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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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).

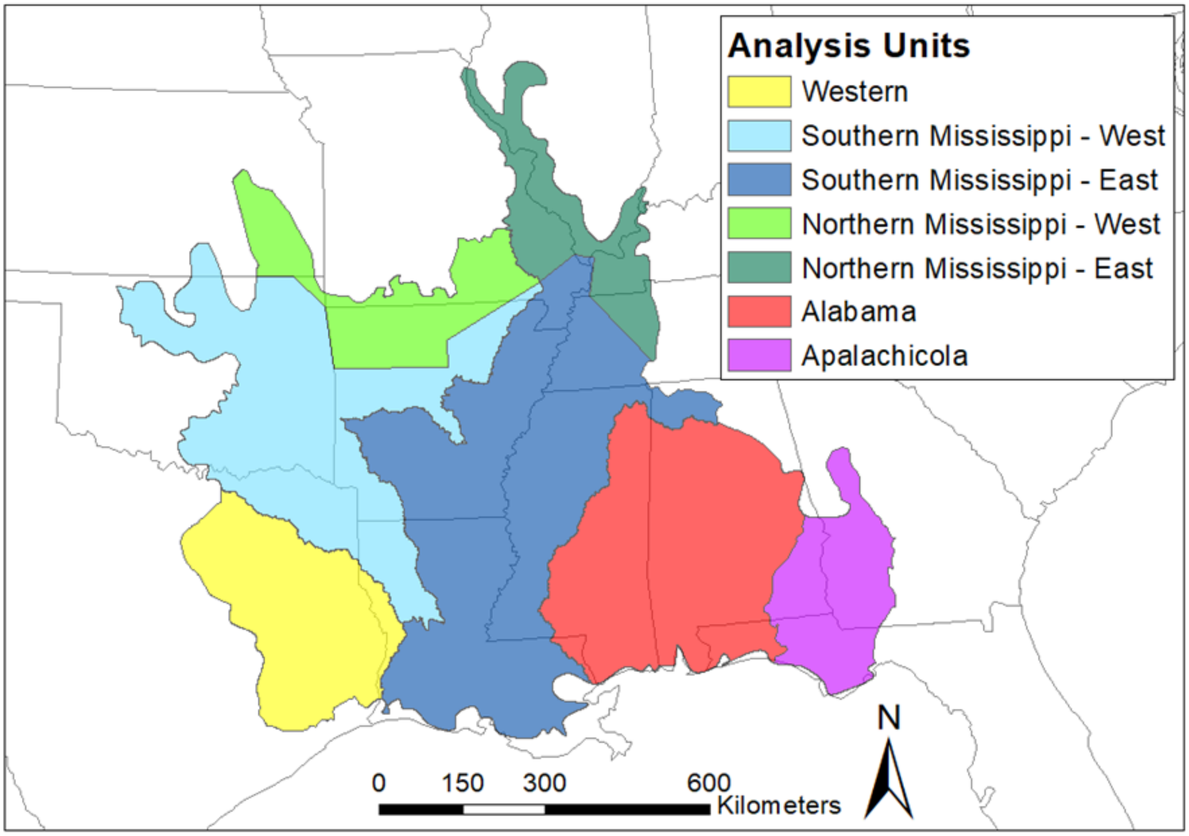

Figure ES1. Alligator snapping turtle analysis units. The two Southern Mississippi units (blues) make up one representative unit and the two Northern Mississippi units (greens) make up one representative unit; the remaining analysis units each make up a single representative unit.

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.

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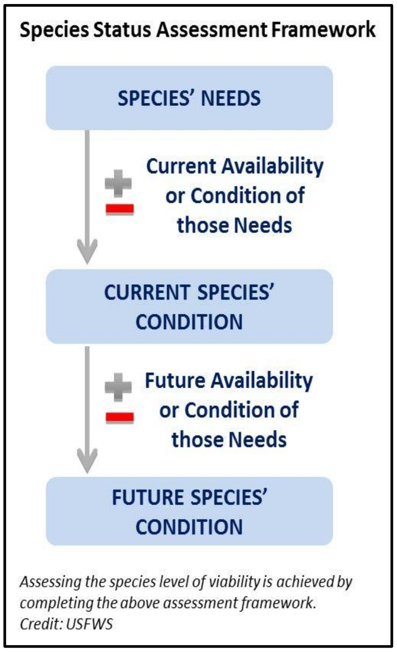
CHAPTER 1 – INTRODUCTION AND ANALYTICAL FRAMEWORK

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).



Assessing the species level of viability is achieved by completing the above assessment framework. Credit: USFWS

Figure 1. Species Status Assessment Framework

- **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.

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).

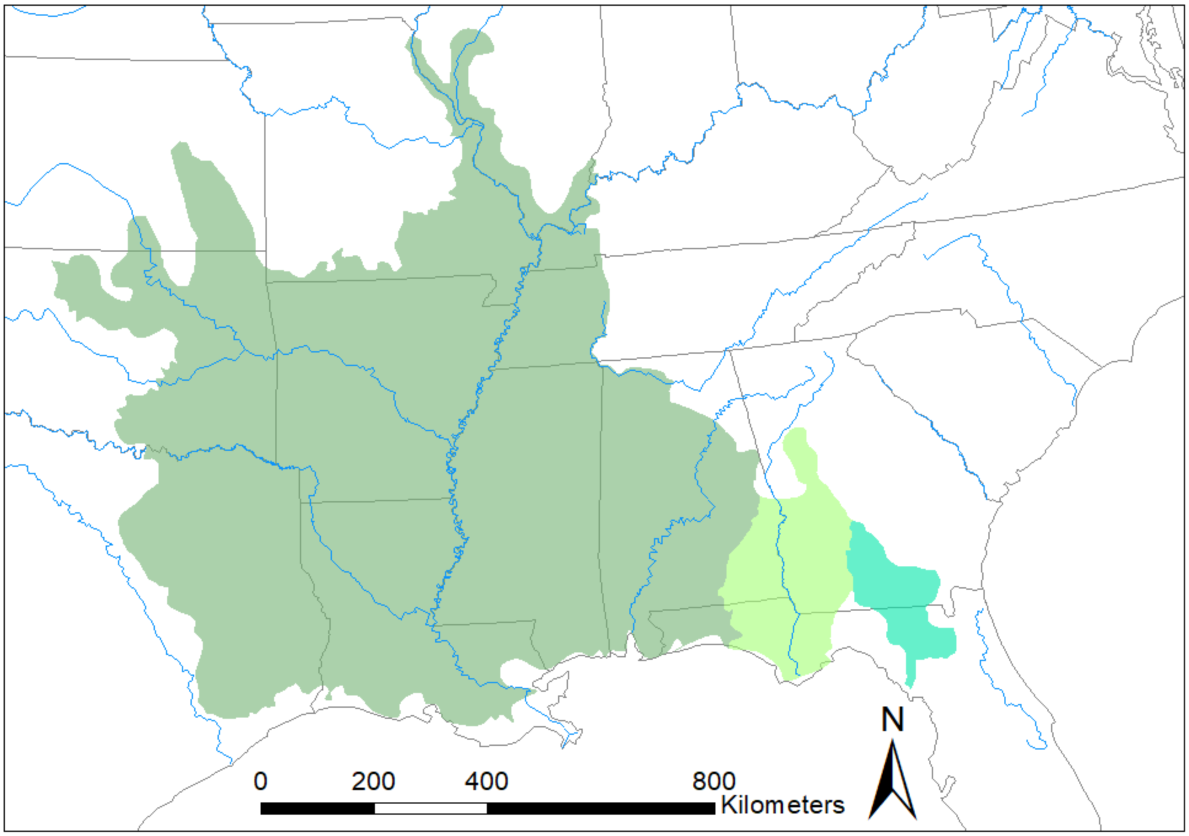


Figure 3. Range of the alligator snapping turtle. Different shades represent three main genetic lineages.

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).

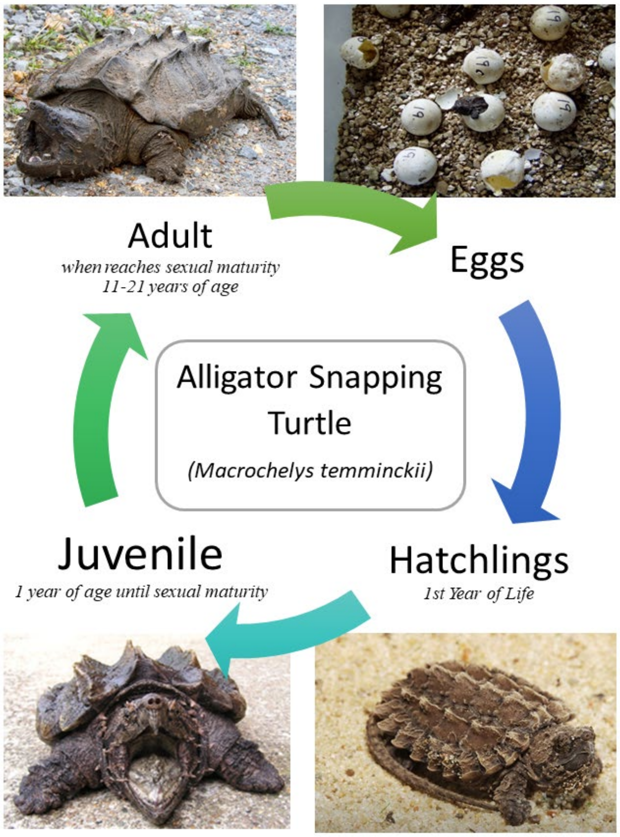


Figure 5. Alligator snapping turtle life cycle. Photo credits: Eva Kwiatek (top left), U.S. Fish and Wildlife Service (top right), Indiana DNR (bottom left), Kory Roberts (bottom right).

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).

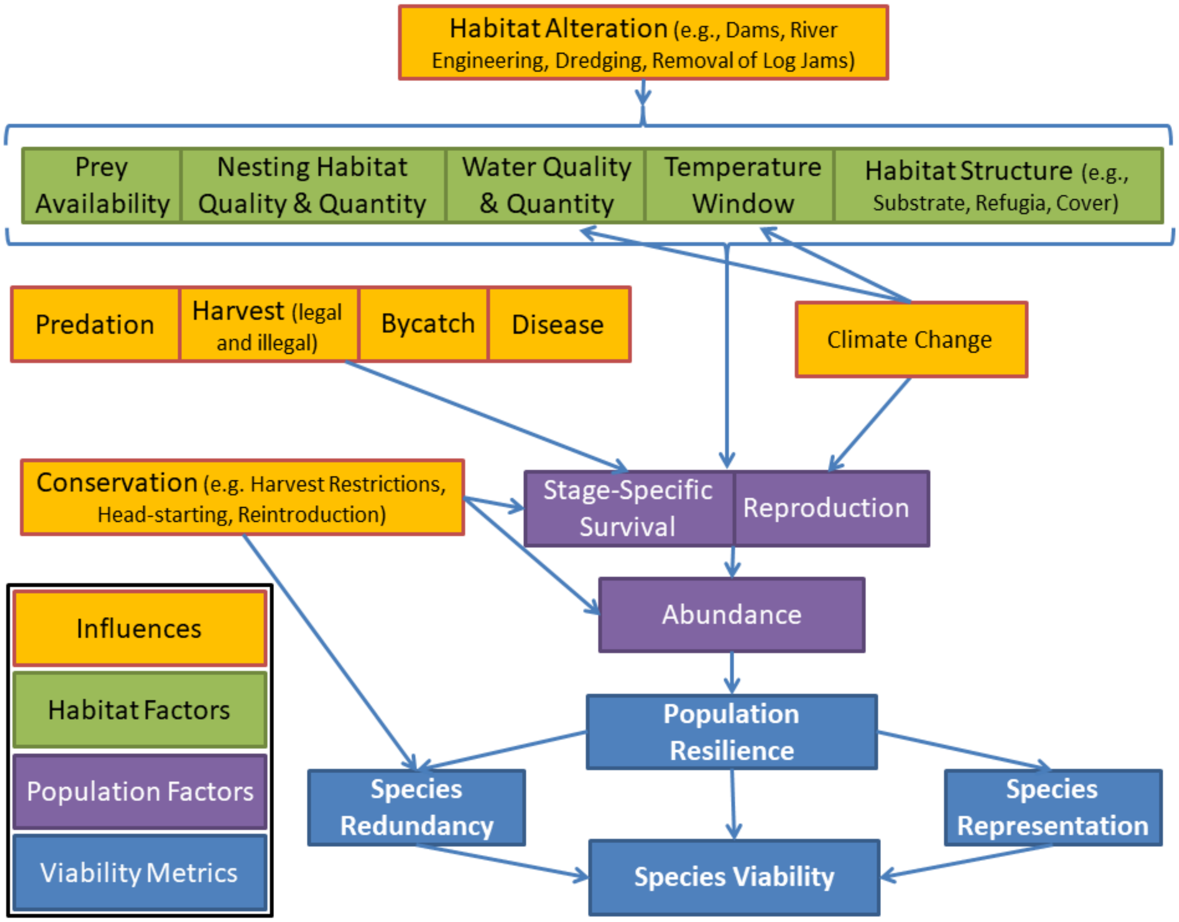


Figure 6. Simplified influence diagram illustrating how various impacts influence habitat and population factors that in turn influence the resilience of populations and viability of the species.

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).

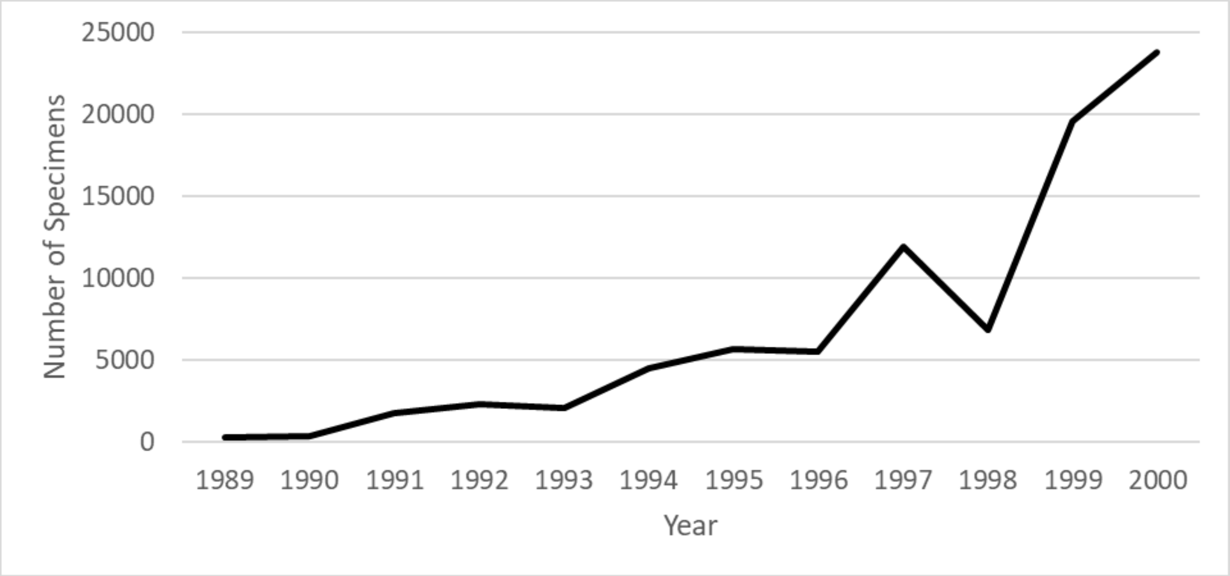


Figure 7. Number of alligator snapping turtle specimens shipped from the U.S. by year from 1989 to 2005 (data from USFWS 2005, p. 74702).

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).

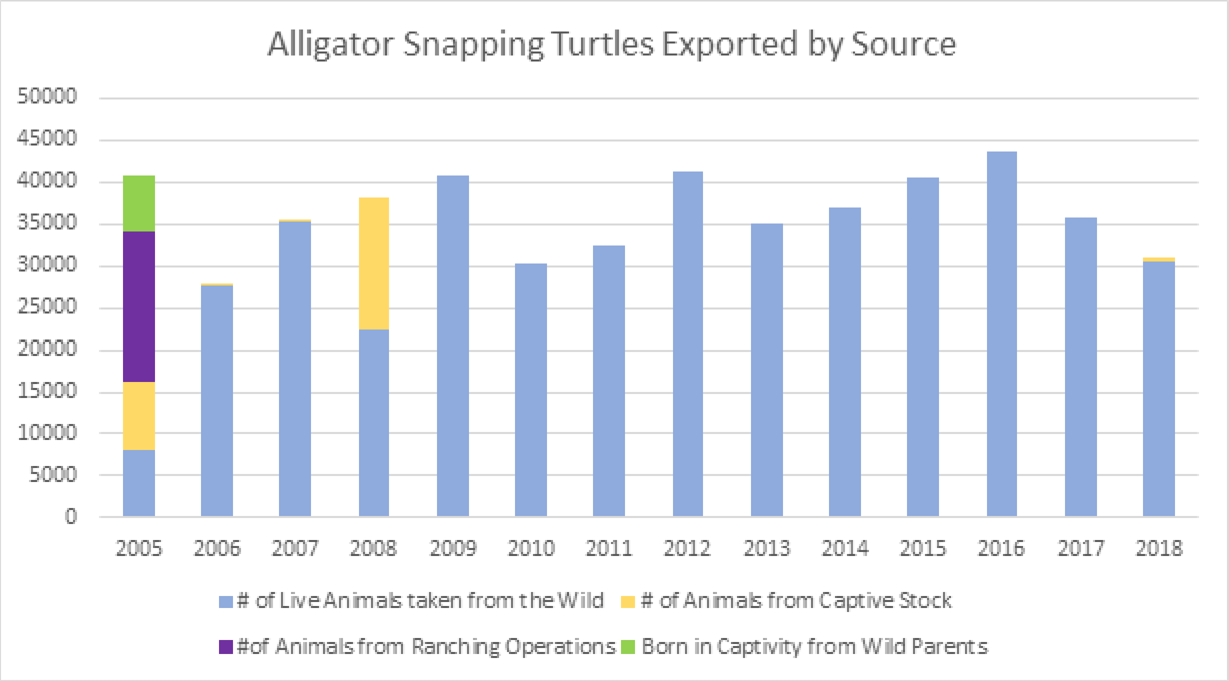


Figure 8. Alligator snapping turtle exports from a variety of sources since 2005. Though most of the turtles have been labelled as wild-caught, nearly all were likely sourced from captivity (USFWS 2018).

