainage Impacts	0	2	each
Fish Passage	7000	7000	feet
Canton Club Levee	n/a	n/a	miles
OPERATIONS AND MAINTENANCE			
Weir	Unknown		each
Fish Passage			each
Terrestrial Habitat Mitigation			events
Riverine Habitat Mitigation			events
Lake			each
Pump Station			each
Levees			each
MITIGATION			
Sandbars (material from excavation)	31	NA	acres
Reforest top bank of fish passage	?	?	acres
Riverbank preservation	10	NA	miles
Removal of obsolete aquatic barriers	0	1	structure
connect occupied and suitable unoccupied riverine habitat	0	NA	acres
Open historically lost riverine habitat	0	NA	acres

Terrestrial Habitat Mitigation	5,000	24,760	acres

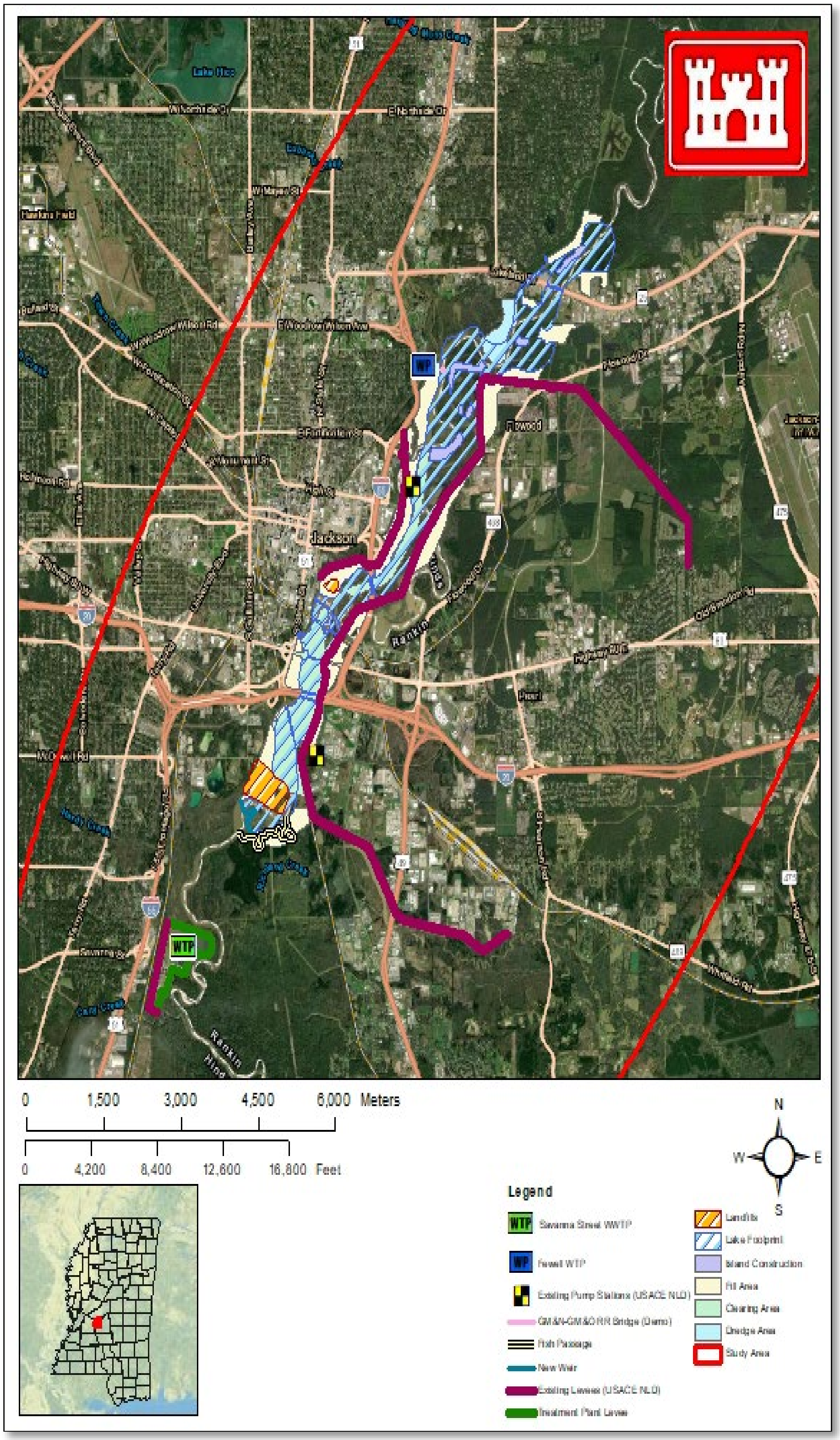

Figure I-2. Alternative C Key Features

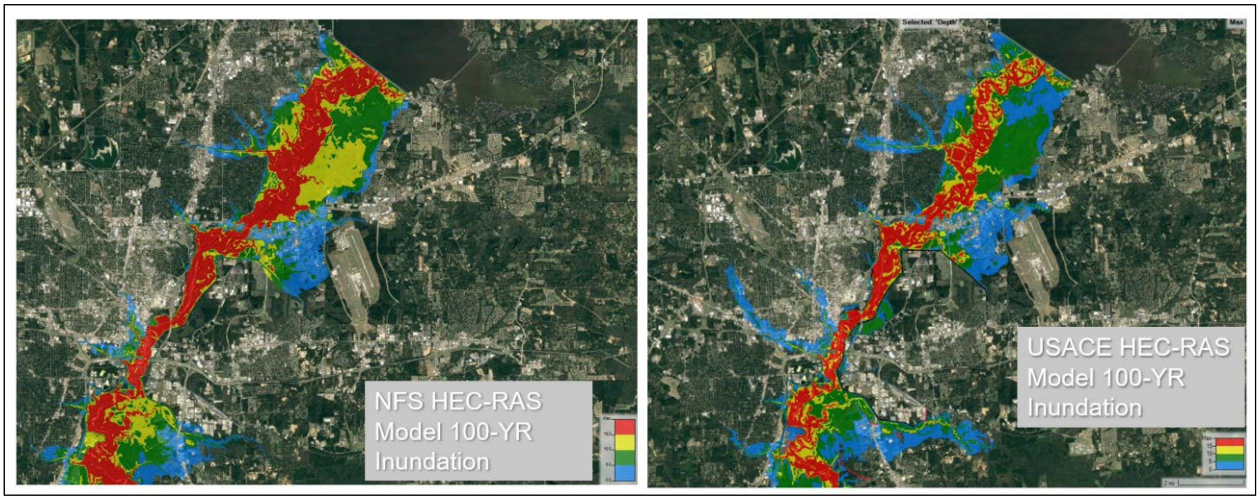

Figure I-3. NFI versus USACE modeling Results for the 1% AEP (100-year) Without Project Routing Scenario

Channel Improvements

Channel improvements (Figure I-4) consist of excavating areas along the Pearl River to improve conveyance from RM 284 to 294. The channel improvement footprint includes approximately 2,557 acres (1034.7 hectares (ha)) in which disturbance would occur. The excavation would be of various widths ranging from 400 to 2,000 feet (121.9-609.6 m) to be determined during the PED phase. Excavation depths would vary between 5-20 feet to meet the proposed bottom elevation of 248.0 NGVD. This total includes 1,692 acres (684.7 ha) in which excavation would occur to deepen the channel overbanks and 865 acres (350.0 ha) that would be used for placement of the excavated fill material. Approximately 20 million cubic yards (19.1 million m^3) of material would be excavated from the floodplain and channel overbanks. The existing river channel would not be widened, instead excavation of the overbank areas would occur.

The preliminary project layout also includes islands within the channel improvement excavation area that would be maintained and/or expanded upon from RM 289.5 to RM 292.0. Further, sand bars would be constructed inside the floodplain and along the existing islands to compensate for the loss of sand bar habitat.

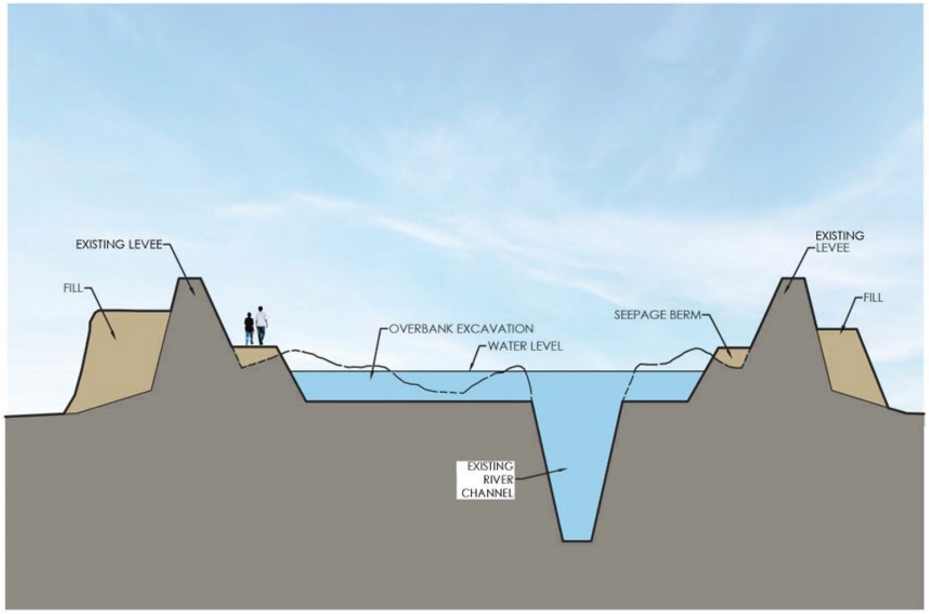

Figure I-4. Channel Improvements with a Relocated Weir

OVERBANK MODIFICATIONS

The existing overbank areas of the Pearl River channel would be lowered to increase conveyance of flood flows. Existing levees would remain in place and would be maintained for flood control and to aid in haul access. The excavation limits near the existing levees would be determined during final design.

The progression upstream would naturally allow for positive and continued dewatering of flooded areas ahead of moving into the next section. The three segments and their main areas of activity are further described in these stationed reaches listed below.