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).

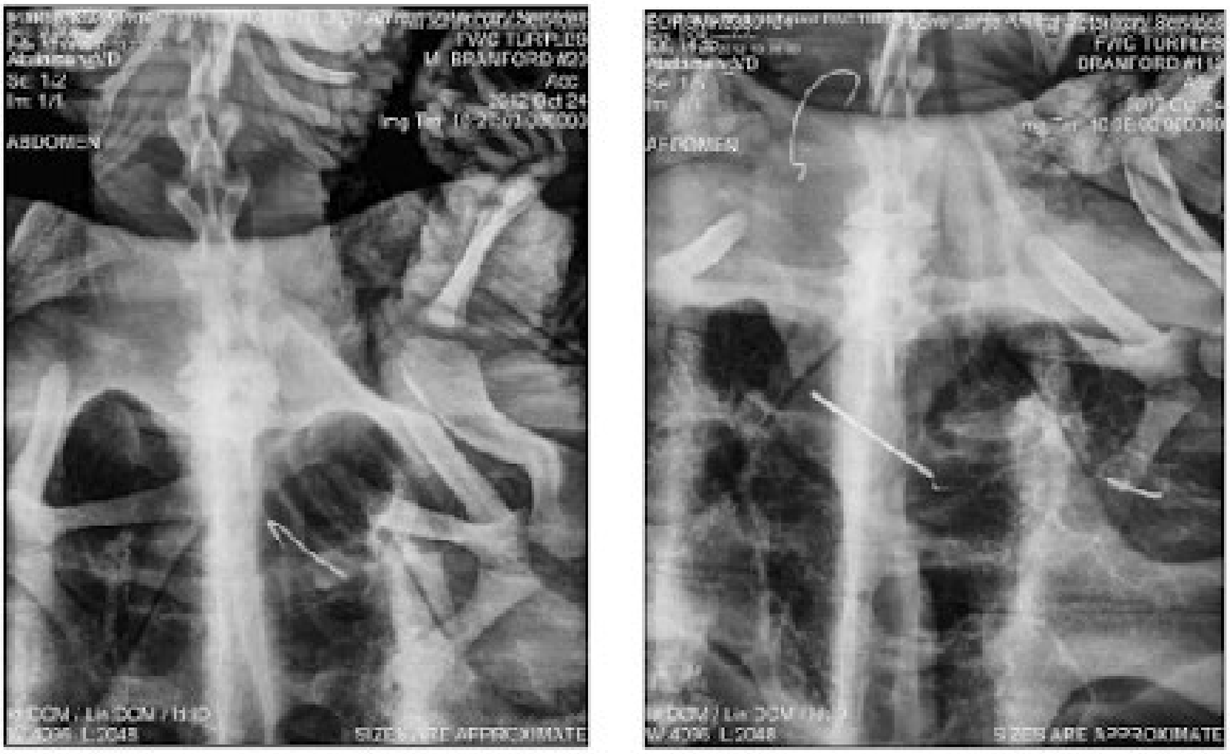


Figure 9. Radiographs of fishing hooks ingested by alligator snapping turtles. Photos from Enge et al. 2014, p. 32.

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.

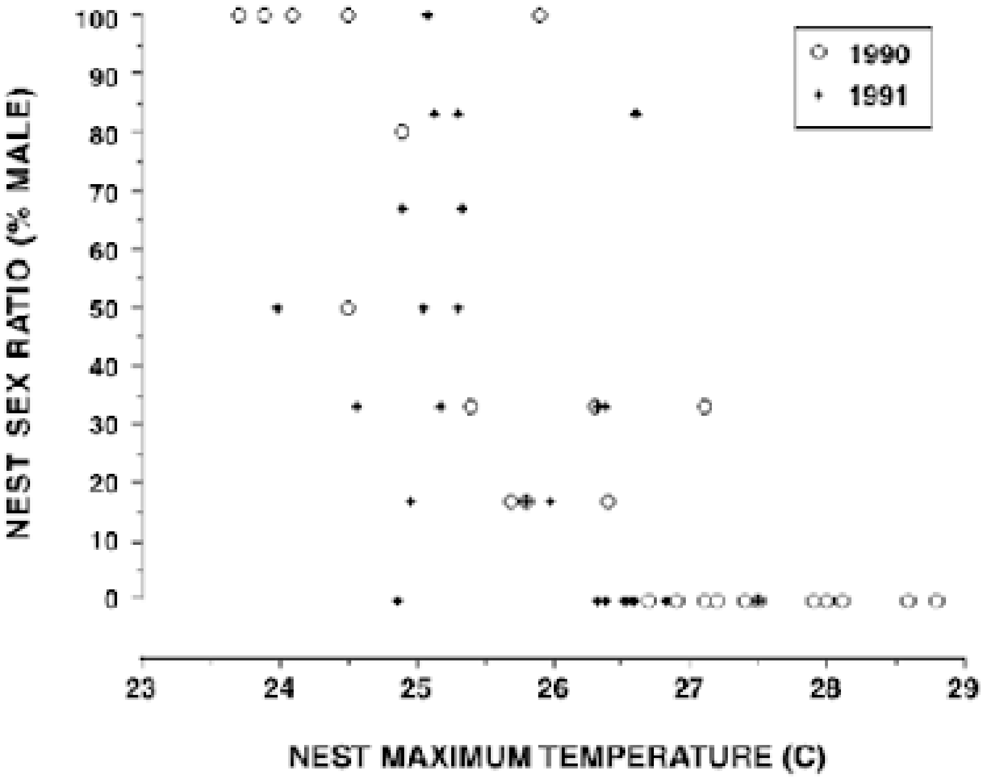


Figure 11. Hatchling sex ratios in nests of alligator snapping turtles in relation to nest temperature. Figure from Ewert and Jackson 1994, p. 26.

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).

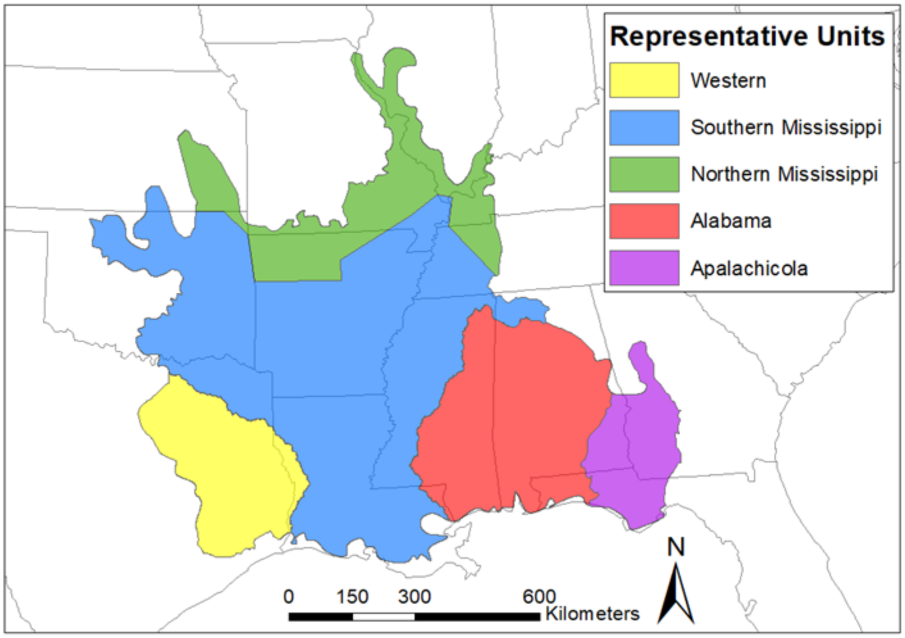


Figure 12. Alligator snapping turtle representative units.

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.

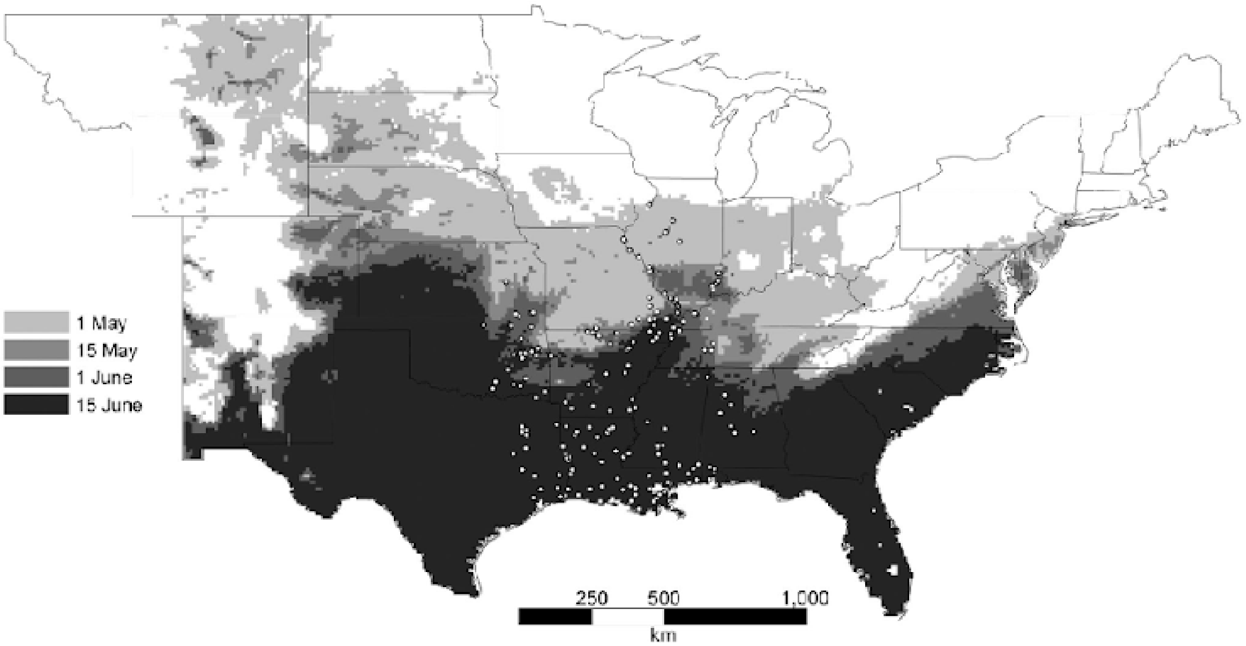


Figure 13. Areas predicted to be suitable for complete embryonic development of alligator snapping turtle eggs based on number of suitable degree days under four nesting scenarios with different nesting initiation dates: 1 May, 15 May, 1 June, and 15 June (dots are alligator snapping turtle occurrences) (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.

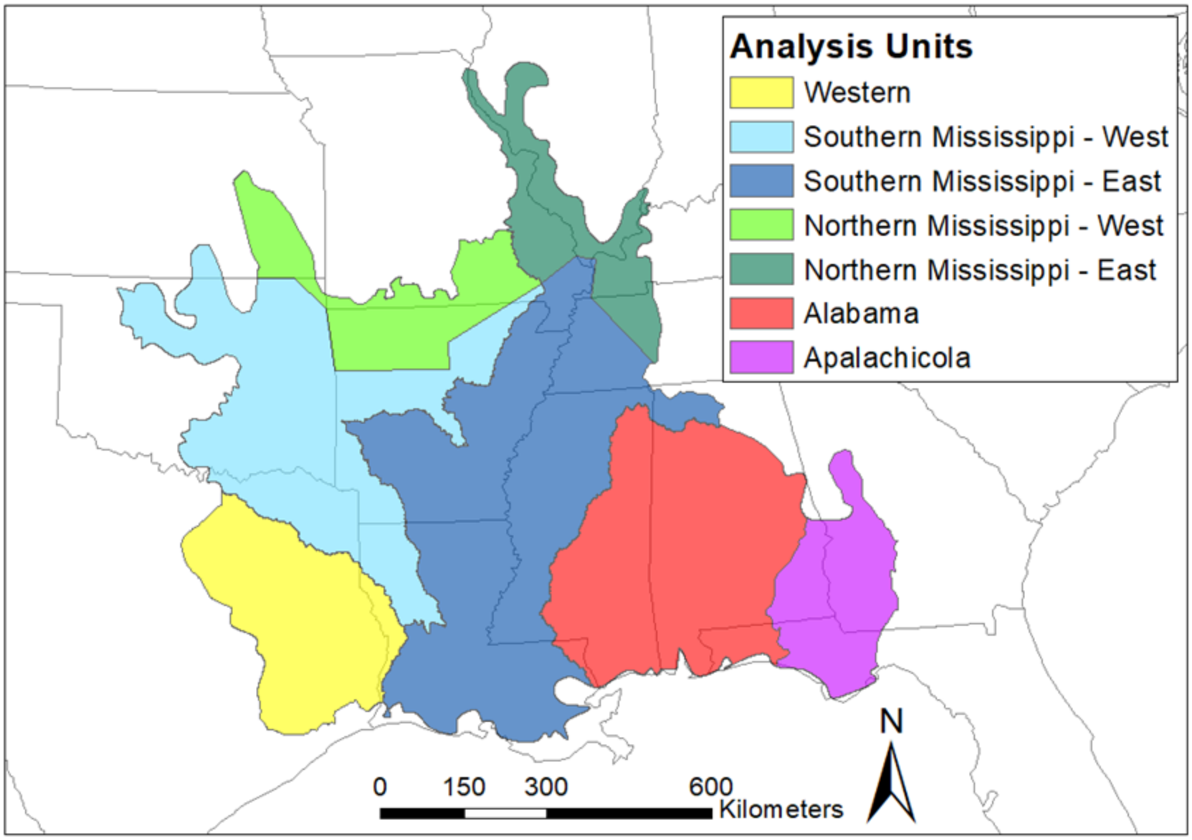


Figure 14. Alligator snapping turtle analysis units. The two Southern Mississippi units (blues) make up one representative unit and the two Northern Mississippi units (greens) make up one representative unit; the remaining analysis units each make up a single representative unit.

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.

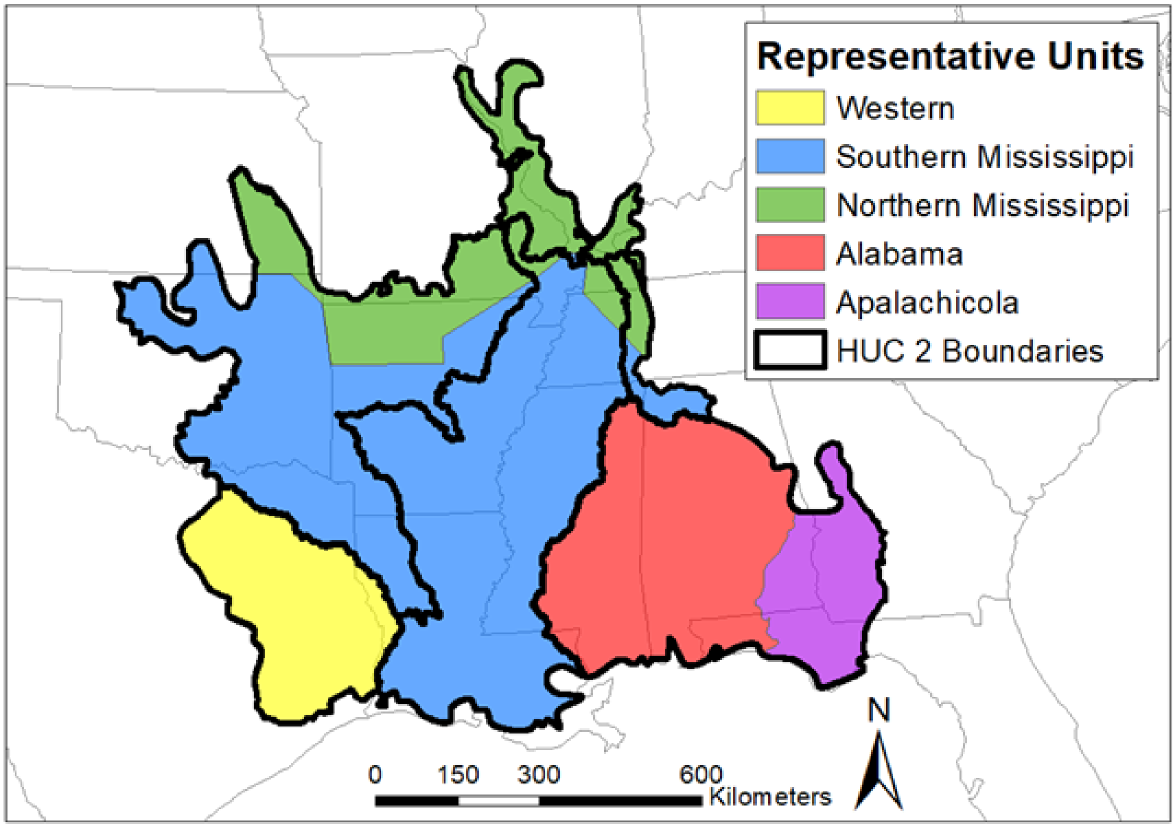


Figure 15. HUC 2 watershed boundaries within alligator snapping turtle representative units.