1. <u>Station 10+00 through 140+00</u>. Specific items included in this reach are the I-20 Interstate bridges (Sta. 95+00±) as well as the U.S. Highway 80 (Sta. 110+00), Old Brandon Road (Sta. 135+00±), and railroad bridges (Sta. 70+00±, Sta. 130+00±). Two high-pressure gas lines run through this reach and would have to be carefully monitored as excavation and grading activities progress. Multiple access points on both sides of the river would have to be maintained and monitored from a perspective of public safety and construction use. Projected quantities for earthwork are approximately 6 million cubic yards (yd³).

2. Station 140+00 through 290+00. This reach contains the eastward expansion of the east side levees and the construction of islands in the deepened overbank. Islands would be formed as part of the excavation activities. As with the previous reach segment, numerous access points would require management and maintenance for use and safety. A creosote slough area (Sta. 240+00±) would be avoided, when possible, to not disturb or cause any objectionable material to be exposed or mixed with other excavated material. In the event avoidance is not possible, the slough area may be excavated and hauled to a separate disposal site, and the remaining exposed surface capped prior to final grading. Projected excavation quantities are 6 million yd^3.

3. Station 290+00 through 400+00. As with the previous downstream reaches, there are bridges to work around (Highway 25 near Sta. 360+00), and gas lines and transmission lines that must be monitored during earthmoving operations. Depending on the final design, Mayes Lake (Sta. 310+00±) may need tie-in work to maintain its current level. A determination about the tie-in work would be made during the PED phase. An existing abandoned railroad embankment of the Gulf, Mobile & Northern/Gulf Mobile and Ohio (GM&N/GM&O) Railroad Bridge could also be affected and was removed in H&H modeling. Some island forming work would be required in this reach. The existing weir at the water works bend near Station 290+00 would remain undisturbed until completion of the new weir at the downstream terminus as to maintain water supply for the treatment plant. Projected excavation quantities in this reach are approximately 8 million cubic yards.

Hardpoints at Base of Tributaries

Multiple tributary inflow points exist within this reach and Alternative C would add a hardpoint, via a riprap chute to prevent backward erosion at each tributary inflow where the excavation of overbanks decreased the tributary channel bottom elevation at or near the confluence of those tributaries with the Pearl River.

Maintenance and Reinforcement of Bridge Abutments of Bridges (if required)

Stabilization or armoring, such as riprap, slope paving, slide repairs, etc., is required to ensure structural of integrity of various bridge structures due to changed conditions with this alternative. This work will be carried out prior to clearing and any major channel work. Following its own analysis, the Mississippi Department of Transportation (MDOT) has informed the Rankin-Hinds Flood Control District (the Flood Control District), MDOT agrees to collaborate with the Flood Control District in “the advancement of this project and to ensure countermeasures are included, if determined necessary during the future design process.” (Letter to G. Rhoads, dated February 26, 2024) To this end, the Flood Control District developed a range of cost estimates for potential structural and hydraulic countermeasures that could be recommended if countermeasures are determined necessary. The array of countermeasure features analyzed will mitigate potential impacts to MDOT bridges that will be identified during the PED phase. The estimated cost for these features is based upon known costs for the construction of hydraulic and structural countermeasures on another MDOT project at downstream hydraulic crossings of the Pearl River. When additional information becomes available during PED, adjustments to the design can and will be made to reduce potential impacts. Any proposed countermeasure design and implementation will be conducted with MDOT's concurrence, review, and approval.

Rough estimations of the level of effort required to mitigate for bridge impacts include improvements for approximately 36 bents, 12 piers, abutment scour, as well as funding to conduct monitoring surveys. A pile is a concrete post that is driven into the ground to act as a leg or support for a bridge. A bent is a combination of the cap and the pile. Together, with other bents, act as supports for the entire bridge.

There are a total of 2 active railroad bridges within the project area. All efforts would be made to avoid, monitor, and protect these structures. Additional modeling is required to validate these assumptions during PED. If avoidance is not possible, then coordination with the operating entity to determine specific requirements of each railway bridge will be conducted during PED. All alterations of railroad bridges would be in accordance with Section 3 of the 1946 Flood Control Act (22 USC 701p).

Description of work is consistent for both Alternative C and CTO. The difference is that the extent of improvements for the selected structures would be expected to be larger for the Alternative C.

Excavated Material Plan

Federal levees exist within much of this reach and Alternative C would use the existing levees, upgraded with excess excavation placed behind them. Excavated fill would be placed in designated disposal areas on the protected side of existing levees. These areas would be graded to be at the same elevation or lower than existing levees and grassed to establish long-term erosion control. Additional riprap or other armoring would be placed as required during the final grading operations.

The excavated material disposal fill areas placed on the protected side of levees would impact approximately 465.6 acres (188.4 ha) (Figure I-5). Clearing of wooded areas to the east of the proposed new banks (small areas on the west side) would be cleared and grubbed ahead of receiving excavated material from the channel overbank excavation. The excavated material would be used to create a substantial new land mass within the Jackson MSA. The new land mass created behind the levees would range from 200 to over 1,000 feet (121.9-304.8 m) in width. The newly created riverfront area would allow for expanded riverfront access, natural areas, and commercial development, along with recreational opportunities.

If any structures are to be built on top of any portion of the maintenance berm designed or used a seepage control, the berms would be overbuilt and utilities or any other structure or penetrations would be limited to within the overbuilt section. Penetrations trough the berm could become seepage exit points, and this is specified to limit fracture through the main berm.

Where water would be permanently ponded against the riverside slope, these areas would require a 40-foot-wide semi-compacted impervious riverside maintenance berm to limit seepage through the levee. The typical details include a detail of the berm assumed to extend the entire length of any levee section where water is pooled. The berm would have a crown elevation 3 feet above normal pool, a 1V on 40H top slope and a 1V on 3H toe slope. No removal of the riverside blanket near the existing levees is anticipated.

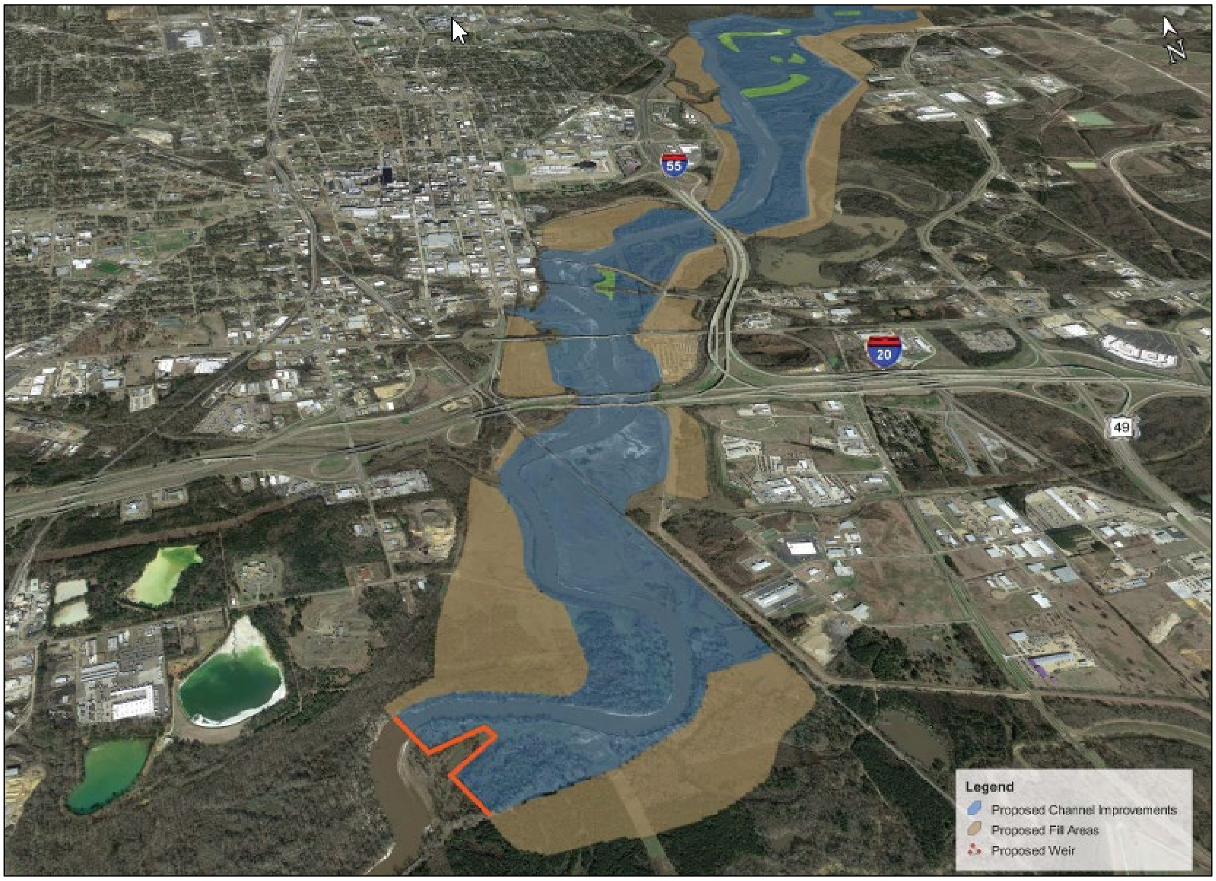

Figure I-5. Plan View of Proposed Channel Improvements Excavated Material Plan, and Weir with Gate

Structure Demolition

The existing weir located at RM 291 near the J. H. Fewell WTP site would be demolished and replaced with a new weir further downstream near RM 284.3 at the south end of the channel improvements area. In the area surrounding the J. H. Fewell WTP, Plan C calls for the demolition of the J.H. Fewell Weir located at RM 291, which is currently set to approximately elevation 250 feet. Dredging would be conducted to elevation 248 feet. It is undetermined if the water intake structures and access way of the J. H. Fewell WTP would need further modification. Demolition may also be required at all or part of the abandoned GM&N/GM&O Railroad Bridge since it was removed in H&H modeling. Figure I-6 shows the excavation extent provided in the black polygon with the WTP, weir and intake structures. The length of area (including the island) directly along the railroad bridge is approximately 3,600 feet.

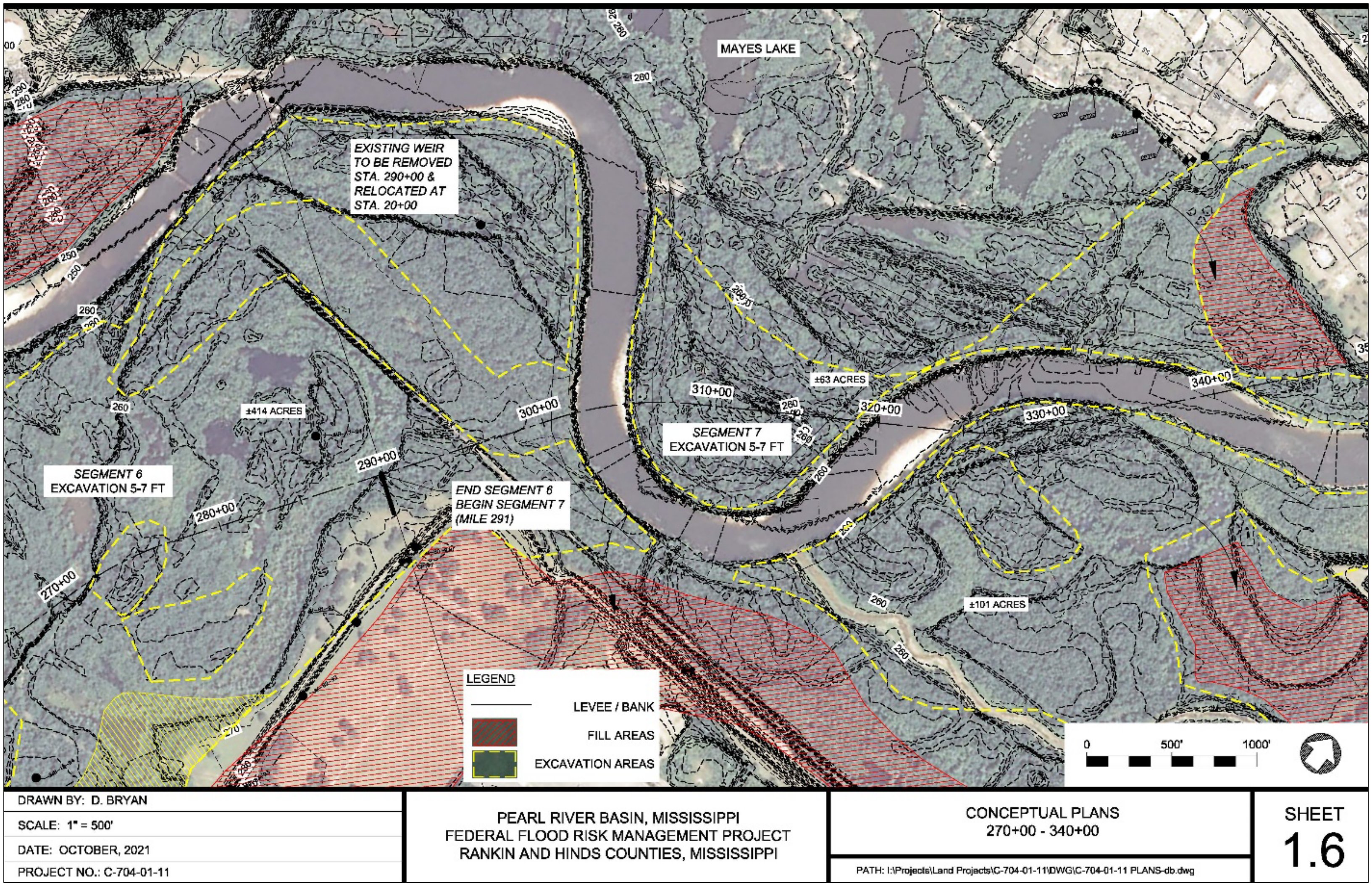

Figure I-6. Proposed Excavation Extent for Demolition of the J.H. Fewell Weir

Construction of New Weir and Gate with Fish Ladder

The demolished weir would be replaced with a new weir constructed downstream near RM 284.3 at the south end of the channel improvements area. The purpose of the new weir would be to maintain the baseline low-water level for water supply at the J. H. Fewell WTP within the channel improvements area. The new weir would provide for a significantly larger body of water within the Pearl River channel to the north of the weir. Downstream low-water hydrologic flows (extreme drought condition minimum flows) within the Pearl River channel would be maintained by means of a 12 x 12-foot low-flow gate. The gate is also required for any future maintenance which requires drawdown of the lake. Portions of weir would be submerged during flood events thereby allowing excess water to pass downstream. Water would pass over the weir with inflow into the lake approximately equaling outflow at any given time (with the exception of the extreme drought, which has a minimum release and outflow could be greater than inflow. However, this is expected to rarely occur, as the Ross Barnett Reservoir also has a minimum release requirement that would pass through the system). As opposed to the existing weir, the replacement weir would be constructed to a higher elevation of 258 NGVD vs. the current of 250 NGVD, and a larger width of 1,500 feet along an approximately 1 mile (1.6 km) stretch on the southern end of the proposed channel improvements area. This weir would impound an area of approximately 2600 acres. Baffle blocks to help prevent floating solids from flowing over the weir are part of the conceptual designs. Further, additional excavation for the fish ladder would occur along the left descending bank of the relocated weir in the project area. The fish ladder has been conceptually designed to be approximately 7,300 feet (2,225.0 m) in length. The fish passage design will be coordinated with The Service and state agencies during the PED phase.

The proposed weir meets USACE and State criteria to be defined as a dam based on the height of the structure and water storage. Additional costs were added to the NFI project cost to account for a redesign and constructing the weir to higher USACE and State criteria for a dam. Rough cost estimates were derived using some unit costs from the NFI. A more refined cost estimate would be done once the dam is redesigned to meet USACE and State criteria.

The proposed weir does not provide any flood control benefits, and construction of the weir necessitates additional pumping needs at existing levees as well as seepage protection in the form of berms and slurry walls on existing levee features upstream of the weir. However, the weir provides a lake surface for future water supply concerns, as well as adding attractive locations for recreation and future economic development. Public recreation facilities within the floodplain (i.e., boat ramps and landings, pedestrian access points, public and RV parks, natural areas, and trails) are not part of Alternative C; however, at a later time, those features may be added by other entities as a result of the weir’s new expanded year-round recreational water body.

Additional Pumping Needs at Existing Levees

The existing levees contain drainage structures that allow water to drain from the interior of the leveed area when the Pearl River is low. When the Pearl River water level is high, the drainage structures are closed, and pump stations are used to pump water out of the leveed area. The original design (original levee construction) of these features called for the drainage structure to handle a 1 percent AEP interior drainage flow and the pumps were originally designed for a smaller event. Later additional pump capacity was added without additional study (see: 2007 Report for details). The proposed new weir would maintain a minimum pool at elevation 258.0 ft. Due to the new pool elevation, the drainage structures would have at least 9 ft of water covering the structures at all times and would no longer be able to operate and prevent the new reservoir from flooding the interior leveed areas. Additional pumping capacity would be installed to mitigate for the loss of capacity of the drainage structures. In addition, some of the proposed fill areas in the NFI plan would fill in part of the sump that is presently used to store water for pumping. The NFI did not perform an interior flooding analysis to determine mitigation features for the loss of the use of the drainage structures. This analysis would need to be completed if Alternative C is selected for construction. Additionally, the Operation and Maintenance (O&M) of the additional pumping would need to be substantially updated from the existing O&M plan for the pumping ability and constant operations prior to construction. Costs for this effort are estimated to range from $100 million to $200 million depending on the size of the pump stations needed. Cost estimates (adjusted for inflation) were based off recent experience with pump cost estimation from studies or actual construction, such as the proposed pump station for the *Raritan Bay and Sandy Hook Bay Hurricane Sandy Limited Reevaluation Report*, dated September 2016, and pump station construction in the Trinity River Corridor were also used to verify cost ranges.

Newly Federalized Levee

An existing non-Federal levee protects the Savanna Street WWTP near RM 282. As part of Alternative C, the levee would undergo maintenance and additional upgrades, so the levee meets the freeboard needed for certification for a 1 percent AEP flood event in advance of the main construction phases (Figure I-7). The levee section proposed for the new Federalized levee around the WWTP consists of a 10-foot crown width with 1V on 3H landside and riverside slopes. If needed, a slurry wall for seepage mitigation would be added. At this location, additional pumps would not be needed to provide protection behind levees since the existing pumps are already in progress of being replaced as part of the Section 219 Environmental Infrastructure Program as discussed in Section 1.5.2 of this report.

Principal features of the work include mobilizing and demobilizing, clearing, and grubbing, removing, and stockpiling any existing crushed stone surface, semi-compacted levee embankment, traverses, adding new crushed stone surfacing, mowing, turfing, erosion control matting, preventing storm water pollution, and providing environmental protection. Additional work could include trenching and the creation and backfill of a concrete slurry wall within the levee footprint.