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.

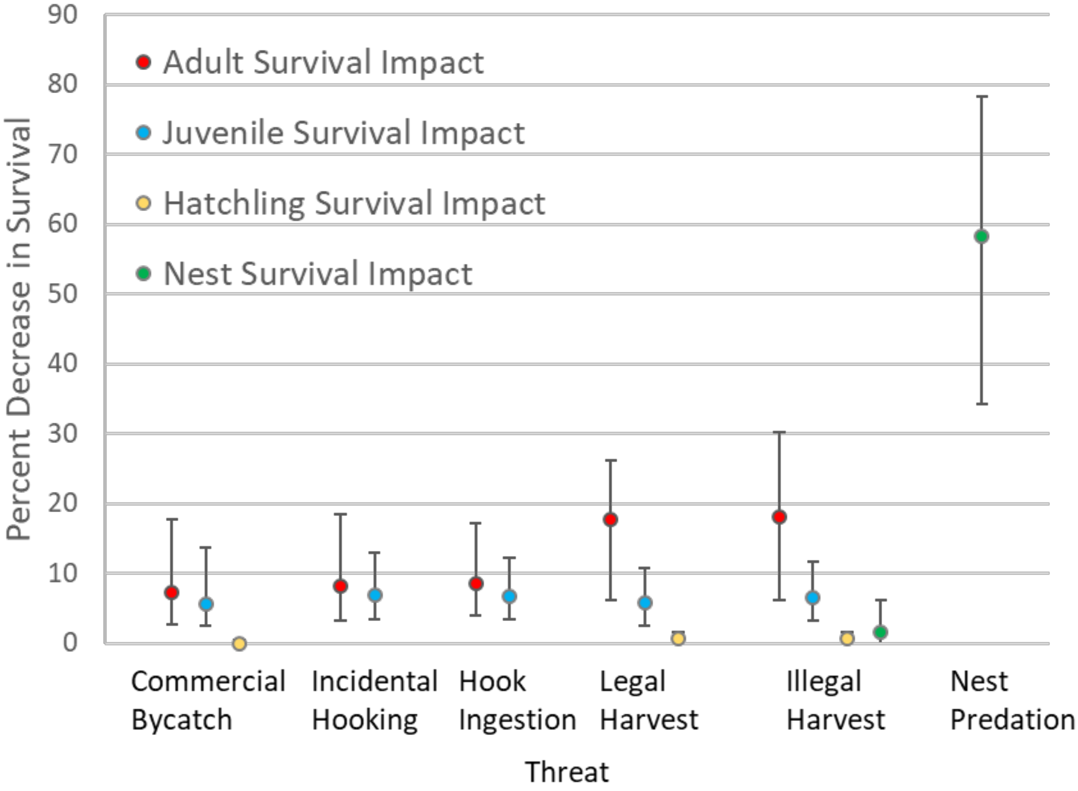


Figure 16. Expert-elicited magnitude of threats facing alligator snapping turtles in terms of the percent decrease to survival rates. Error bars indicate the average of lowest likely and highest likely estimates of impacts on survival, while circles indicate the average of most likely impacts on survival. 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.

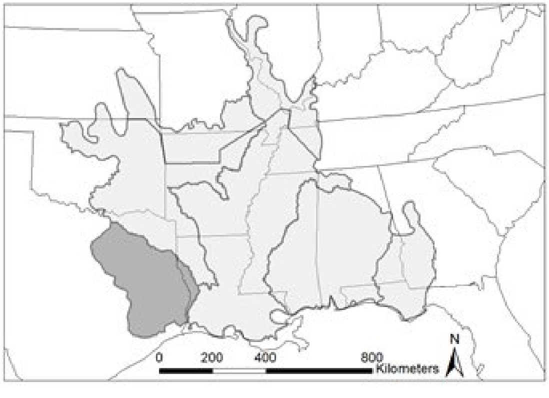


Figure 17. Western Analysis Unit.

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.

Current Threats and Conservation Actions

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.

Comparison with Historical Range

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.

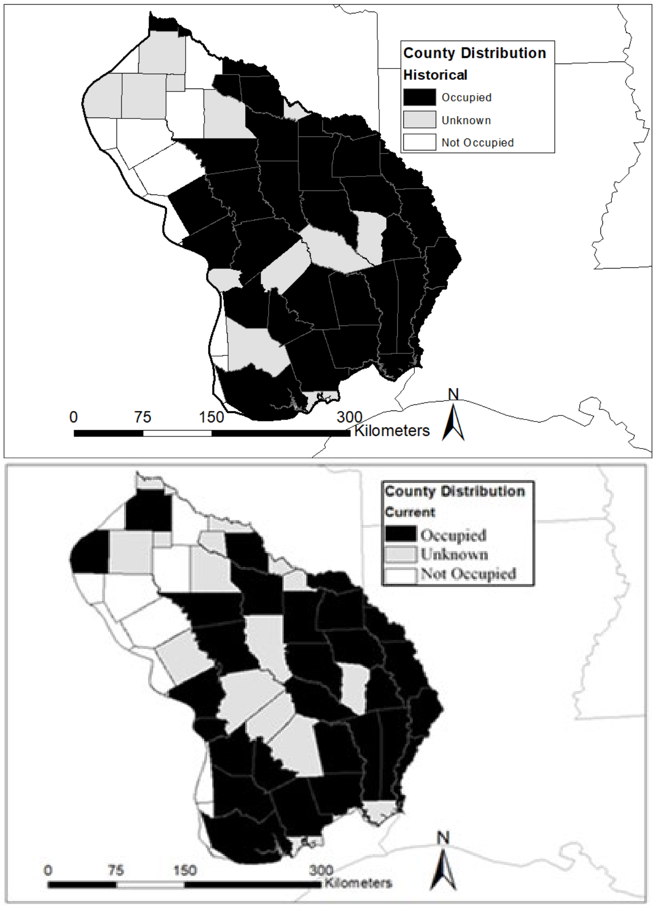


Figure 18. Historical and current distribution of alligator snapping turtles in the Western Analysis Unit.

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.

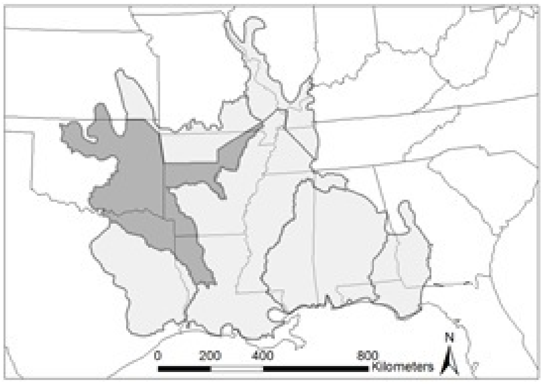


Figure 19. Southern Mississippi – West Analysis Unit

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.

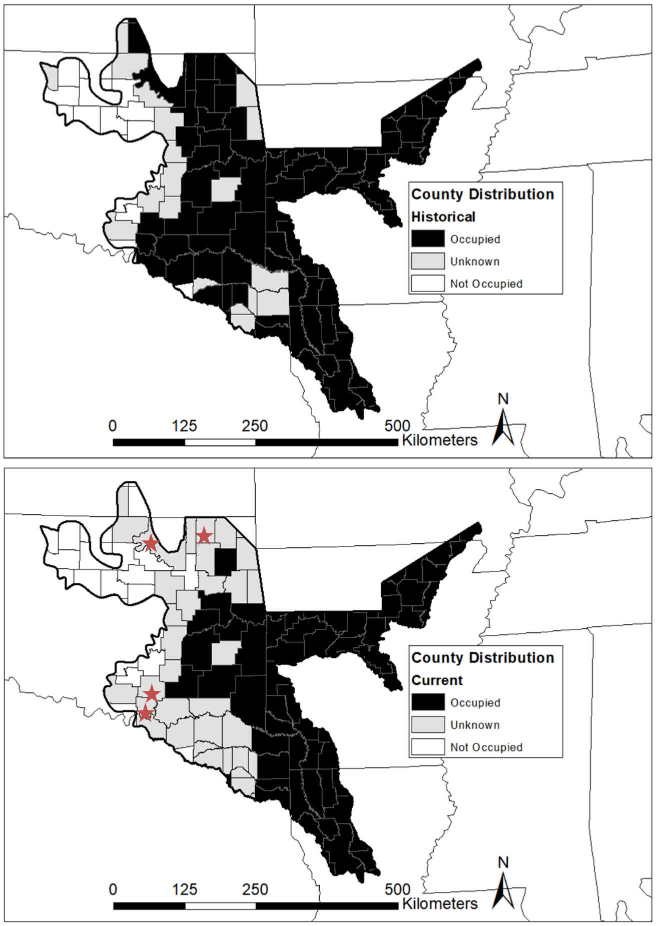


Figure 20. Historical and current distribution of alligator snapping turtles in the Southern Mississippi – West Analysis Unit. Counties in Oklahoma with ongoing reintroductions are indicated with stars.

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.

<u>Current Abundance</u>

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.

<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 (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.

Comparison with Historical Range

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.

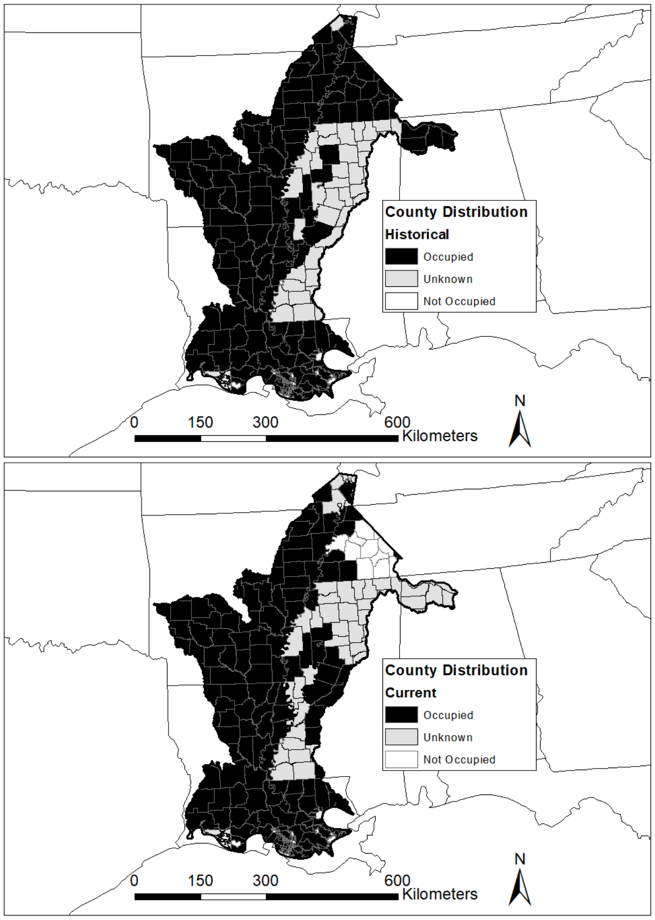


Figure 22. Historical and current distribution of alligator snapping turtles in the Southern Mississippi – East Analysis Unit.

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.

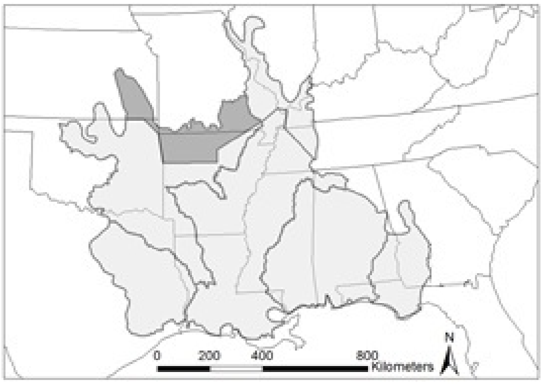


Figure 23. Northern Mississippi – West Analysis Unit.

<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.

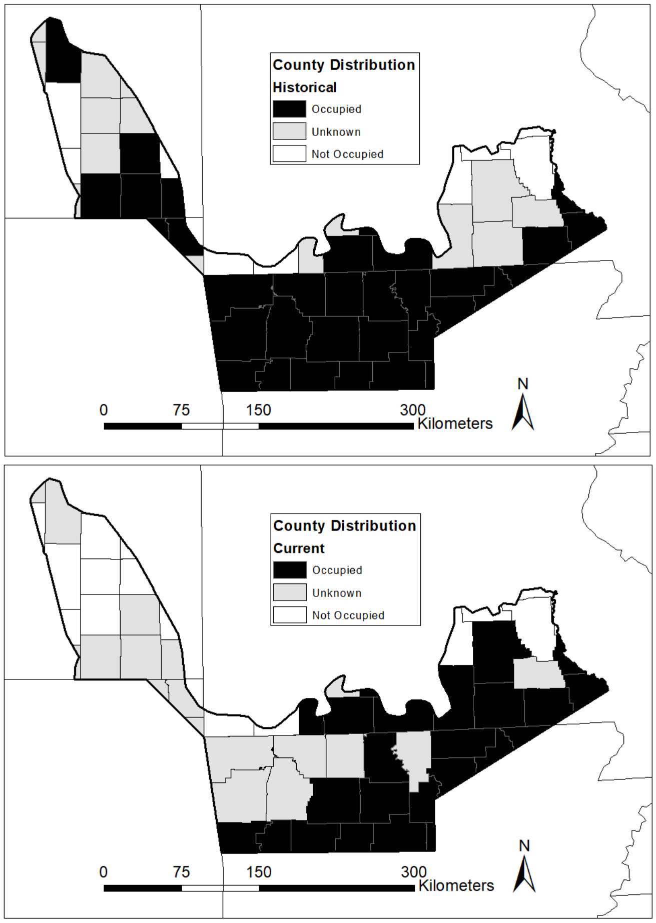


Figure 24. Historical and current distribution of alligator snapping turtles in the Northern Mississippi – West Analysis Unit.

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.

Current Abundance

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

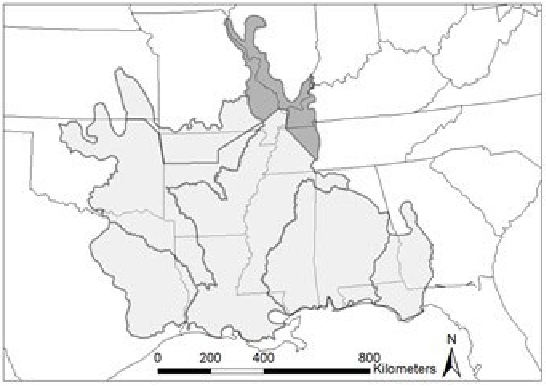


Figure 25. Northern Mississippi – East Analysis Unit

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.

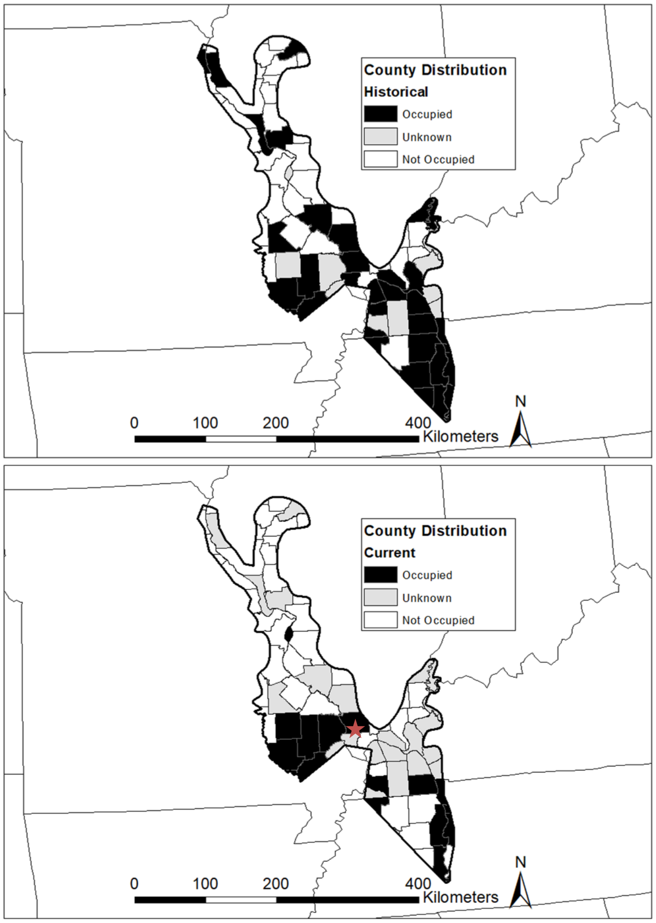


Figure 26. Historical and current distribution of alligator snapping turtles in the Northern Mississippi – East Analysis Unit. Union County in Illinois with ongoing reintroductions is indicated with a star.