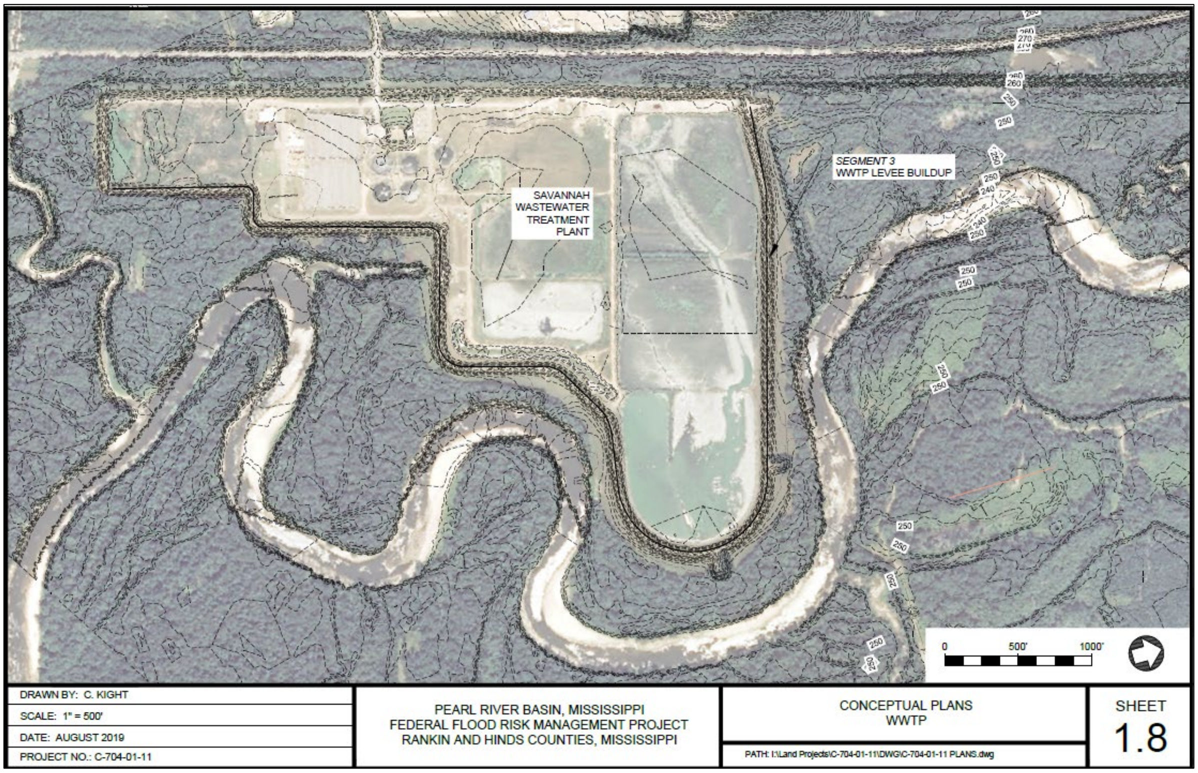

Figure I-7. Proposed Federalized Levee at WWTP

BORROW PLAN

A borrow plan has not been developed at this stage of the analysis. It is conceivable that there is enough borrow material from the material excavated from within the channel but it is unknown at this time if the material is suitable for constructing levees. Should the excavated material within the channel be determined to be unsuitable, borrow material would need to be obtained from another source for construction of any levees. There are potential borrow sources identified within close proximity of the project area (10-mile radius). Reference

Figure 3-8 for a potential source. Borrow opportunities would be further investigated during PED and a supplemental NEPA document would be prepared at that time.

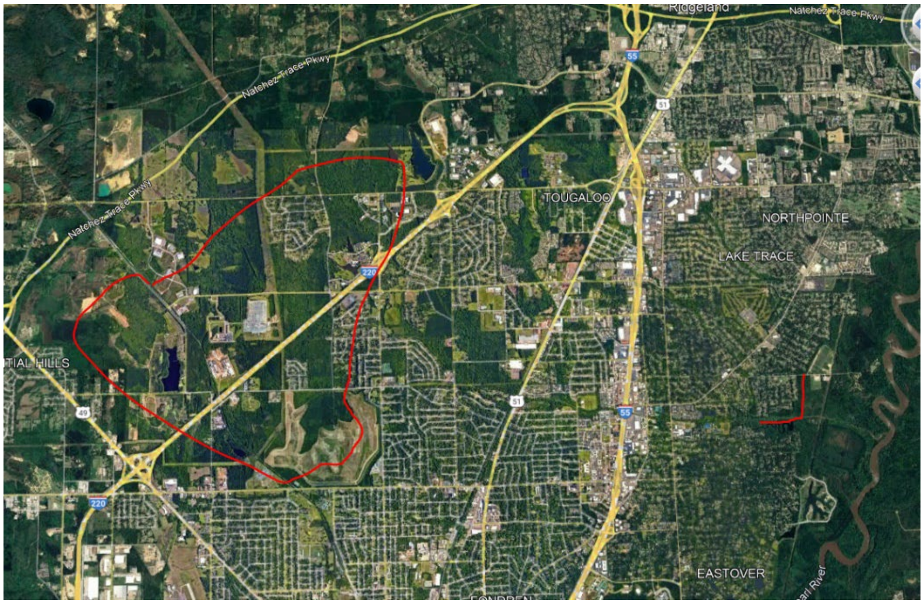

Figure 3-8 Potential Borrow Sources

Property Relocations

Alternate C includes removing the abandoned GM&N/GM&O Railroad Bridge and embankment, relocating or reconstructing property of others, bridge counter measures, utilities and lands or interests purchased for such relocations and conveyed to others. All alterations of railroad bridges would be in accordance with Section 3 of the 1946 Flood Control Act (22 USC 701p). Of the 2,750 acres needed for the implementation of Alternative C, the NFI owns the real estate for approximately 1,120 acres.

Relocations also include the removal of existing historical unpermitted solid waste units in the floodplain, removal and capping of an existing potential HTRW site, and remediating as necessary at full NFI responsibility, including (Figure I-9):

- An existing automotive salvage yard.
- Mitigation features may be required for Gulf States Creosote Company Site.
- Additional capping and bank stabilization features would be required for unpermitted LeFleur’s Landing Site (Jefferson Street Landfill).
- Excavation and removal of approximately half of the closed and sealed Gallatin Street Landfill Site of proposed channel improvements.

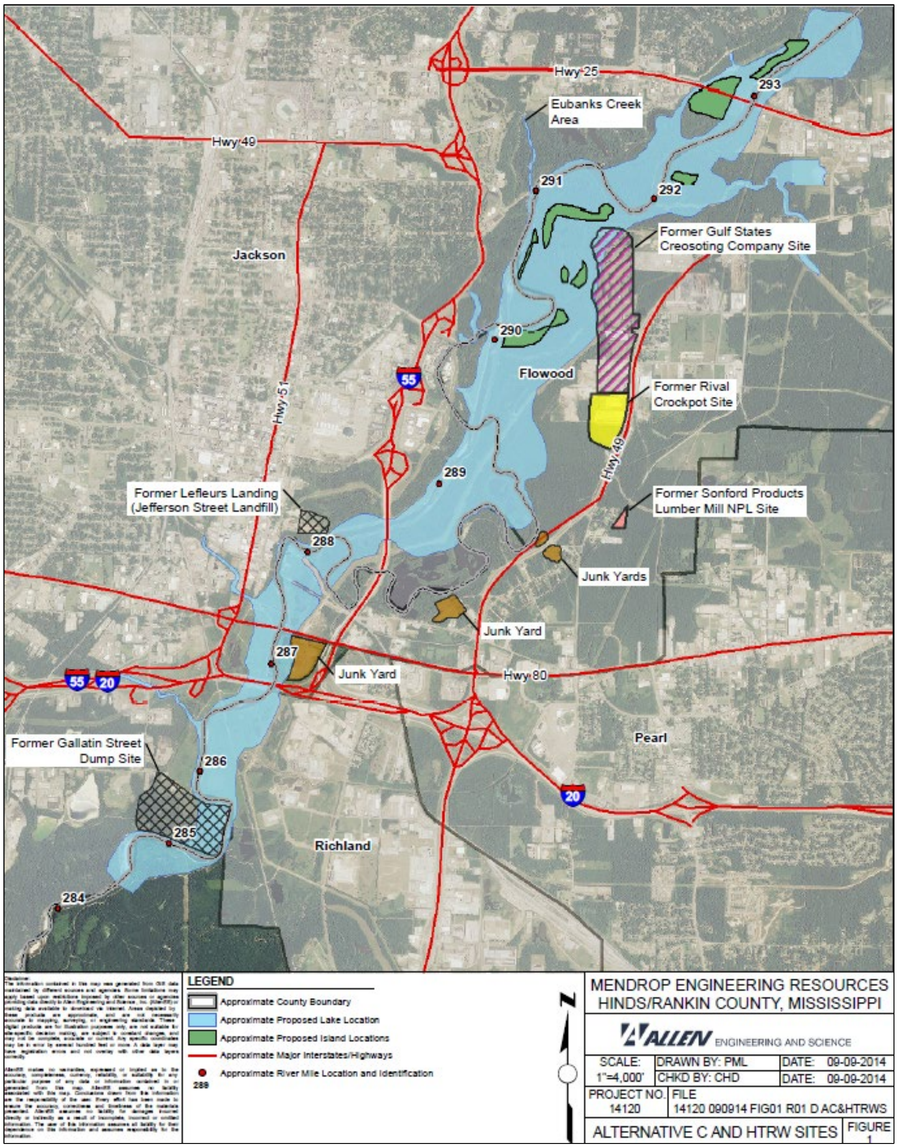

Figure I-9. Known and Potential HTRW Sites within Project Area

The Gulf States Creosote Company Site is located within the project area. The site, or portions thereof, may require avoidance, remediation, or some other mitigating features. The unpermitted LeFleur's Landing Site is also located along the edge of the proposed channel improvement excavation area. It would require additional capping and bank stabilization features due to potential leaching of landfill waste and groundwater movement in the area. Remediation design and coordination with appropriate local, State, and Federal agencies would determine site actions to eliminate potential leaching of landfill waste to the groundwater and movement of groundwater into the proposed channel improvement.

Groundwater controls and a slurry wall may be appropriate remedial actions in this event. The proposed channel improvement excavation area would also bisect the unpermitted Gallatin Street Landfill Site; therefore, excavation and removal of approximately half of the landfill site would be required to construct the proposed channel improvement. This excavated material would then be incorporated into the current remaining landfill area to further elevate the area, cap the area, and provide bank stabilization. Final remedial designs would be coordinated with appropriate Federal and State agencies to determine necessary actions to prevent and/or eliminate potential leaching of landfill waste chemicals to the groundwater and movement of groundwater into the proposed channel improvement area prior to the initiation of excavation activities at this location. Again, groundwater controls and a slurry wall may be appropriate remedial actions.