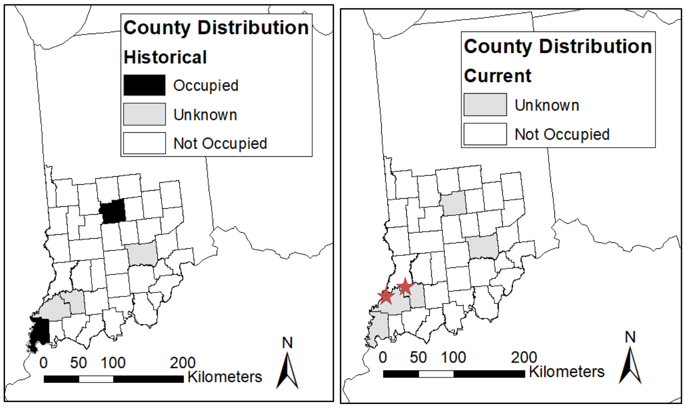


Figure 27. Historical and current distribution of alligator snapping turtles in Indiana. Stars indicate counties where environmental DNA from alligator snapping turtles was detected in 2017.

4.6.6 Alabama Analysis Unit

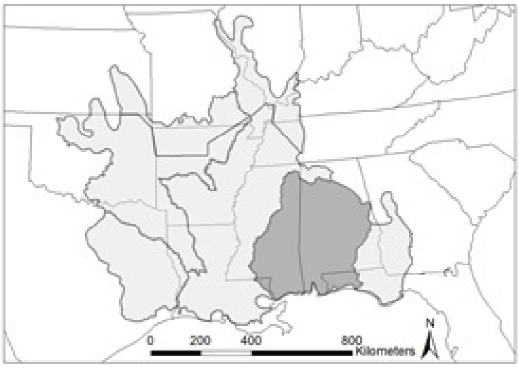


Figure 28. Alabama Analysis Unit.

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.

Current Abundance

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.

Current Threats and Conservation Actions

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.

Comparison with Historical Range

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.

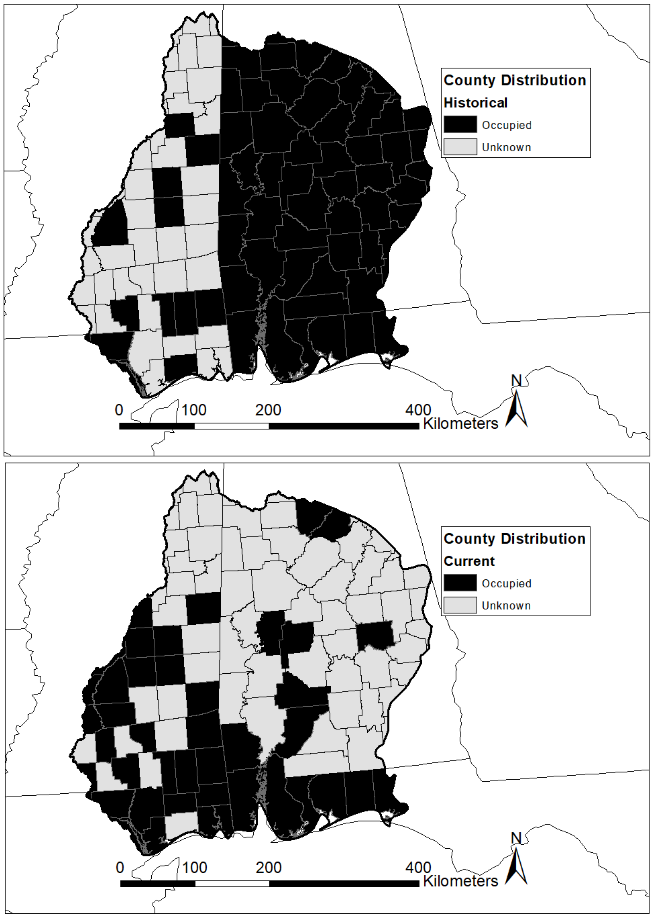


Figure 29. Historical and current distribution of alligator snapping turtles in the Alabama Analysis Unit.

4.6.7 Apalachicola Analysis Unit

Figure 30. 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.

Current Abundance

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).

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 (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.

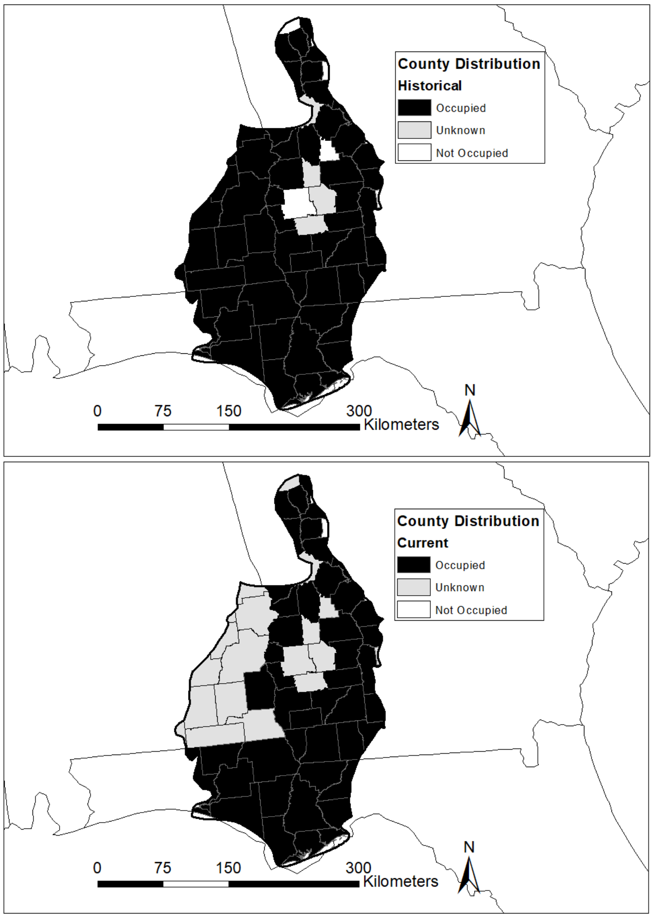


Figure 31. Historical and current distribution of alligator snapping turtles in the Apalachicola Analysis Unit.

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).

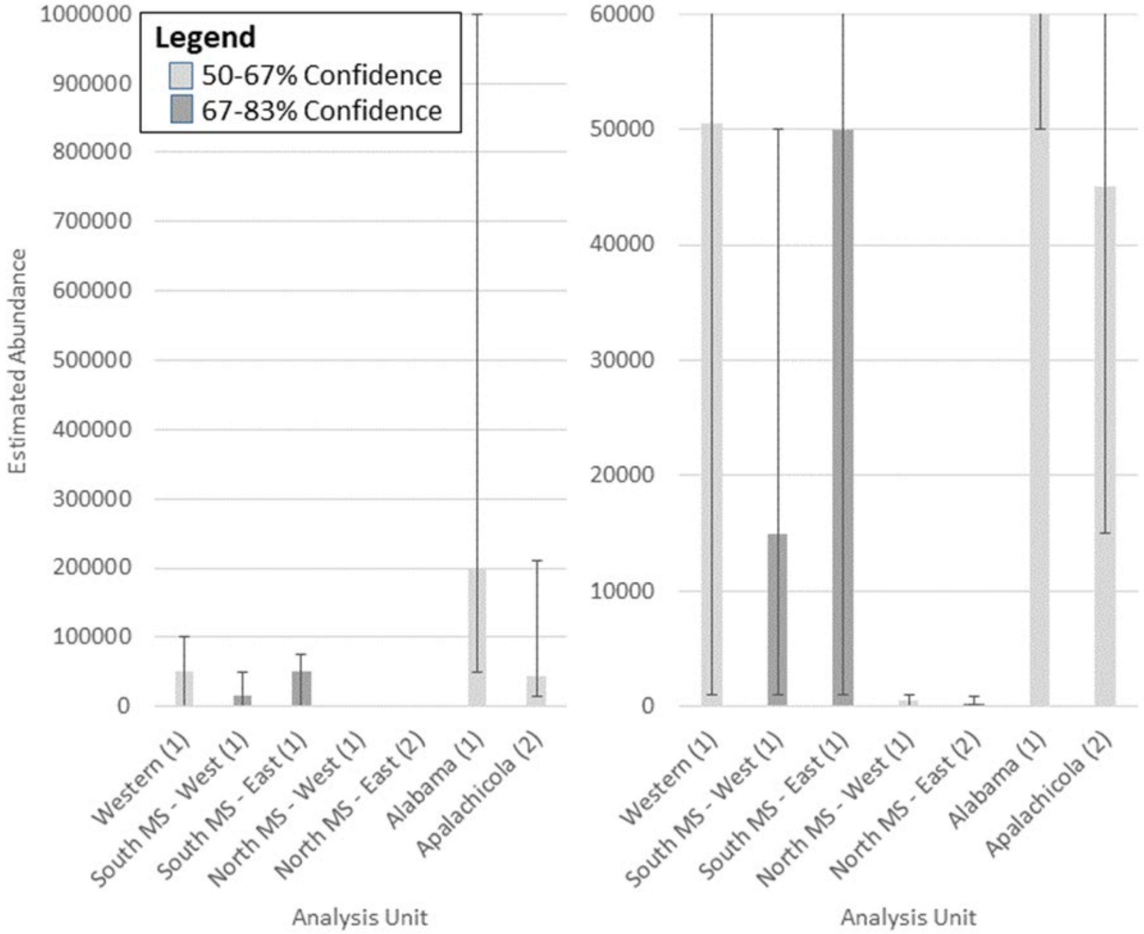


Figure 32. Estimated abundance of alligator snapping turtles in each analysis unit. Y-axis zoomed in on right. Darker bars show higher confidence of species experts in their estimates, and the number of experts that provided estimates for each unit is indicated in parentheses. Though the bars cannot 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.

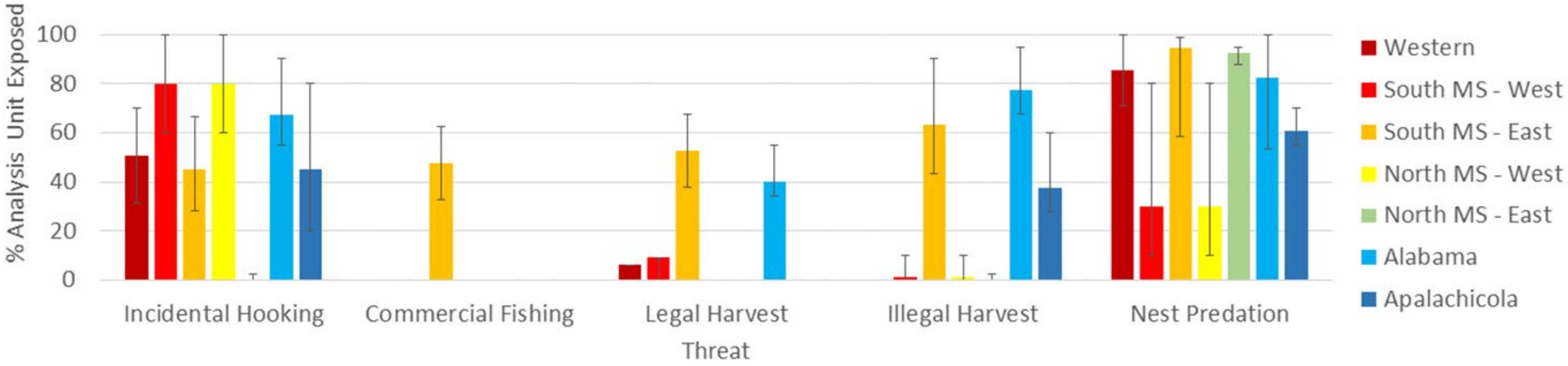

% Analysis Unit Exposed
100
80
60
40
20
0
Incidental Hooking
Commercial Fishing
Legal Harvest
Illegal Harvest
Nest Predation
Threat
Western
South MS - West
South MS - East
North MS - West
North MS - East
Alabama
Apalachicola

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).

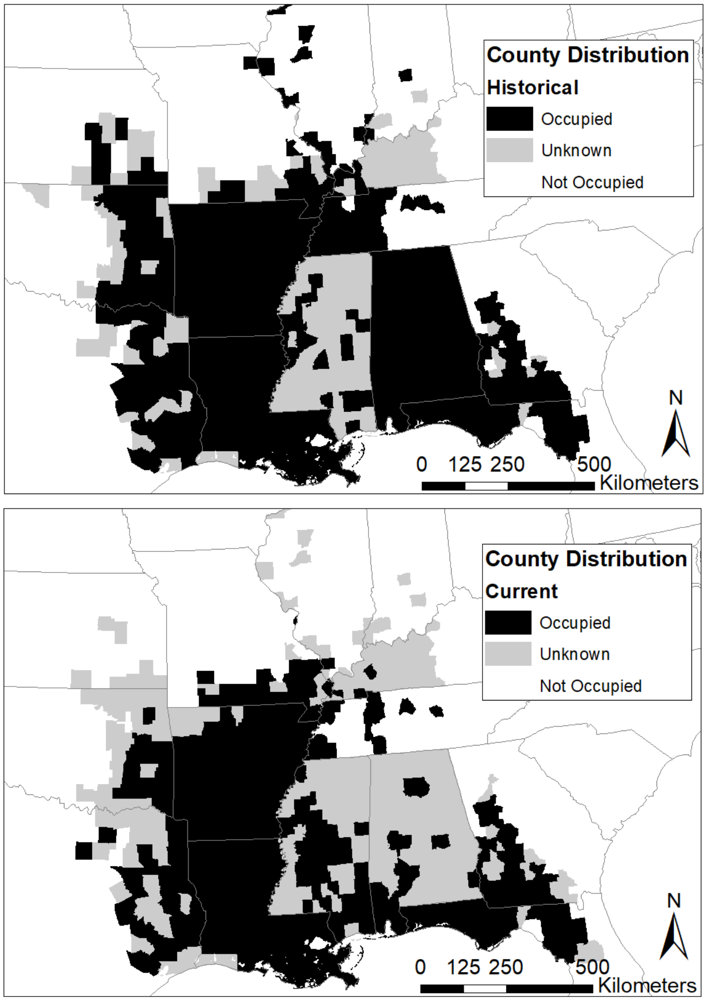


Figure 34. Historical and current alligator snapping turtle and Suwannee snapping turtle distribution by county or parish. Unknown status can be caused by a lack of recent surveys to detect turtles where they likely still exist (especially in core of the species' range), or can represent counties or 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).

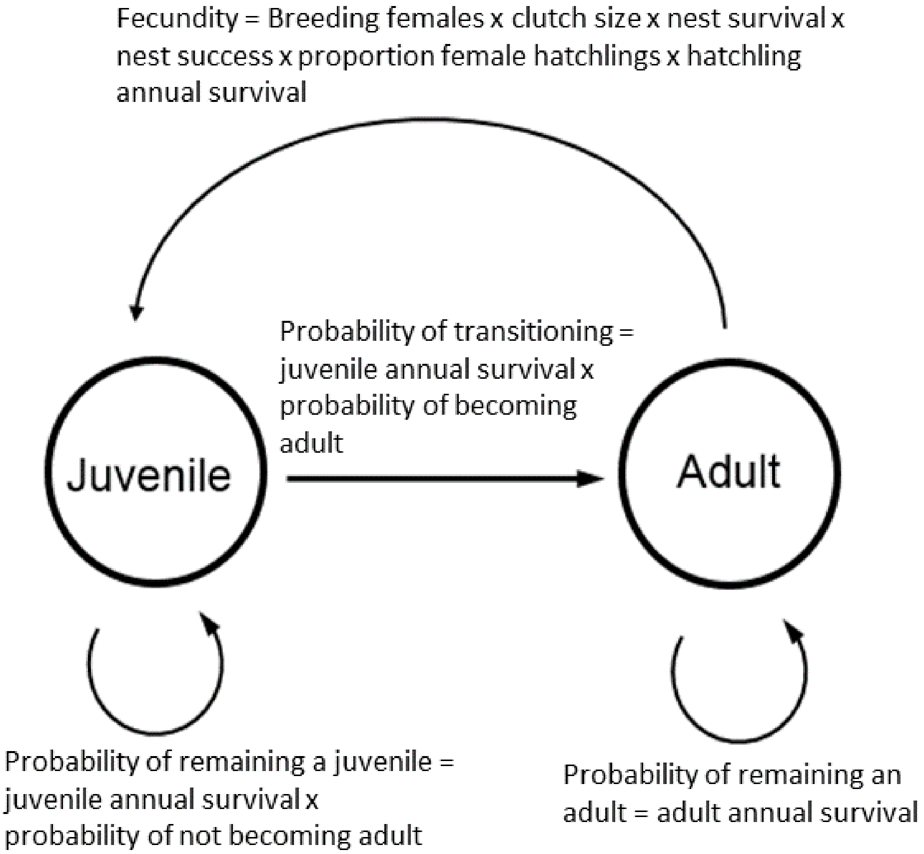


Figure 35. Alligator snapping turtle life cycle diagram for a female only two-stage pre-breeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals). At each time step, juveniles can remain in their current stage, which is the 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 $\lambda = 0.85$, 0.81, and 0.78 respectively for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, and mean $\lambda = 0.87$, 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).