OPERATIONS AND MAINTENANCE (CHANNEL, WEIR, SEEPAGE BERMS, FISH PASSAGE, LEVEE UPDATES)

Operations and Maintenance is ongoing for existing features within the Rankin-Hinds AOR, additional Operations and Maintenance will be implemented for each constructed feature to USACE Standards. Existing Levee and Pumping Plant manuals will be updated. New features, such as the new weir and lake will require development of new O&M manuals. - The district commander is responsible for developing an OMRR&R manual for each project and separable element constructed under a separate project cooperation agreement (PCA), or functional portion of a project or separable element, reporting the status of the manual through the project management system as required by ER 5-7-1(FR). Normally, the Engineering Division will be assigned the overall responsibility for preparing a draft OMRR&R manual with appropriate inputs from other disciplines and, in consultation with the project sponsor, furnishing the draft manual to the project manager for coordination with the project sponsor, and preparing the final OMRR&R manual for approval. For a functional portion, the OMRR&R manual is an interim manual pending completion of the entire project or separable element. The major subordinate commander is responsible for review and approval of the manual. The project sponsor, normally through a permanent committee consisting of our headed by an official usually called the "superintendent" is responsible for carrying out the provisions of the OMRR&R manual. The OMRR&R manual will include coverage of all OMRR&R subjects required by the PCA and existing regulations, in detail sufficient to ensure proper OMRR&R accomplishment by the project sponsor. Project sponsors, subject to review and approval of the district commander, may prepare supplements to the manual.

Section 3

ALTERNATIVE COMBINATION THEREOF PLAN

The USACE evaluated various combinations of the project features to determine a combination that would maximize the flood risk reduction benefits while reducing adverse impacts and costs. Based on H&H modeling and agency coordination, the CTO Alternatives could be comprised of the following features with or without a weir (Alternative D and Alternative E):

- Alternative A1 Non-Structural Plan
- Excavation of Main Channel
- Federal levee improvements
- New weir construction including a fish ladder.
- Non-Federal levee improvements (Savannah Street WWTP)
- Levees
- Bridge modifications
- Mitigation features

CTO FEATURE SUMMARY

The Alternative CTO would provide similar flood risk reduction as the NFI Alternative C with a smaller footprint. Table I-6 Provides a listing of the project features of the CTO alternative with and without a weir. Based on H&H modeling, the weir would be located in a different location from the weir identified in Alternative C. Figure I-10 shows the location of the proposed weir.

***Table I-6**. CTO Alternative Project Features and Quantities*

Feature	ALT CTO W/WEIR (Alt D)	ALT CTO WO/WEIR (Alt E)	Units
	Quantity	Quantity	
NON-STRUCTURAL			
Non-structural plan	60 43 residential 17 nonresidential	60 43 residential 17 nonresidential	structures
STRUCTURAL			
Lake Surface Water Area	1706	0	acres
Clearing and Grubbing *	1,501	1,501	acres
Channel Improvements Excavation *	1016 (11.3-14.1)	1016 (11.3-14.1)	acres (mcy)

Fill Area *	485 (14.7-18.4)	585 (14.7-18.4)	acres (mcy)
Stabilization or armoring for bridge abutments *	7	7	bridges
Hard Point in tributary channels to prevent incision/sediment into newly constructed lake *	750	750	Feet (crossing river)
Newly Federalized Levee (inc. slurry wall*	1.7	1.7	miles
Slurry Wall Savanna Street WWTP*	1.7	1.7	miles
New Slurry wall for seepage of existing features**	Up to 1.3	0	miles
Weir and new gate **	1	0	each
Pumps to address interior drainage Impacts **	1	0	each
Fish Passage **	5,000-6,000	0	feet
Canton Club Levee***	1.4	1.4	miles
OPERATIONS AND MAINTENANCE			
Weir	1	0	each
Fish Passage	?	0	each
Terrestrial Habitat Mitigation	11	11	events
Riverine Habitat Mitigation	?	0	events
Lake	1	0	each
Pump Station	1	0	each
Levees	2	2	each
MITIGATION			
Sandbars (material from excavation)	31	0	acres
Reforest top bank of fish passage	?	0	acres
Riverbank preservation	10	10	miles
Removal of obsolete aquatic barriers	1	0	structure
Connect occupied and suitable unoccupied riverine habitat	?	0	acres
Open historically lost riverine habitat	?	0	acres
Terrestrial Habitat Mitigation	10,762	10,762	acres

* Components of Alt C Excavation

** Components of Alt C Weir

***Feature from Alternative B

Nonstructural Component

The nonstructural analysis was conducted based on a residential and non-residential structure inventory developed by USACE in 2023 using the National Structural Inventory database of structures, version 2.0. An assessment of structures located in the 10 percent, 4 percent, 2 percent, and 1 percent AEP floodplains in the Post Project Construction was performed (reference Appendix N for more details). The NS features Elevation and floodproofing of structures were used to determine the effectiveness of a nonstructural alternative. For the analysis, residential structures would be elevated to the 1 percent AEP BFE based on year 2082 hydrology up to 13 feet above the ground and nonresidential structures to be floodproofed up to 3 feet above the ground. Participation in the nonstructural plan would on a voluntary basis by the property owner.

As a result of feedback from the public meetings held in May and June 2023, the option to include property acquisition (buyout) on a voluntary basis is included in the nonstructural implementation plan (Appendix N). Full details regarding the Non-structural Implementation Plan are included in Appendix N.

NFI Channel Improvement/Weir/Levee Plan Components

The Alternative CTO provides similar flood risk reduction at the NFI Alternative C with a smaller footprint. Alternative CTO consists of the construction of channel improvements, a new weir with a low-flow gate structure downstream for future potential water supply while simultaneously creating a lake area for recreational opportunities (Figure I-10). Federal levee improvements (excavated material plan) and raising an existing non-Federal ring levee (the Savannah Street WWTP Levee).

Modifications include constructing a weir upstream of the location identified for Alternative C, reducing excavation limits which reduces fill areas and thus reducing environmental impacts throughout the project footprint. The new weir would have a lower elevation than proposed for alternative C as well as a reduction in the overbank excavation limits. These changes could reduce environmental impacts especially to HTRW sites within the project footprint.

The Alternative CTO seeks to realize flood risk management through a reduced scope of measures that provide similar levels of flood risk reduction as Alternative C. Flood risk management is realized through lowering of the channel overbanks within the project footprint, thereby improving conveyance of water through the project area and lowering the water surface elevation of the river in some places within the project area over 4 feet (1.2 m). Water surface elevation reductions due to this excavation would provide reduction of flood elevations not only within the reach of excavation, but additional elevation reductions upstream for over 8 miles upstream of the excavation limits.

Construction of the project would require relocations and/or improvements to various public and private utilities and infrastructure, mitigating potential HTRW and other hazardous waste sites within the floodplain, avoidance and minimization features required under the Endangered Species Act, and the creation of new habitat mitigation areas to offset losses within the project's construction footprint areas.

There are a total of 9 transmission lines within the project area. All efforts would be made to avoid, monitor, maintain required clearance, and protect these structures. If avoidance is not possible, then utility relocation or raising of lines/protection of structures would be necessary. It is estimated that 4 to 5 of these lines will require additional utility relocation costs. Coordination with the operating entity to determine specific requirements of each transmission line will be conducted during PED.