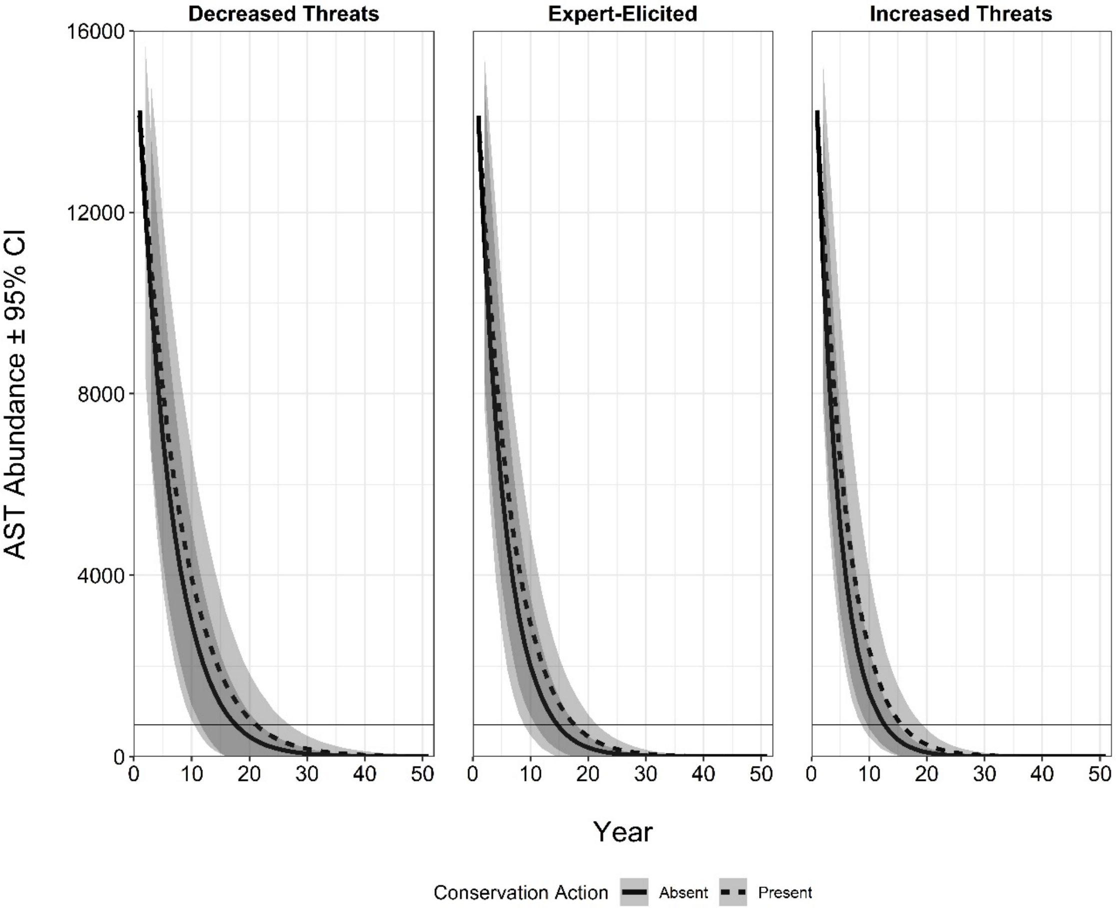


Figure 36. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Southern Mississippi – East Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% 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).

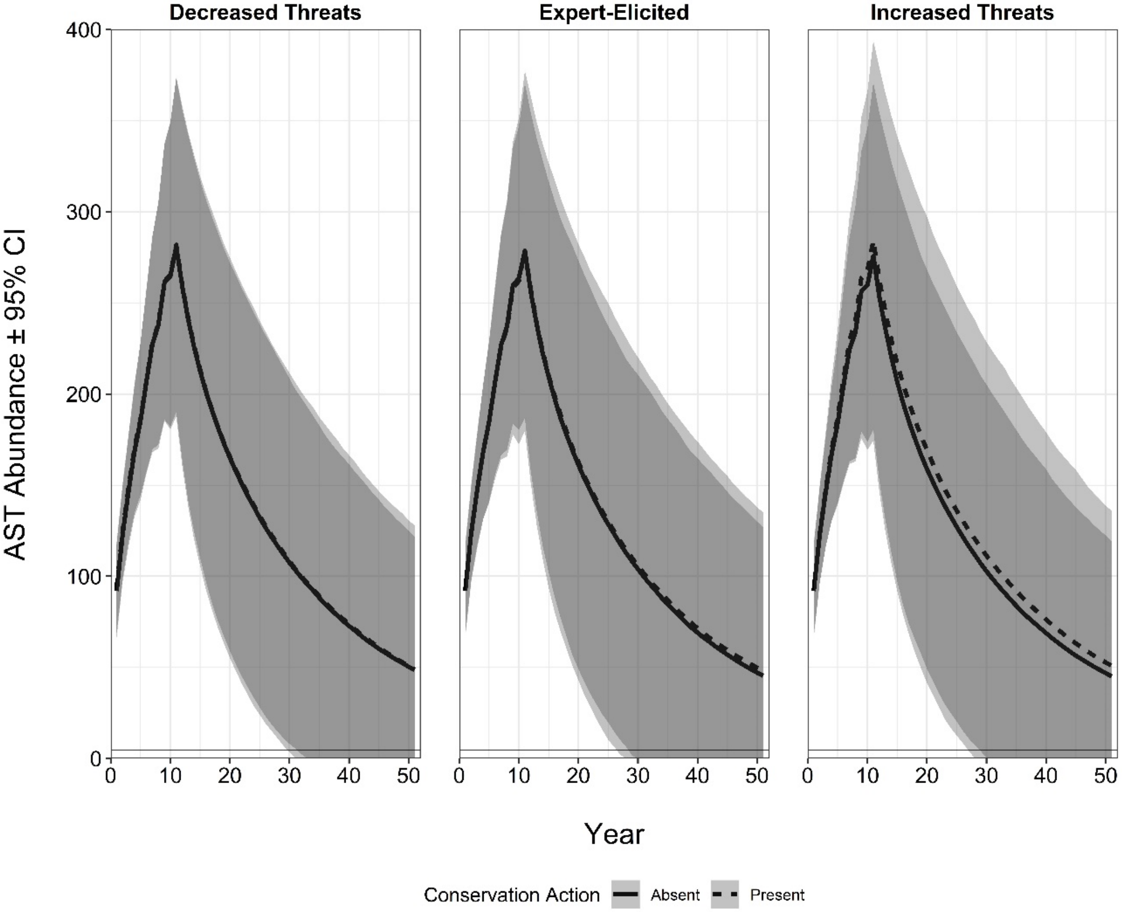


Figure 37. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Northern Mississippi – East Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% 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 $\lambda = 0.83$, 0.78, and 0.75 respectively for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, and mean $\lambda = 0.86$, 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).

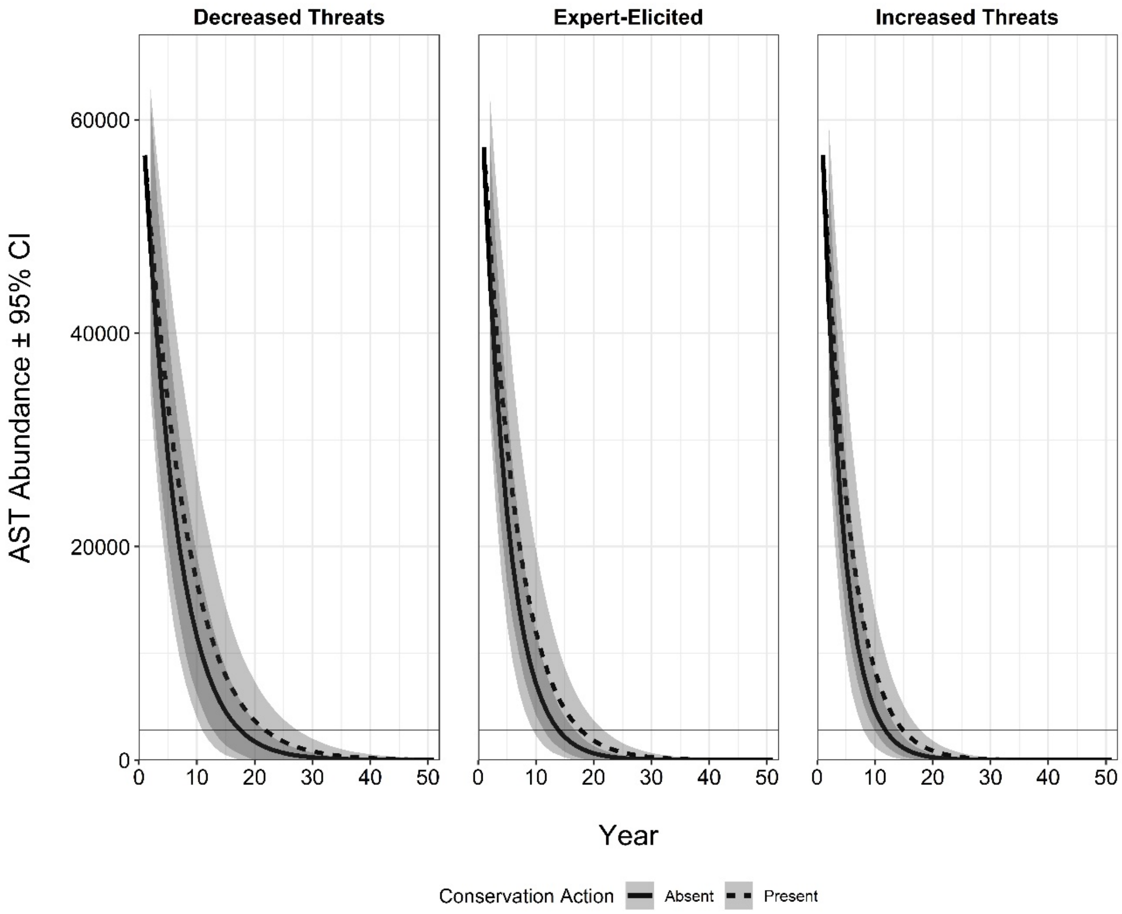


Figure 38. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Alabama Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (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	51 ± *	1.0	1.0	17.68 ± 2.27	22.84 ± 3.20
			(43, 51)	(51, 51)			(12, 29)	(14, 33)
Expert-Elicited	0.846	0.114	45.64 ± 3.36	49.14 ± 2.23	1.0	1.0	14.20 ± 1.6	17.91 ± 2.27
			(36, 51)	(40, 51)			(10, 20)	(13, 26)
Increased	1.0	0.658	40.19 ± 3.47	47.21 ± 2.76	1.0	1.0	12.11 ± 1.35	15.11 ± 1.72
			(30, 51)	(40, 51)			(8, 16)	(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).

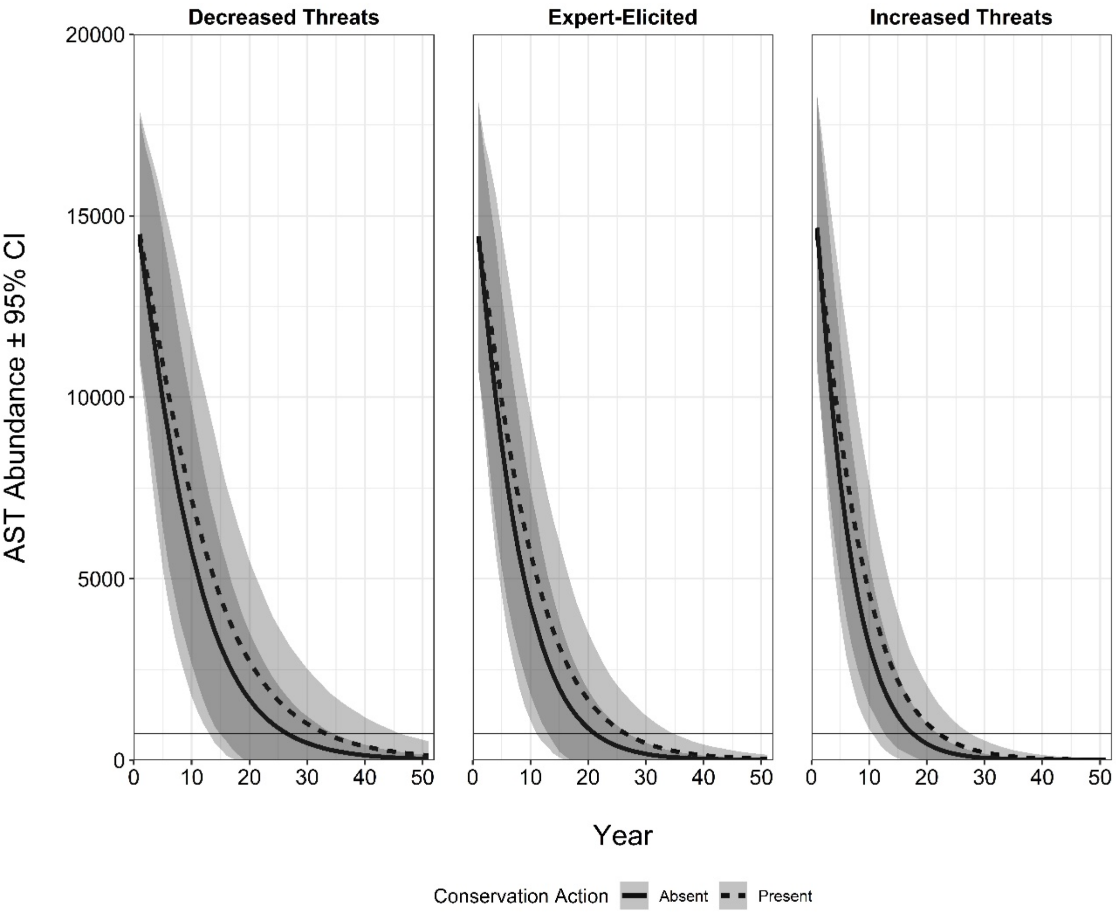


Figure 39. Simulated alligator snapping turtle total abundance (females only, adults and juveniles) over a 50-year period within the Apalachicola Analysis Unit. The curved lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (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.

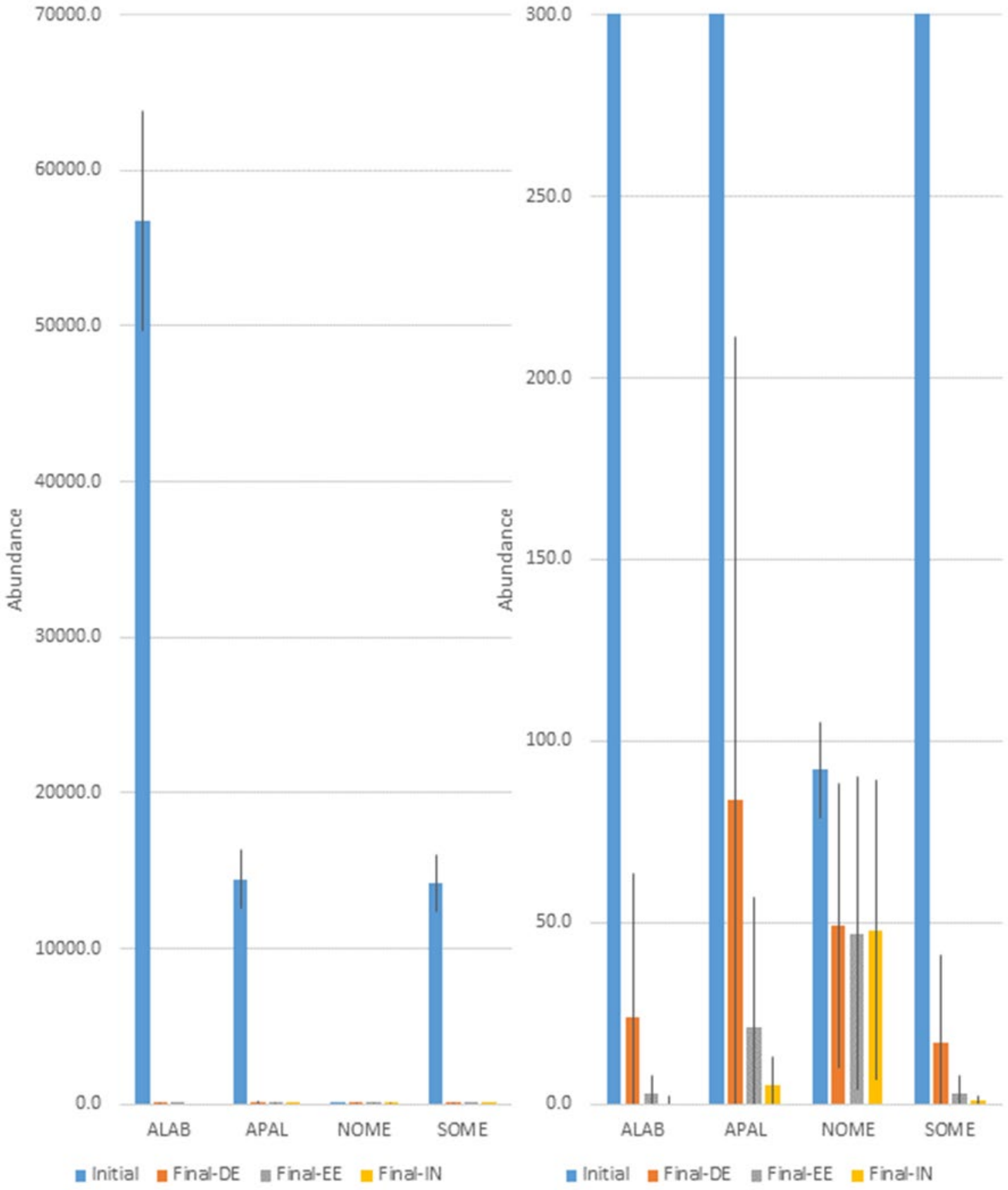


Figure 40. Initial and final projected alligator snapping turtle abundances with standard deviations (zoomed in on right panel). The four modeled analysis units are shown (ALAB = Alabama, APAL = Apalachicola, NOME = Northern Mississippi – East, SOME = Southern Mississippi – East), with initial female-only 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.

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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.

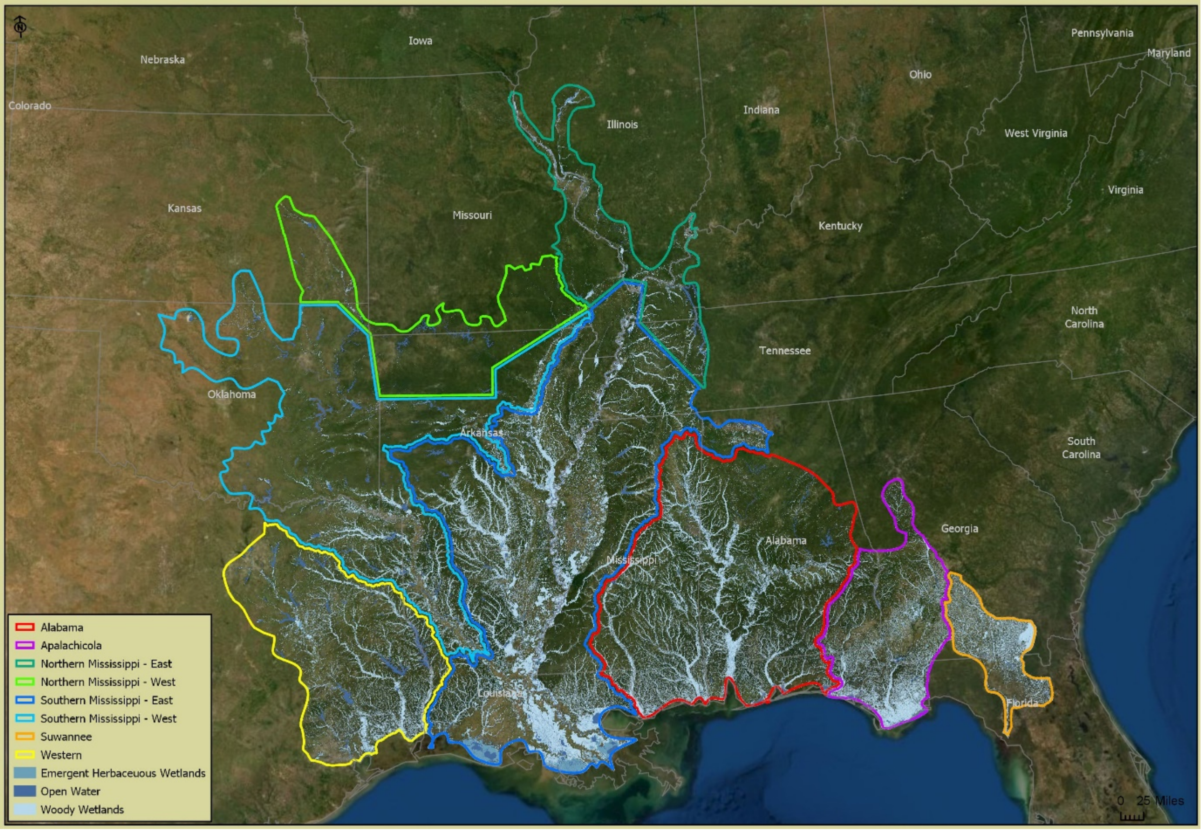

Figure A1. Suitable alligator snapping turtle habitat within the range of the species.

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%**

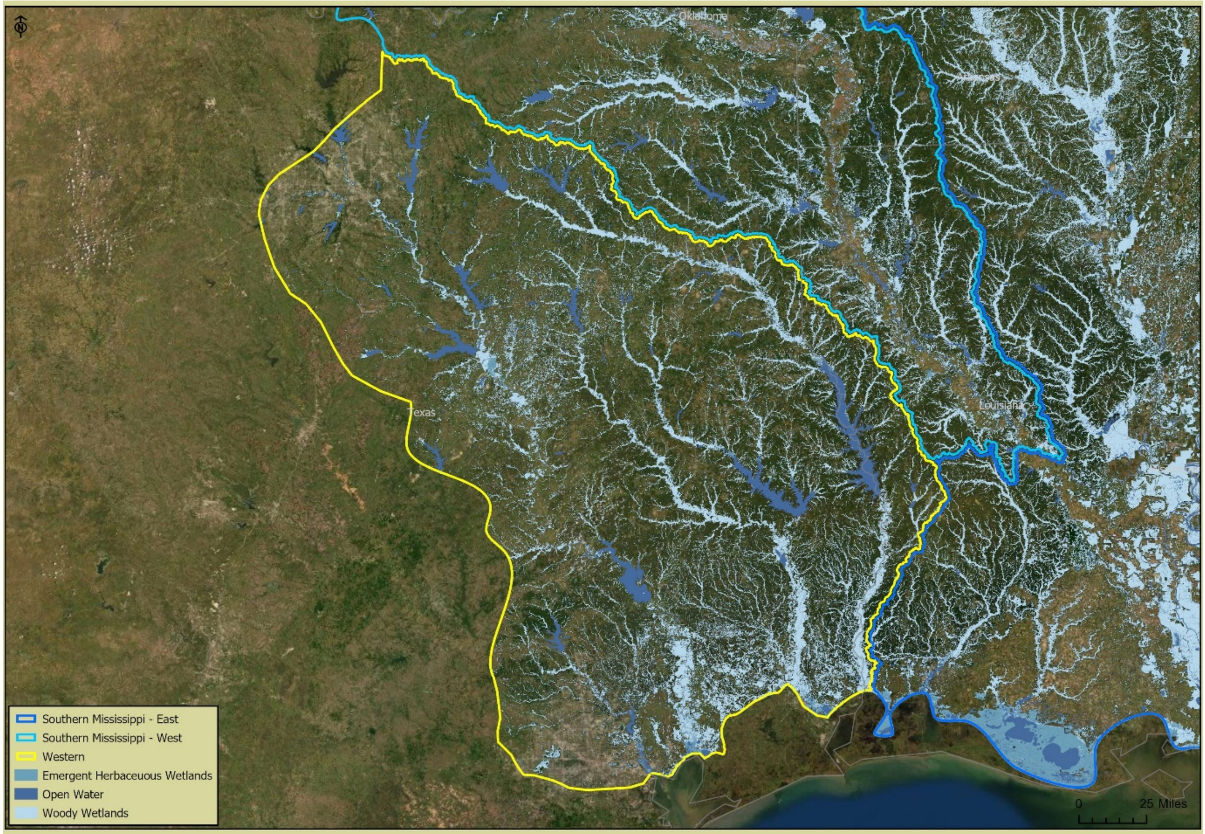

Figure A2. Suitable alligator snapping turtle habitat within the Western Unit.

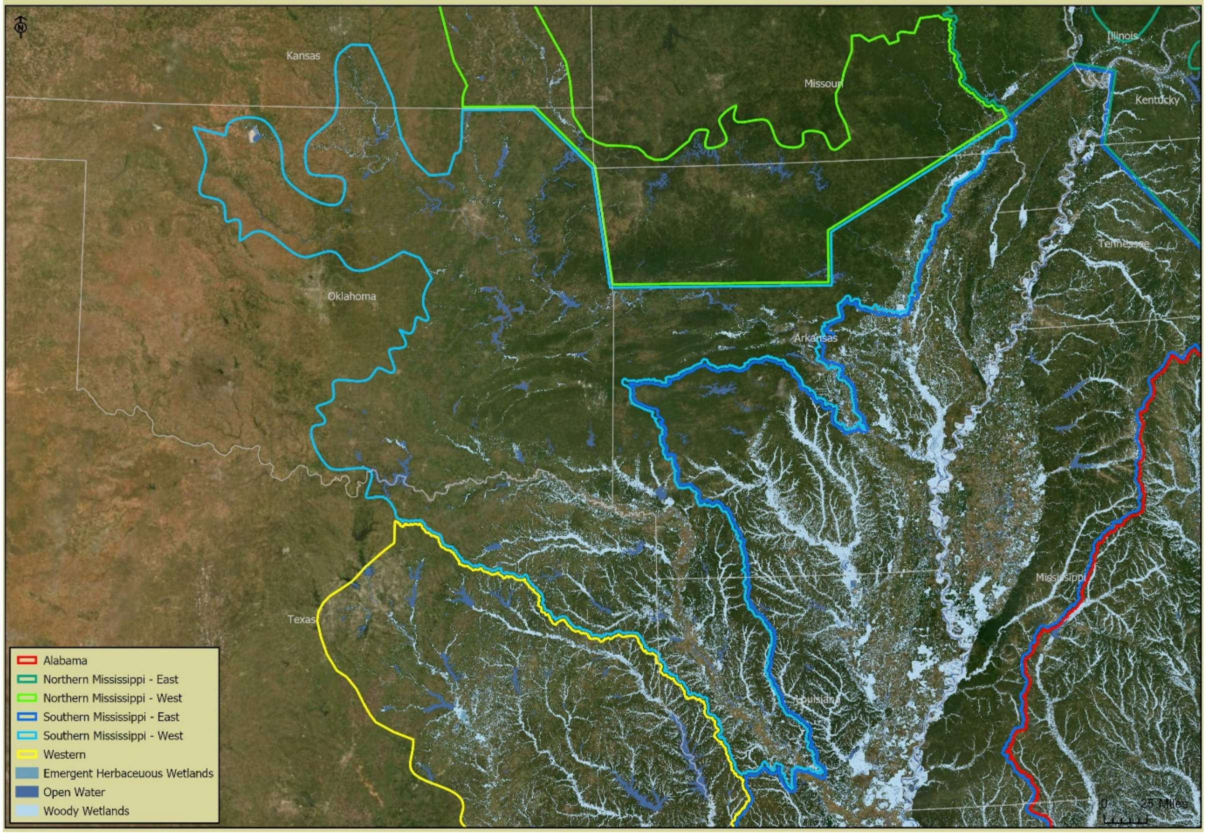

Figure A3. Suitable alligator snapping turtle habitat within the Southern Mississippi – West Unit.

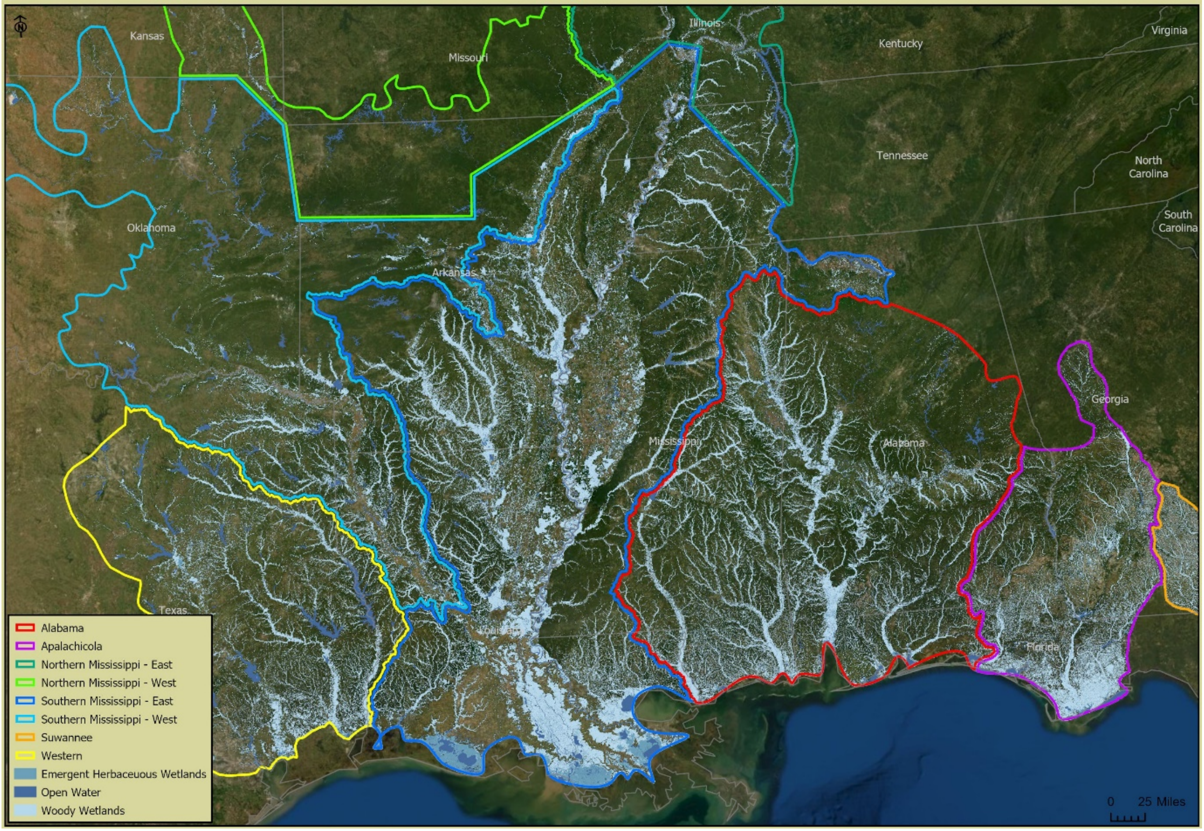

Figure A4. Suitable alligator snapping turtle habitat within the Southern Mississippi – East Unit.

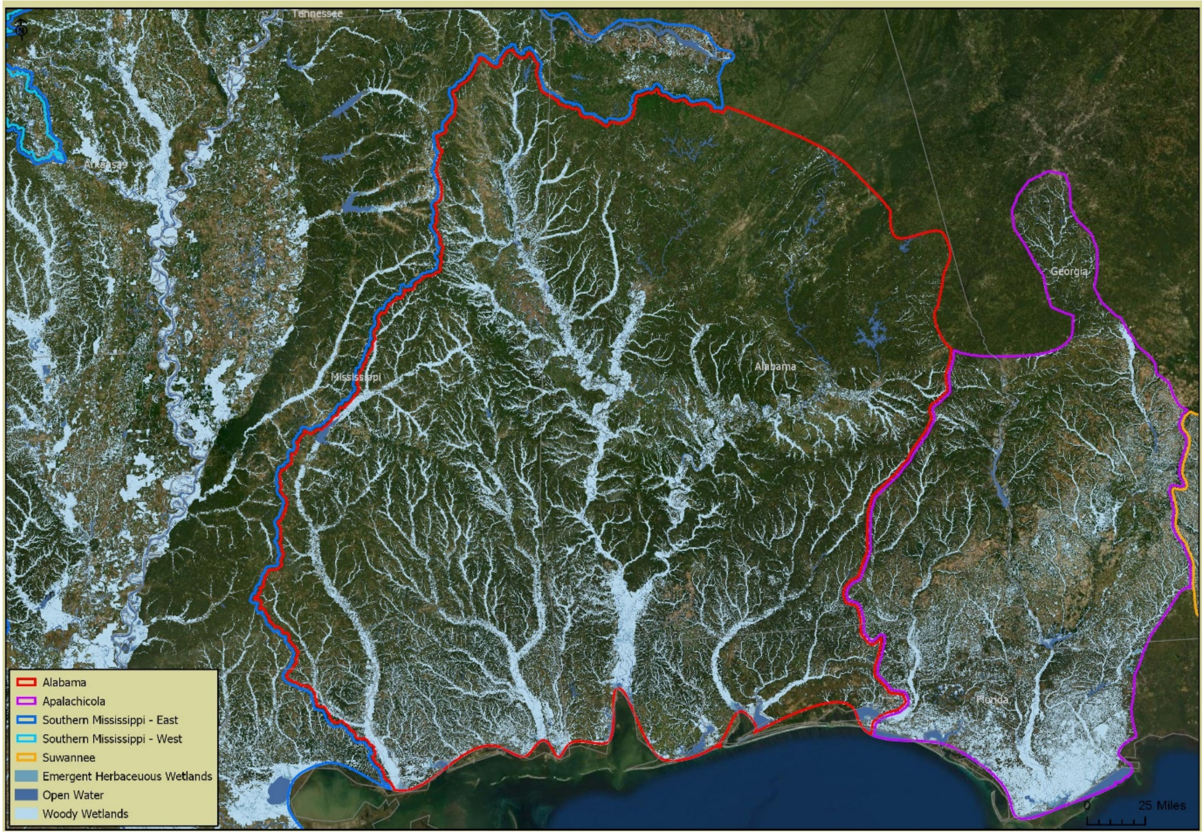

Figure A5. Suitable alligator snapping turtle habitat within the Alabama Unit.