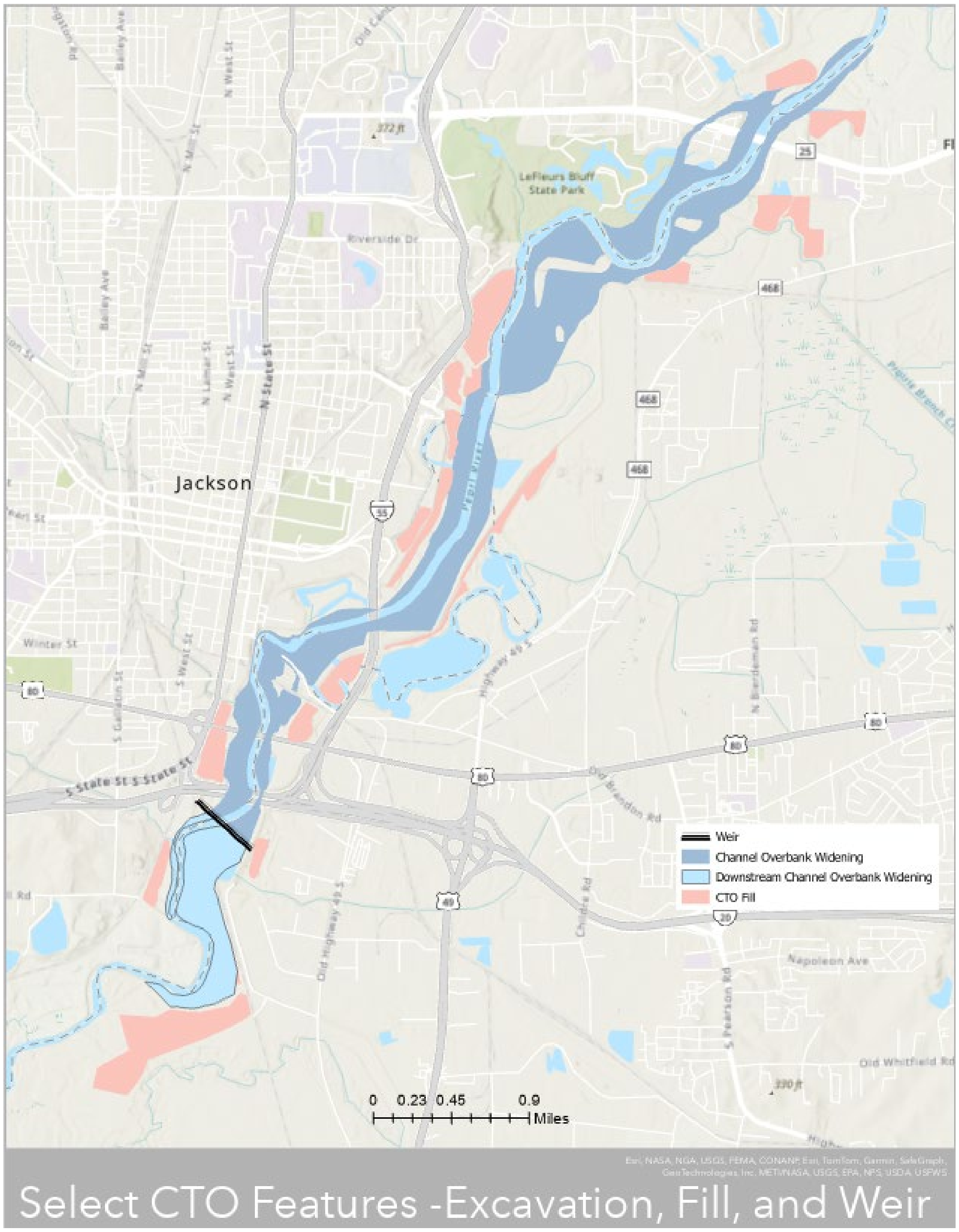

***Figure** I-10. Select CTO Features – Excavation, Fill, and Weir*

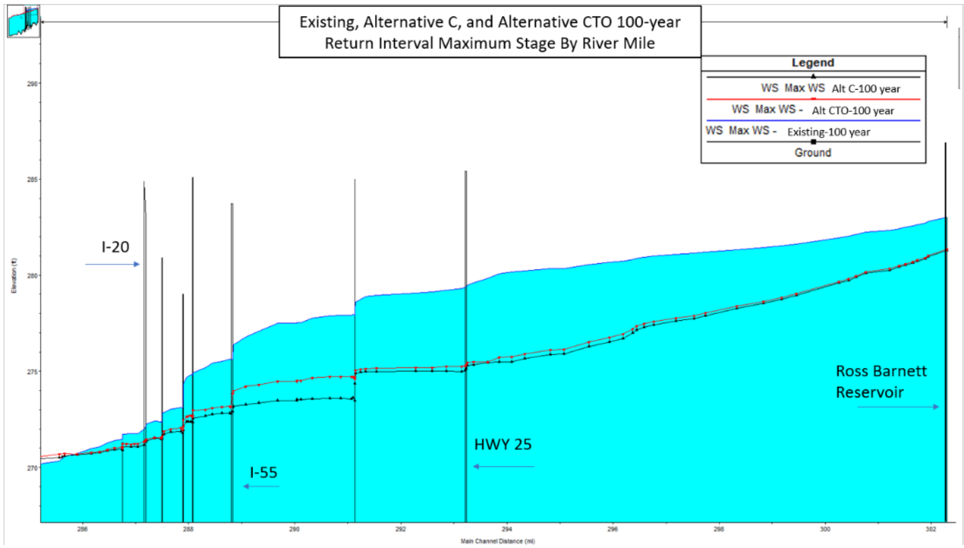

Figure I-11. *USACE modeling Results for the 1% AEP (100-year) With and Without Project Routing Scenario*

Channel Improvements

Channel improvements (Figure I-12) consist of excavating areas along the Pearl River to improve conveyance from RM 285 to 294., which included river reaches previously channelized during the existing levee construction. The channel improvement footprint includes excavation of up to 1,016 acres. Of the total 1,016 acres, approximately 853 acres are located above the proposed weir, and approximately 163 acres are located below the proposed weir. The width of excavation would vary ranging from 500 to 2,600 feet (152-793 m) including the river width. The actual widths would be determined during the PED phase. The depth of excavation would vary between 0 -15 feet to meet the proposed bottom elevation of 250.0 feet NGVD. The quantity of material excavated from the floodplain and channel overbanks would range from 11.3 to 14.1 million cubic yards (8.6-10.7 million m³) of material. The existing river channel will not be widened, instead excavation of the overbank areas will occur.

The preliminary project layout includes islands within the channel improvement excavation area that would be maintained and/or expanded upon from RM 288.0 to RM 292.0. Further, sand bars could be constructed inside the floodplain and along the existing islands to compensate for the loss of sand bar habitat.

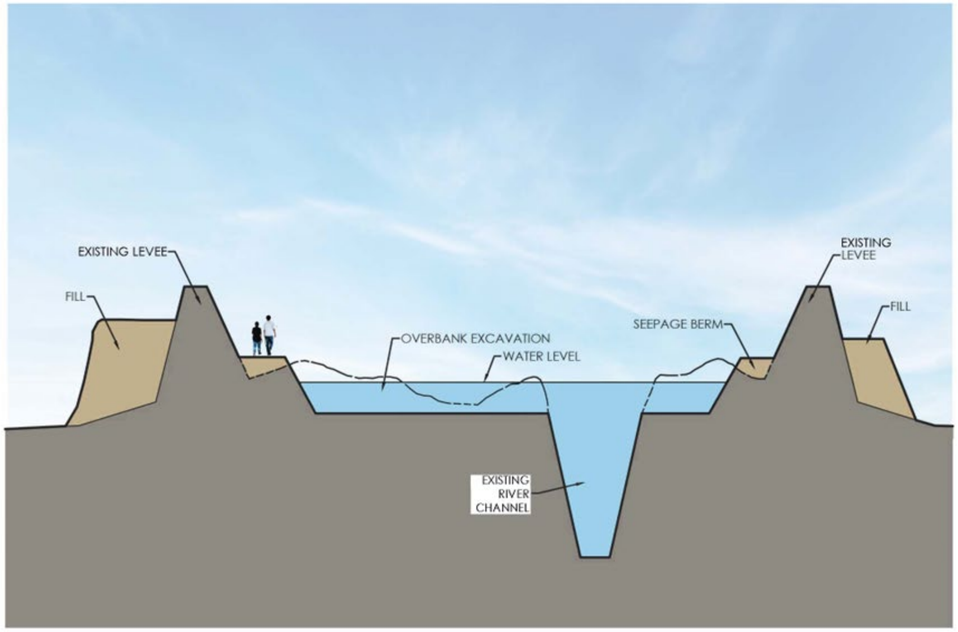

Figure I-12. *Channel Improvements with a Relocated Weir*

Overbank Modifications

The existing overbank areas of the Pearl River channel would be lowered to increase conveyance of flood flows. Existing levees would remain in place and would be maintained to increase this control and to aid in haul access. Excavation limits near the existing levees would be determined during final design.

<u>Station 10+00 through 140+00</u>. Specific items included in this reach are the I-20 Interstate bridges (Sta. 95+00±) as well as the U.S. Highway 80 (Sta. 110+00), Old Brandon Road (Sta. 135+00±), and railroad bridges (Sta. 70+00±, Sta. 130+00±). Two high-pressure gas lines run through this reach and will would have to be carefully monitored as excavation and grading activities progress. Multiple access points on both sides of the river would have to be maintained and monitored from a perspective of public safety and construction use.

<u>Station 140+00 through 290+00.</u> This reach contains excavating the overbank areas around high points such that high points would appear as islands. As with the previous reach segment, numerous access points would require management and maintenance for use and safety. A creosote slough area (Sta. 240+00±) will be avoided during construction, to not

disturb or cause any objectionable material to be exposed or mixed with other excavated material.

Station 290+00 through 400+00. As with the previous downstream reaches, there are bridges to work around (Highway 25 near Sta. 360+00), and gas lines and transmission lines that must be monitored during earthmoving operations. Depending on the final design, Mayes Lake (Sta. 310+00±) may need tie-in work to maintain its current level. A determination about the tie-in work would be made during the PED phase. An existing abandoned railroad embankment of the Gulf, Mobile & Northern/Gulf Mobile and Ohio (GM&N/GM&O) Railroad Bridge could also be affected and was removed in H&H modeling. Some excavation would be required in this reach such that high points would appear as islands. The existing weir at the water works bend near Station 290+00 would remain undisturbed.

Excavated Material Plan (Fill material)

Alternative CTO would upgrade the existing federal levees by placing excavated material on the protected side of the levees. Excavated fill material would also be placed in designated disposal areas in other locations within the flood plain. The disposal fill areas would impact approximately 485 acres (151 ha) (Figure I-10).

Clearing and grubbing of approximately 1501 acres would occur prior to placement of the excavated fill material from the channel lowering. The excavated fill material would be used to create land areas ranging from 6.5 to 88 acres (2.6 – 21 hectares) within the Jackson MSA. The newly created areas could allow for expanded riverfront access, natural areas, and commercial development, along with recreational opportunities. The Jackson MSA has significant historical and cultural site presence, final site locations would be adjusted during PED following completion of cultural resource surveys.

Fill material placed behind levees would be graded to the same elevation or lower than existing levees, compacted for suitably for future land development. However, if any structures are built on top of any portion of the maintenance berm designed or used as a seepage control, the berms would need to be overbuilt and utilities or any other structure or penetrations would be limited to within the overbuilt section.

Where water would be permanently ponded against the riverside slope, these areas will require a 40-foot-wide semi-compacted impervious riverside maintenance berm to limit seepage through the levee. The berm assumed to extend the entire length of any levee section where water is pooled. No removal of the riverside blanket near the existing levees is anticipated. A riverside blanket refers to a top layer of clay and/or silt soil with low permeability constructed on the riverside of a levee to reduce the movement of water underneath the levee.

If any structures are to be built on top of any portion of the maintenance berm designed or used a seepage control, the berms would be overbuilt and utilities or any other structure or penetrations would be limited to within the overbuilt section. Penetrations trough the berm could become seepage exit points, and this is specified to limit fracture through the main berm.

Material Provided to NFI

Up to 1,660,000 cy (1,269,000 3) of fill material (estimated as 100 acres (40.5 hectares) of fill 10 feet high) would be provided to the NFI for additional usage within the project footprint. This material would either hauled directly from the excavation site or moved to a staging area for removal by the NFI. Existing fill areas would be used as staging areas after clearing and grubbing but prior to fill activities.

Hardpoints at Base of Tributaries

Multiple tributary inflow points exist within this reach and Alternative CTO will add a hardpoint, via a rock chute to prevent backward erosion at each tributary inflow where the excavation of overbanks decreased the tributary channel bottom elevation at or near the confluence of those tributaries with the Pearl River.

Reinforcement of Bridge Abutments or Replacement of Bridges (if required)

If any stabilization or armoring, such as riprap, slope paving, slide repairs, etc., is required, it will be carried out prior to clearing and any major channel work. Following its own analysis, the Mississippi Department of Transportation (MDOT) has informed the Rankin-Hinds Flood Control District (the Flood Control District), that MDOT agrees to collaborate with the Flood Control District in “the advancement of this project and to ensure countermeasures are included, if determined necessary during the future design process.” (Letter to G. Rhoads, dated February 26, 2024) To this end, the Flood Control District developed a range of cost estimates for potential structural and hydraulic countermeasures that could be recommended if countermeasures are determined necessary. The array of countermeasure features analyzed will mitigate potential impacts to MDOT bridges that will be identified during the PED phase. The estimated cost for these features is based upon known costs for the construction of hydraulic and structural countermeasures on another MDOT project at downstream hydraulic crossings of the Pearl River. When additional information becomes available during PED, adjustments to the design can and will be made to reduce potential impacts. Any proposed countermeasure design and implementation will be conducted with MDOT’s concurrence, review, and approval.

Rough estimations of the level of effort required to mitigate for bridge impacts include improvements for approximately 36 bents, 12 piers, abutment scour, as well as funding to conduct monitoring surveys. A pile is a concrete post that is driven into the ground to act as a leg or support for a bridge. A bent is a combination of the cap and the pile. Together, with other bents, act as supports for the entire bridge.

There are a total of 2 active railroad bridges within the project area. All efforts would be made to avoid, monitor, and protect these structures. Additional modeling is required to validate these assumptions during PED. If avoidance is not possible, then coordination with the operating entity to determine specific requirements of each railway bridge will be conducted during PED. All alterations of railroad bridges would be in accordance with Section 3 of the 1946 Flood Control Act (22 USC 701p).

Construction of New Weir and Gate with Fish Ladder

Alternative CTO may include a new weir to be constructed near RM 286.5 at the southern end of the channel improvements area. It should be noted that the CTO alternative does not include any modifications to the existing J. H. Fewell weir. This new weir would provide for a larger body of water within the Pearl River channel to the north of the weir and fish ladder. Downstream low-water hydrologic flows (extreme drought condition minimum flows) within the Pearl River channel would be maintained by means of a 12 x 12-foot low-flow gate. Also note that the gate is required for any future maintenance which requires drawdown of the lake. Portions of the weir would be submerged during normal flow allowing excess water to pass downstream. Water would pass over the weir with inflow into the lake approximately equaling outflow at any given time (with the exception of the extreme drought, which has a minimum release and outflow could be greater than inflow. However, this is expected to occur very rarely, as the Ross Barnett Reservoir also has a minimum release requirement that would pass through the system). As opposed to the existing weir, the new weir would be constructed to a higher elevation of approximately 256 feet. NAVD 88 with a length of up to 1,700 feet with a fish ladder located on the southern end of the proposed channel improvements area. The weir would impound approximately 6 feet of water along the excavated overbanks (about 1350 ft) and up to 22 feet in the approximately 350 feet across the main channel. This would impound an area of approximately 1706 acres, of this area approximately 637 acres are upstream of the Fewell Water Treatment Plant Weir. Downstream erosion protection from flow over the weir are part of the conceptual designs.

A fish ladder (Figure I-13) would be excavated around the relocated weir within the project area. The fish ladder is conceptually designed to be approximately between 5,000 - 6,000 feet (1524-1829 m) in length. The fish ladder would be constructed at an approximate 0.004 ft/ft slope and tie into the Conway Slough which connects to the Pearl River 0.8 miles downstream of the CN Railroad Bridge. The fish ladder design would be coordinated with US Fish and Wildlife, state agencies and Tribes during the PED phase.

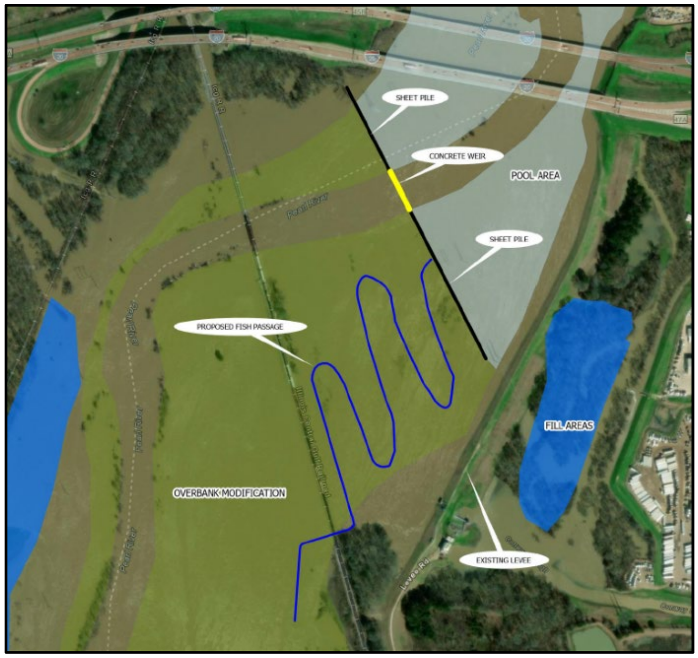

Figure I-13. Proposed Weir (Black) and Fish Ladder (Blue) Exact Dam Design to be determined in PED.

The proposed weir meets USACE and State criteria to be defined as a dam based on the height of the structure and water storage. As a result, the dam would be designed and constructed to meet USACE and State criteria for a dam.

The construction of a weir without excavation of the overbanks has not been sufficiently investigated to ensure that inducements do not occur. Construction of the weir without channel conveyance improvement was not analyzed and would require additional study if selected.

The proposed weir does not provide any flood control benefits, and construction of the weir necessitates additional pumping needs at existing levees as well as seepage protection in the form of berms and slurry walls on existing levee features upstream of the weir. However, the weir provides a lake surface for future water supply concerns, as well as adding attractive locations for recreation and future economic development. The proposed weir would result in an expanded, year-round recreational water body capable of supporting recreational facilities. Potential recreation sites would be limited to areas disturbed by construction and design of these facilities would be coordinated during PED (Figure I-14). The potential recreational opportunities could include boat ramps, camping areas, fishing piers, trails, or wildlife viewing areas.

Implementation of this alternative would be subject to the non-Federal sponsor agreeing to comply with the applicable federal laws and policies prescribed in the model Partnership Agreement for Authorized Structural Flood Risk Management Projects. The Flood Control District, the non-Federal sponsor, anticipates recreation operations will be solely its responsibility. As such, recreation design and construction would be cost shared.

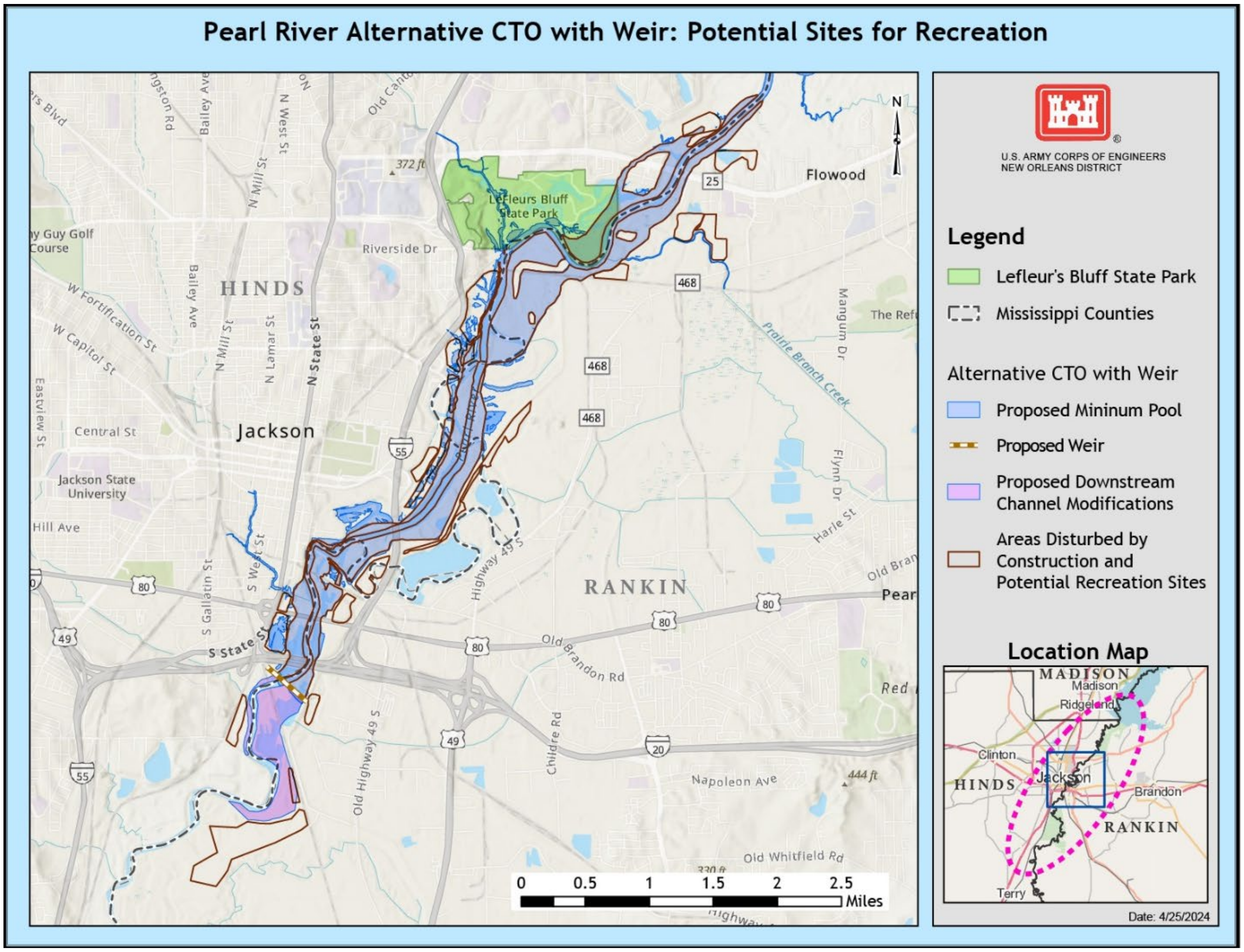

Figure I-14. Potential Sites for Recreational Features

Pumping Needs at Existing Levees

The existing levees contain drainage structures that allow water to drain from the interior of the leveed area when the Pearl River is low. When the Pearl River water level is high, the drainage structures are closed, and pump stations are used to pump water out of the leveed area. The original design of these features called for the drainage structure to handle a 1 percent AEP interior drainage flow and the pumps were originally designed for a smaller event.