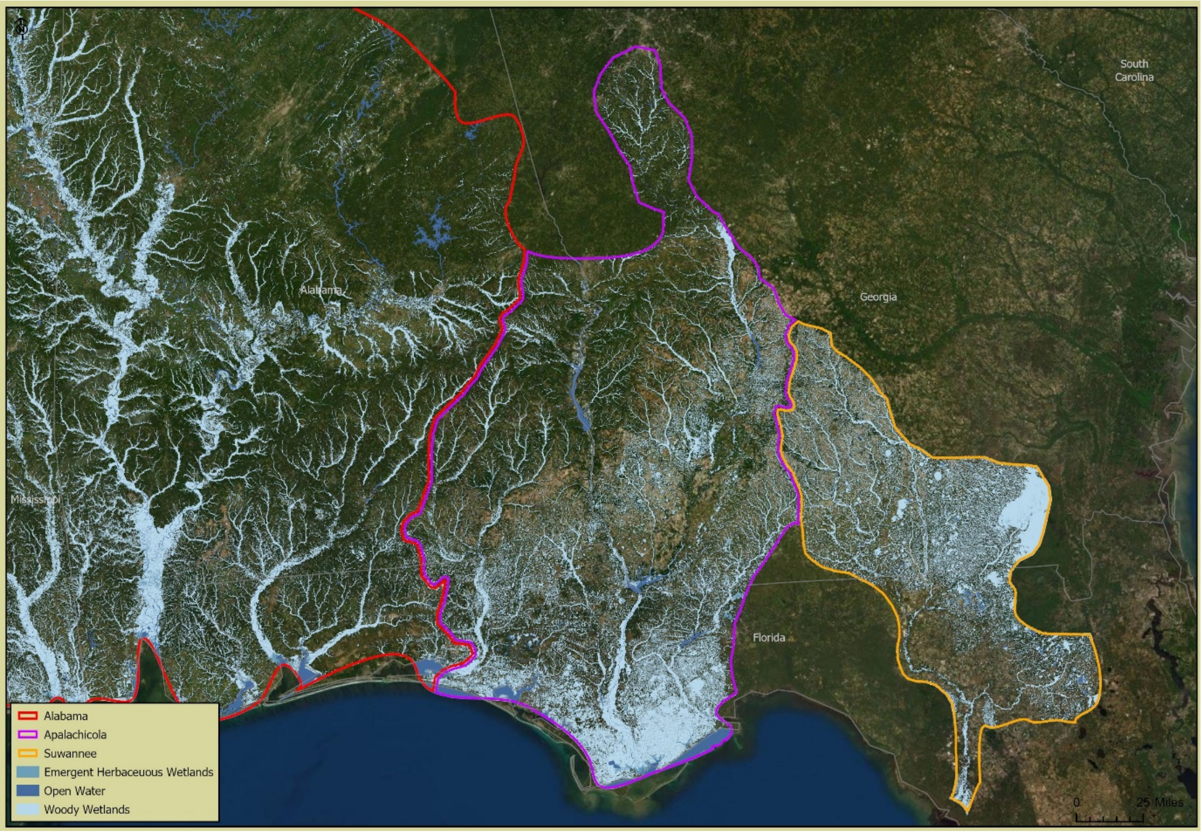

Figure A6. Suitable alligator snapping turtle habitat within the Apalachicola Unit.

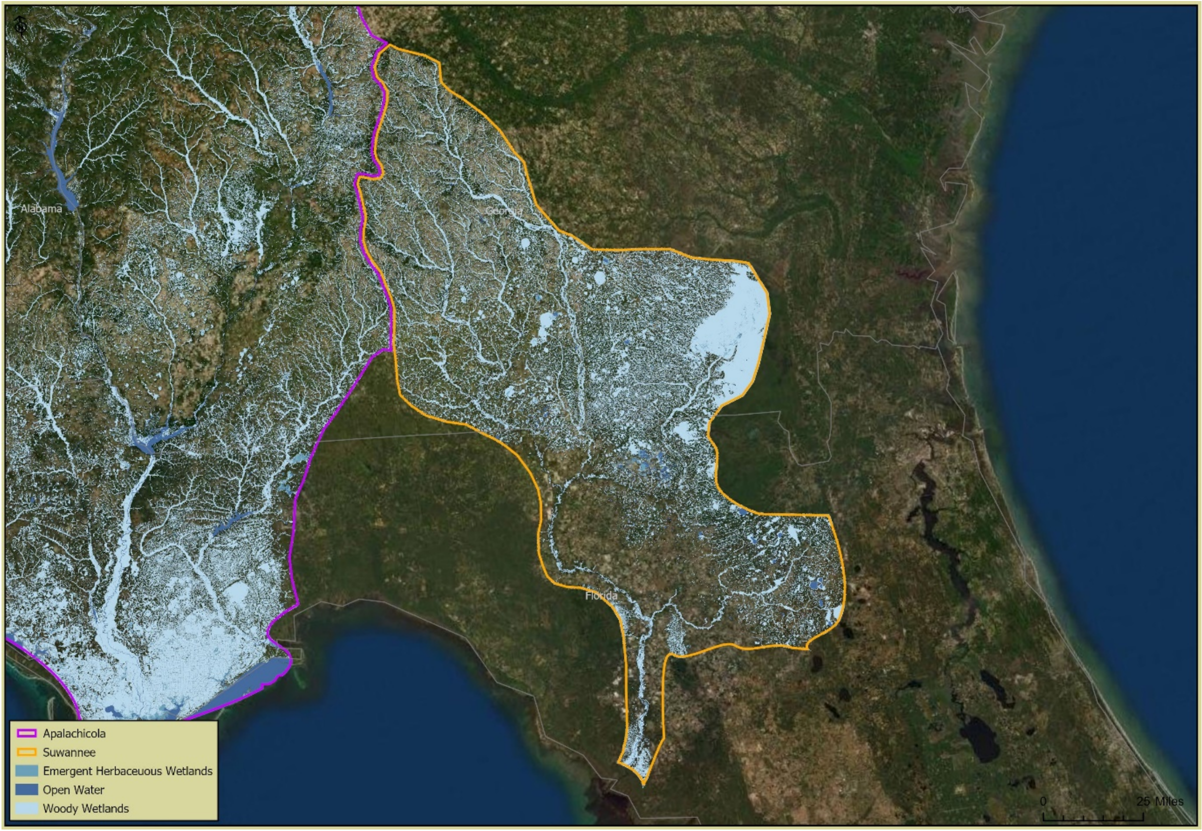

Figure A7. Suitable alligator snapping turtle habitat within the Suwannee Unit.

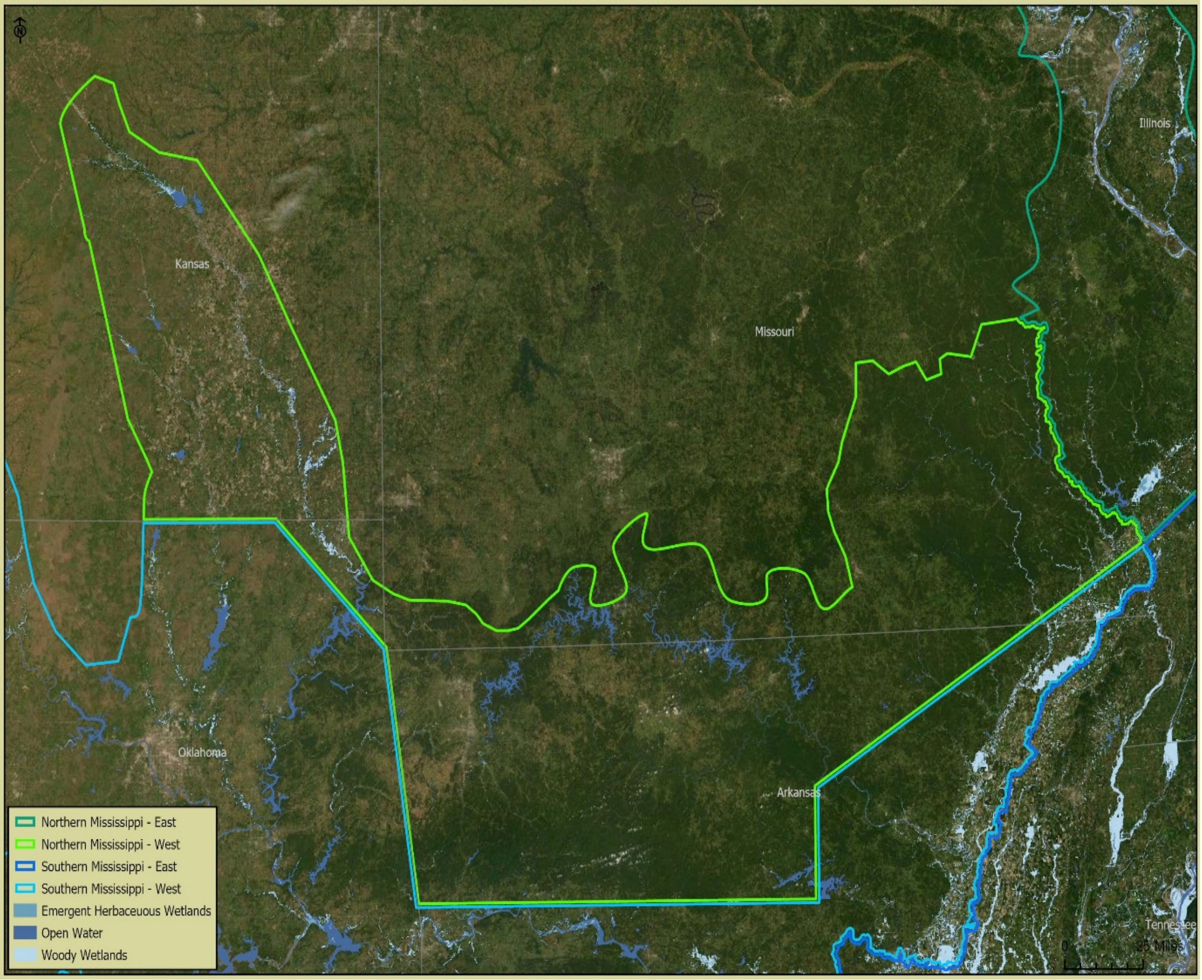

Figure A8. Suitable alligator snapping turtle habitat within the Northern Mississippi – West Unit.

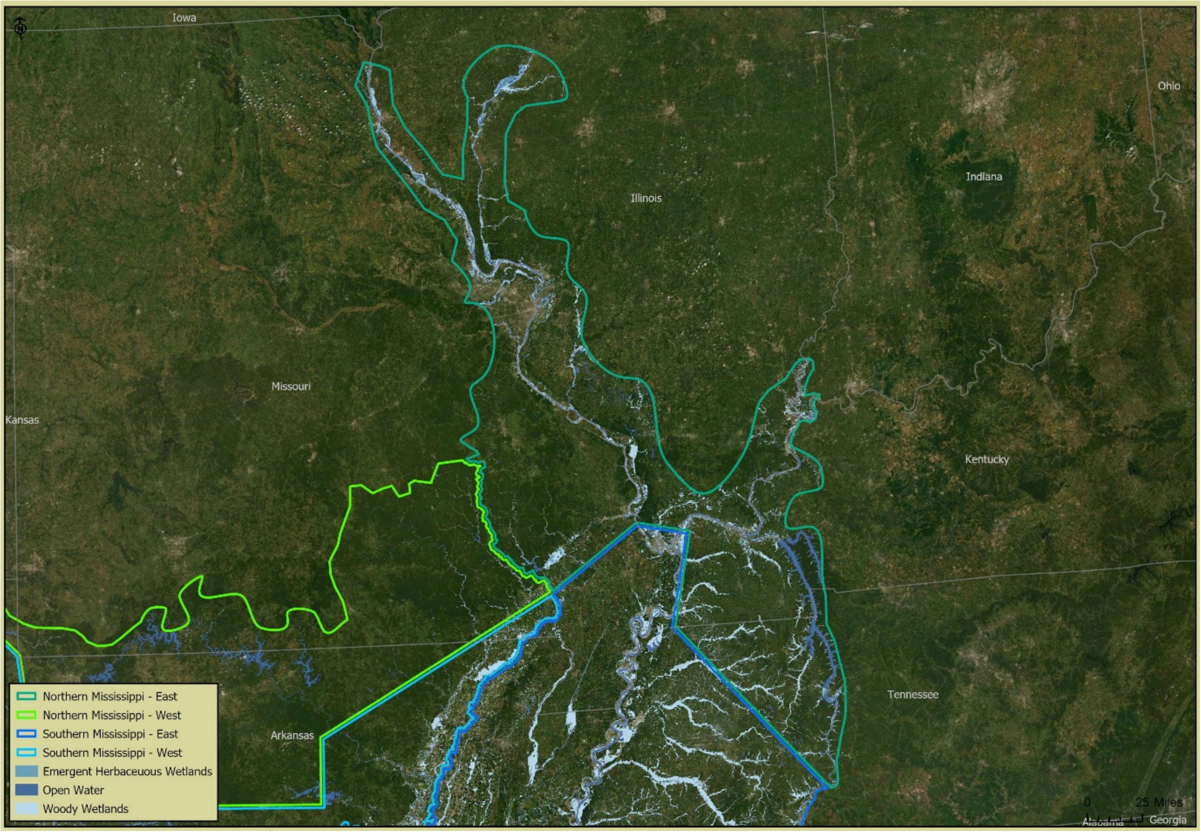

Figure A9. Suitable alligator snapping turtle habitat within the Northern Mississippi – East Unit.

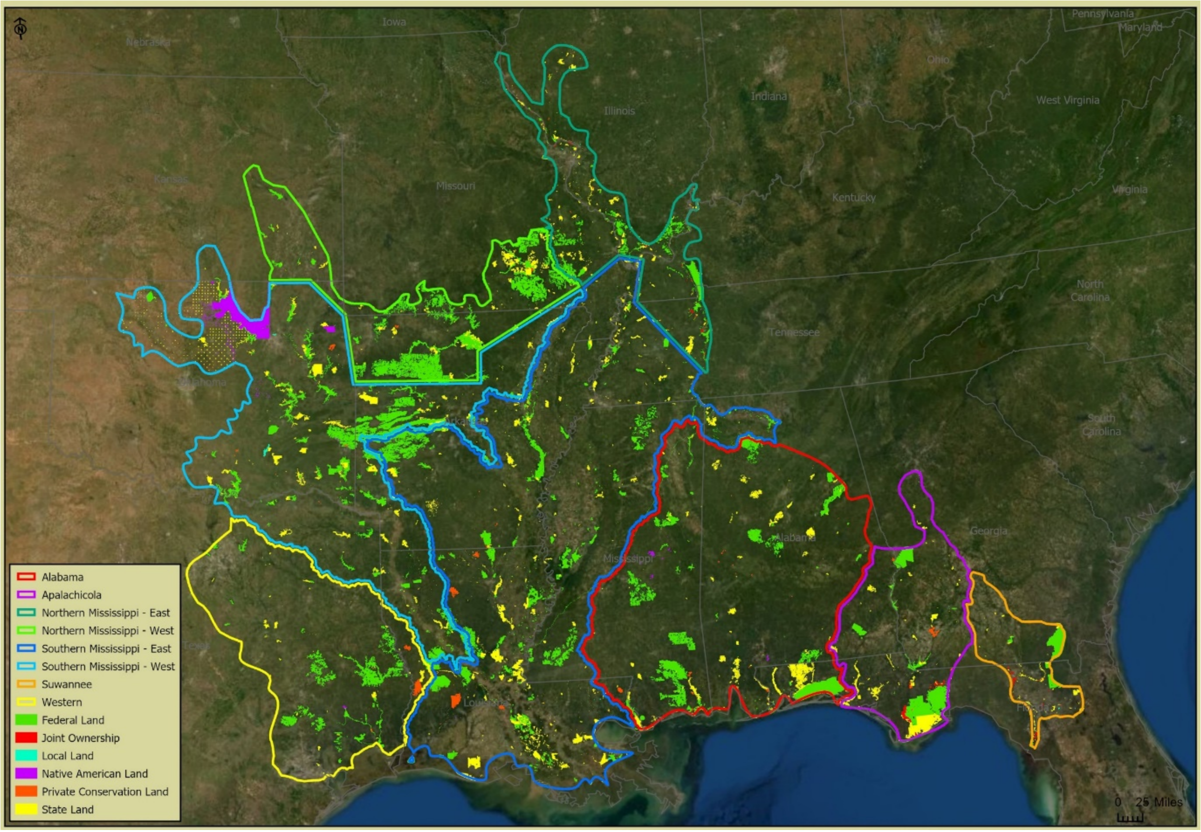

Figure A10. Lands in conservation within the range of the alligator snapping turtle.

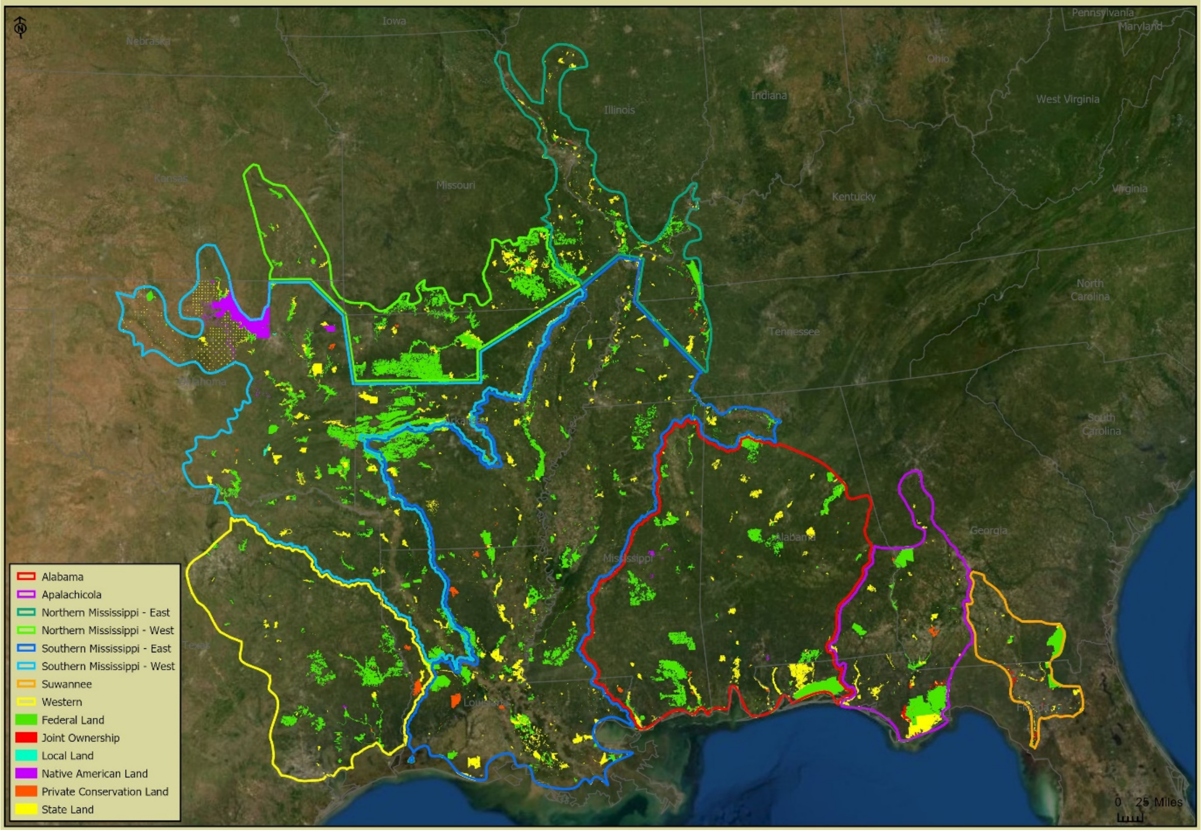

Figure A11. Suitable alligator snapping turtle habitat within conservation lands.