Alternative CTO calls for the construction of a new weir with a minimum pool at elevation 256.0 ft. As a result, the drainage for the Jackson Fairgrounds Levee would always impound at least multiple feet of water on the structure and would no longer be able to operate via gravity flow in order to prevent the new lake from flooding the interior leveed areas.

The proposed new weir was placed upstream of the East Jackson Levee drainage structure, so the pool should not impact the operation of the drainage structure. Additional pumping capacity would be needed to mitigate for the loss of capacity of the gravity flow drainage at the Jackson Fairgrounds Levee. Additionally, the Operation and Maintenance of the additional pumps would need to be substantially updated from the existing O&M plan for the pumping capacity and constant operations.

Savannah Street WWTP Levee

This is an existing non-Federal levee that provides flood risk reduction to the Savanna Street WWTP near RM 282 (Jackson-East Jackson Flood Control Project NLDID: 14050000124). The levee would undergo maintenance and additional upgrades to meet the freeboard necessary to meet a 1 percent AEP flood event in advance of the main construction phases (Figure I-15). The new Federalized levee around the WWTP consists of a 10-foot crown width with 1V on 3H landside and riverside slopes. If needed, a slurry wall for seepage mitigation would be added. Additional pumps would not be needed since the existing pumps are being replaced as part of the Section 219 Environmental Infrastructure Program discussed in Section 1.5.2 of this report.

Principal features of the work include mobilizing and demobilizing, clearing and grubbing, removing and stockpiling any existing crushed stone surface, semi compacted levee embankment, traverses, adding new crushed stone surfacing, mowing, turfing, erosion control matting, preventing storm water pollution, and providing environmental protection. Additional work could include trenching and the creation and backfill of a concrete slurry wall within the levee footprint.

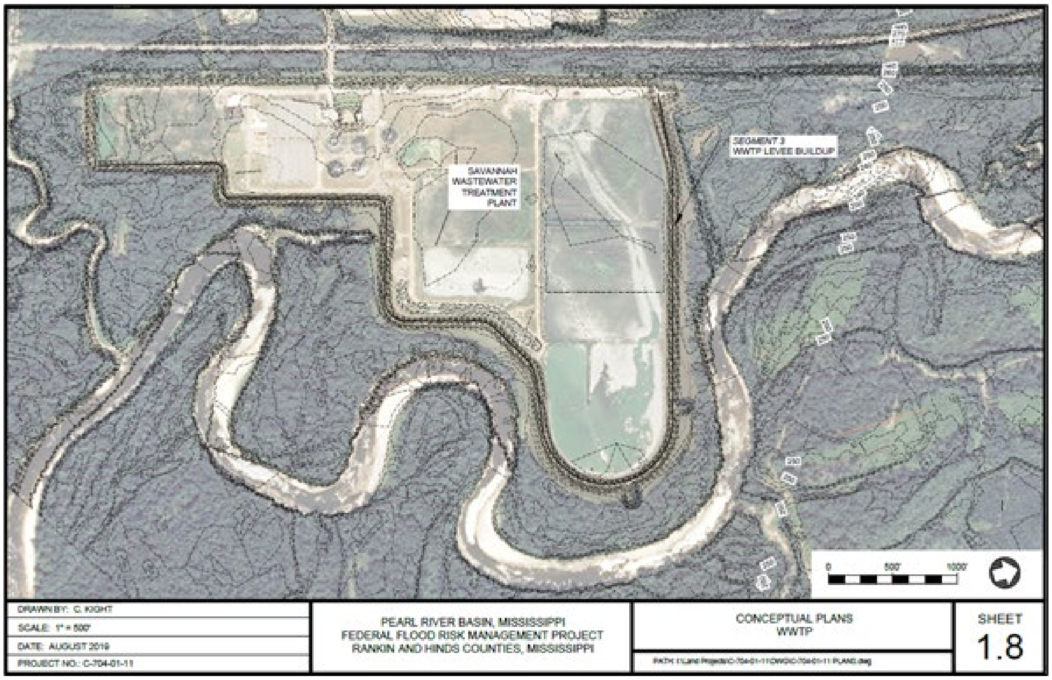

Figure I-15. Proposed Federalized Levee at Savannah WWTP

Operations and Maintenance (Channel, Weir, Seepage Berms, Fish Passage, Levee updates)

Operations and Maintenance is ongoing for existing features within the Rankin-Hinds AOR, additional Operations and Maintenance will be implemented for each constructed feature to USACE Standards. Existing Levee and Pumping Plant manuals will be updated. New features, such as the Canton Club Levee and the new weir and lake will require development of new O&M manuals. The district commander is responsible for developing an OMRR&R manual for each project and separable element constructed under a separate project cooperation agreement (PCA), or functional portion of a project or separable element, reporting the status of the manual through the project management system as required by ER 5-7-1(FR). Normally, the Engineering Division will be assigned the overall responsibility for preparing a draft OMRR&R manual with appropriate inputs from other disciplines and, in consultation with the project sponsor, furnishing the draft manual to the project manager for coordination with the project sponsor, and preparing the final OMRR&R manual for approval. For a functional portion, the OMRR&R manual is an interim manual pending completion of the entire project or separable element. The major subordinate commander is responsible for review and approval of the manual. The project sponsor, normally through a permanent committee consisting of our headed by an official usually called the "superintendent" is responsible for carrying out the provisions of the OMRR&R manual. The OMRR&R manual will include coverage of all OMRR&R subjects required by the PCA and existing regulations, in detail sufficient to ensure proper OMRR&R accomplishment by the project sponsor. Project sponsors, subject to review and approval of the district commander, may prepare supplements to the manual.

Levees Plan

Canton Club Levee

A levee segment of approximately 1.5 miles is proposed on the west bank of the Pearl River in northeast Jackson (Figure 3-16). This levee would provide additional flood risk reduction for approximately 100 acres of high density developed neighborhoods. This area is bounded on the north by the North Canton Club Circle and Beechcrest Drive on the South. It is estimated this would reduce flood risk for over 250 homes.

Figure 3-16. Proposed Canton Club Levee (orange line)

Principal features of the work include mobilizing and demobilizing equipment, clearing and grubbing, removing and stockpiling any existing crushed stone surface, semi compacted levee embankment, traverses, adding new crushed stone surfacing, mowing, turfing, erosion control matting, preventing storm water pollution, and providing environmental protection.

If additional borrow is necessary, the borrow areas would be acquired by the NFI and furnished by the Government to the contractor (government furnished borrow). Some small areas could be more appropriate for the construction of a short floodwall, typically an I or T wall, could be more appropriate for some small areas due to space constraints, though further analysis would be required. Constructing a less designed berm could be more appropriate where smaller loadings would occur.

Construction of the project will require relocations and/or improvements to various public and private utilities and infrastructure, avoidance and minimization features required under the ESA, and the creation of new habitat mitigation areas to offset losses within the project's construction footprint areas.

Borrow Plan

A borrow plan has not been developed at this stage of the analysis. It is conceivable that there is enough borrow material from the material excavated but it is unknown at this time if the material is suitable for constructing levees. Should the excavated material be determined to be unsuitable, borrow material would need to be identified for construction of any levees. There are potential borrow sources within close proximity of the project area (10-mile radius). Reference Figure 3-8 for potential source. Borrow opportunities would be further investigated during PED and a supplemental NEPA document would be prepared at that time.

Operations and Maintenance (Canton Club Levee)

Operations and Maintenance will be implemented for each constructed feature to USACE Standards. The district commander is responsible for developing an OMRR&R manual for each project and separable element constructed under a separate project cooperation agreement (PCA), or functional portion of a project or separable element, reporting the status of the manual through the project management system as required by ER 5-7-1(FR). Normally, the Engineering Division will be assigned the overall responsibility for preparing a draft OMRR&R manual with appropriate inputs from other disciplines and, in consultation with the project sponsor, furnishing the draft manual to the project manager for coordination with the project sponsor, and preparing the final OMRR&R manual for approval. For a functional portion, the OMRR&R manual is an interim manual pending completion of the entire project or separable element. The major subordinate commander is responsible for review and approval of the manual. The project sponsor, normally through a permanent committee consisting of our headed by an official usually called the "superintendent" is responsible for carrying out the provisions of the OMRR&R manual. The OMRR&R manual will include coverage of all OMRR&R subjects required by the PCA and existing regulations, in detail sufficient to ensure proper OMRR&R accomplishment by the project sponsor. Project sponsors, subject to review and approval of the district commander, may prepare supplements to the manual.

Mitigation Component

Habitat Mitigation would be achieved by implementing Corps constructed mitigation projects and/or purchasing of mitigation bank credits. Further planning and analysis would be completed during PED to determine which strategies, stand alone or combined, would fully compensate for habitat impacts.

Mitigation features may be required for Gulf States Creosote Company Site. The Creosote Slough is located within the project area. The site, or portions thereof, may require avoidance, remediation, or some other mitigating features. Groundwater controls and a slurry wall may be appropriate remedial actions in this event. Final remedial designs would be coordinated with appropriate Federal and State agencies to determine necessary actions to prevent and/or eliminate potential leaching of chemicals to the groundwater and movement of groundwater into the proposed channel improvement area prior to the initiation of excavation activities at this location.

Coordination with appropriate local, State, and Federal agencies would determine site actions to eliminate potential leaching of landfill waste to the groundwater and movement of groundwater into the proposed channel improvement.