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%**

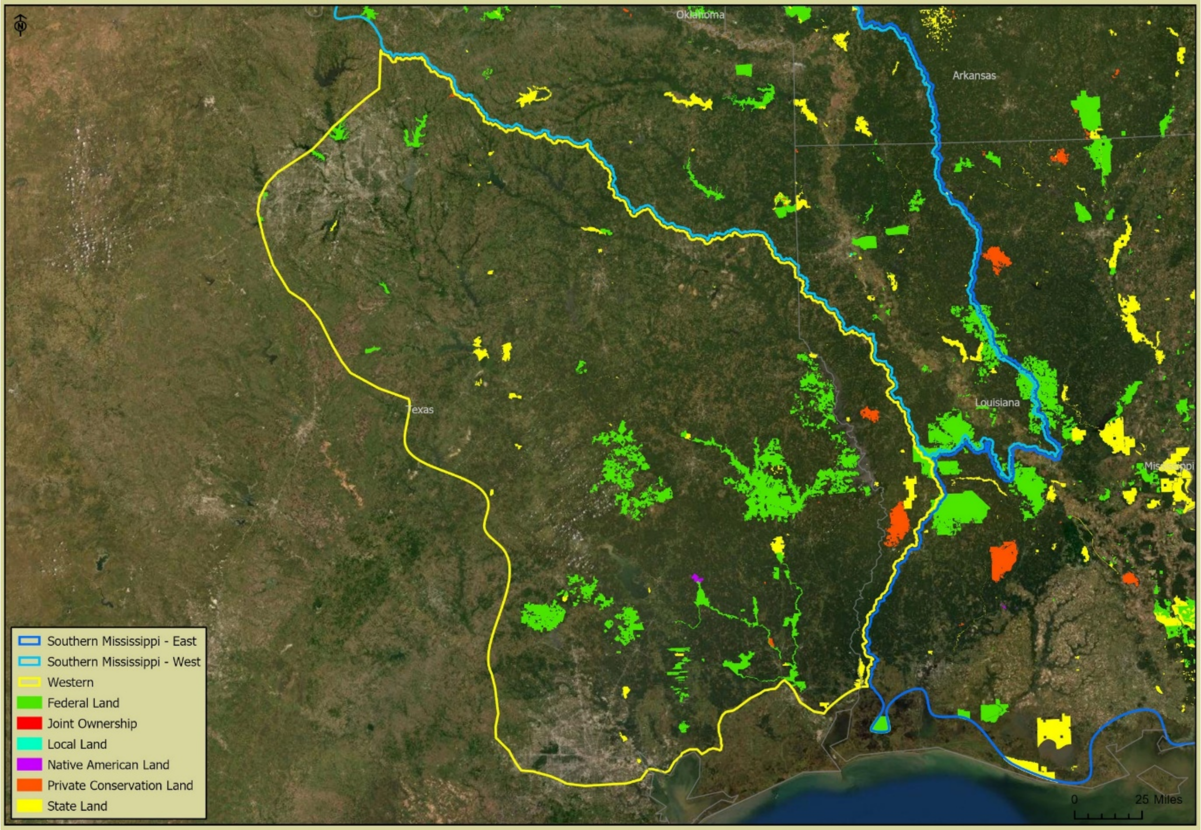

Figure A12. Lands in conservation within the range of the alligator snapping turtle within the Western Unit.

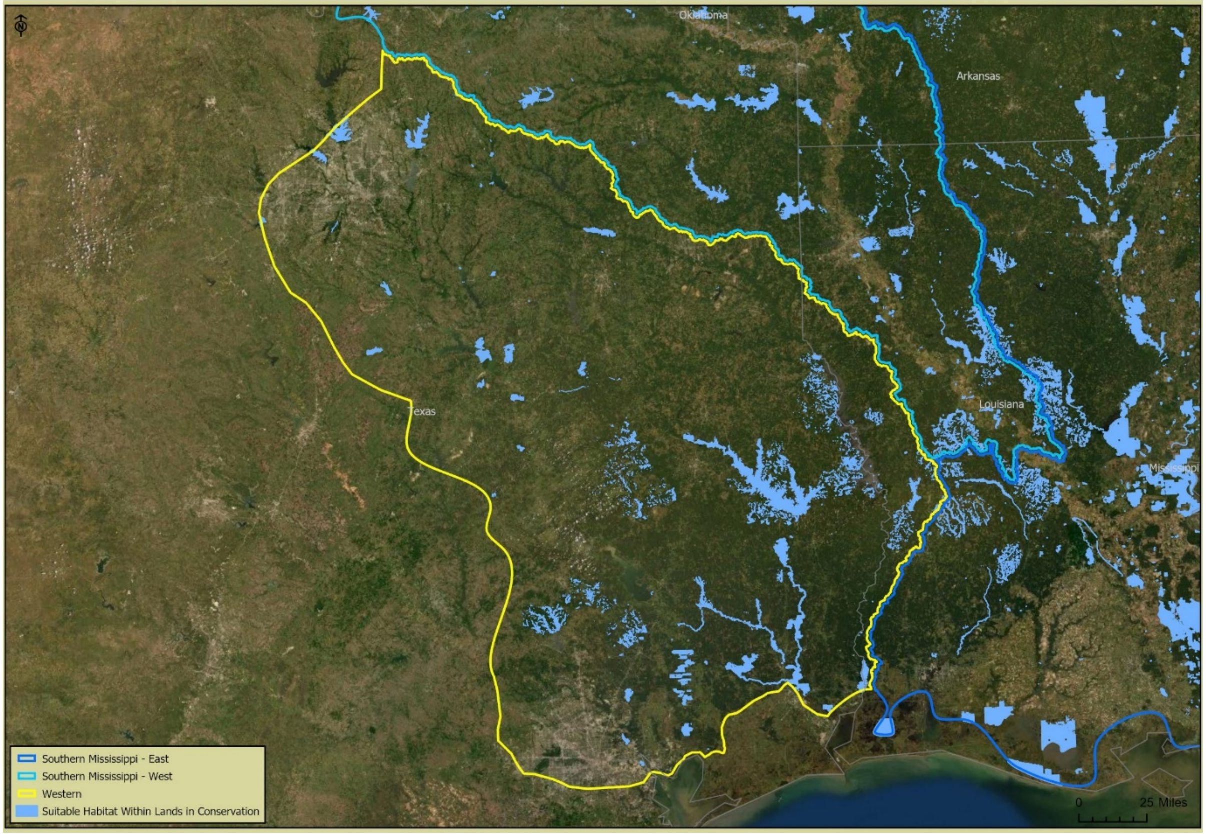

Figure A13. Suitable alligator snapping turtle habitat within conservation lands within the Western Unit.

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%**

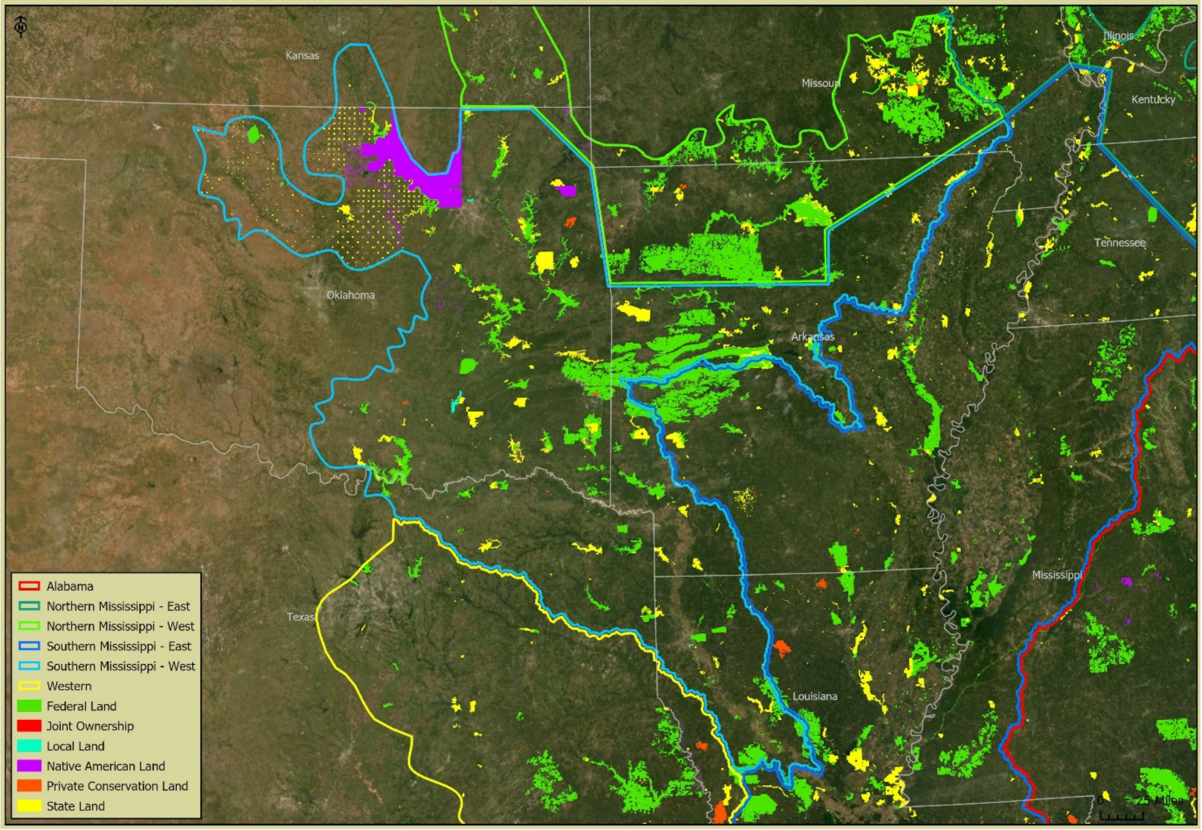

Figure A14. Lands in conservation within the range of the alligator snapping turtle in the Southern Mississippi – West Unit.

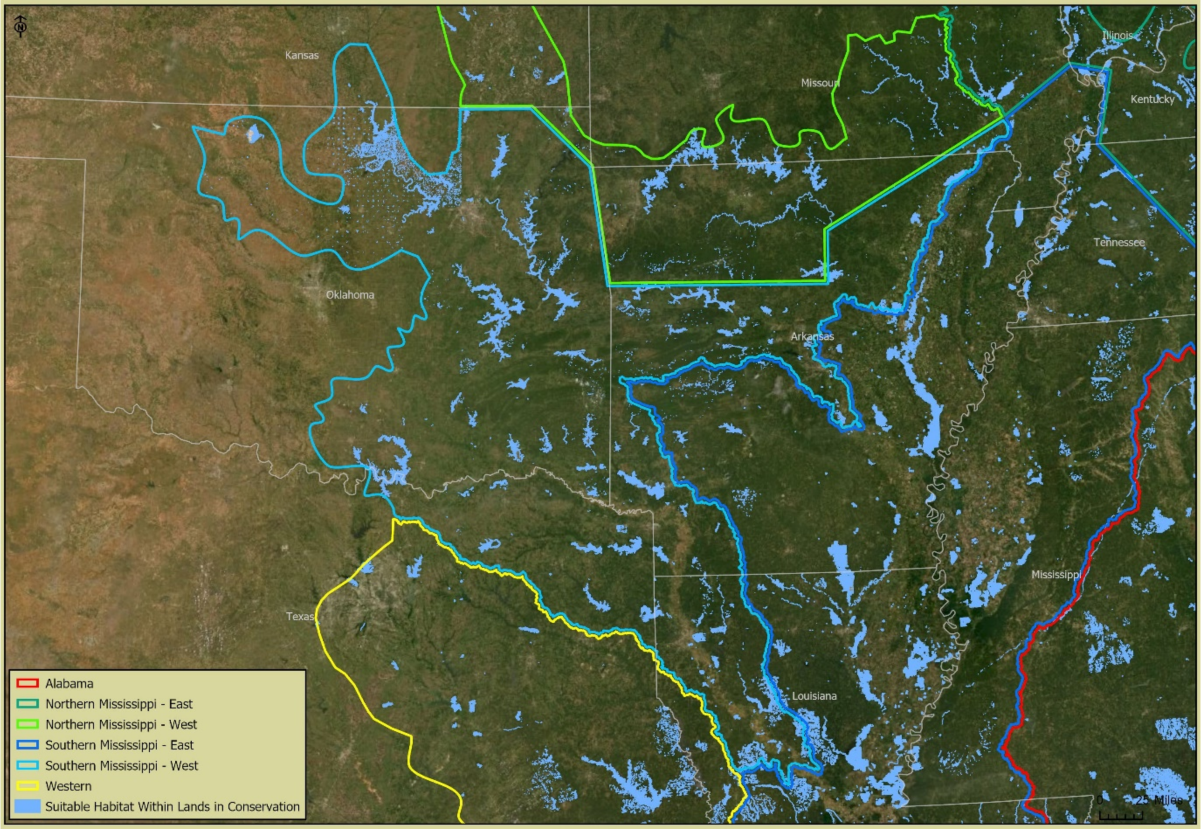

Figure A15. Suitable alligator snapping turtle habitat on conservation lands within the Southern Mississippi – West Unit.

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%**

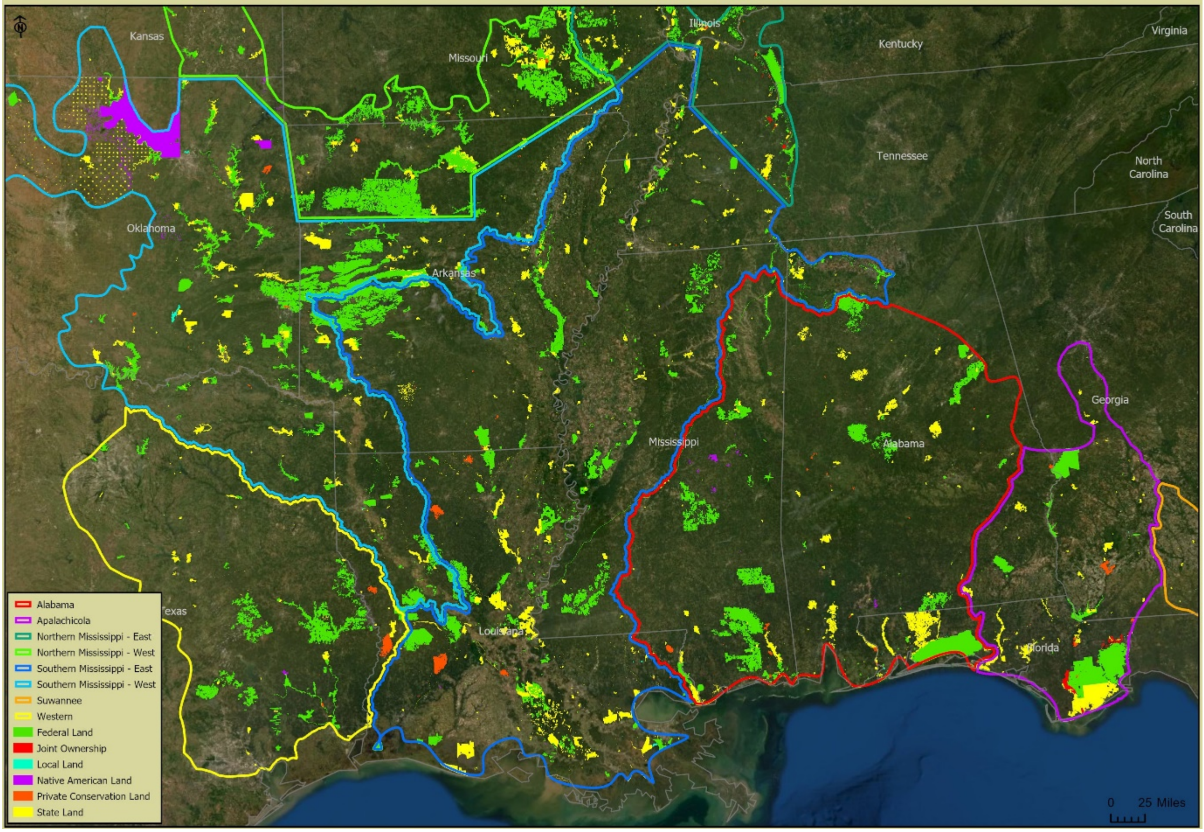

Figure A16. Lands in conservation within the range of the alligator snapping turtle in the Southern Mississippi – East Unit.

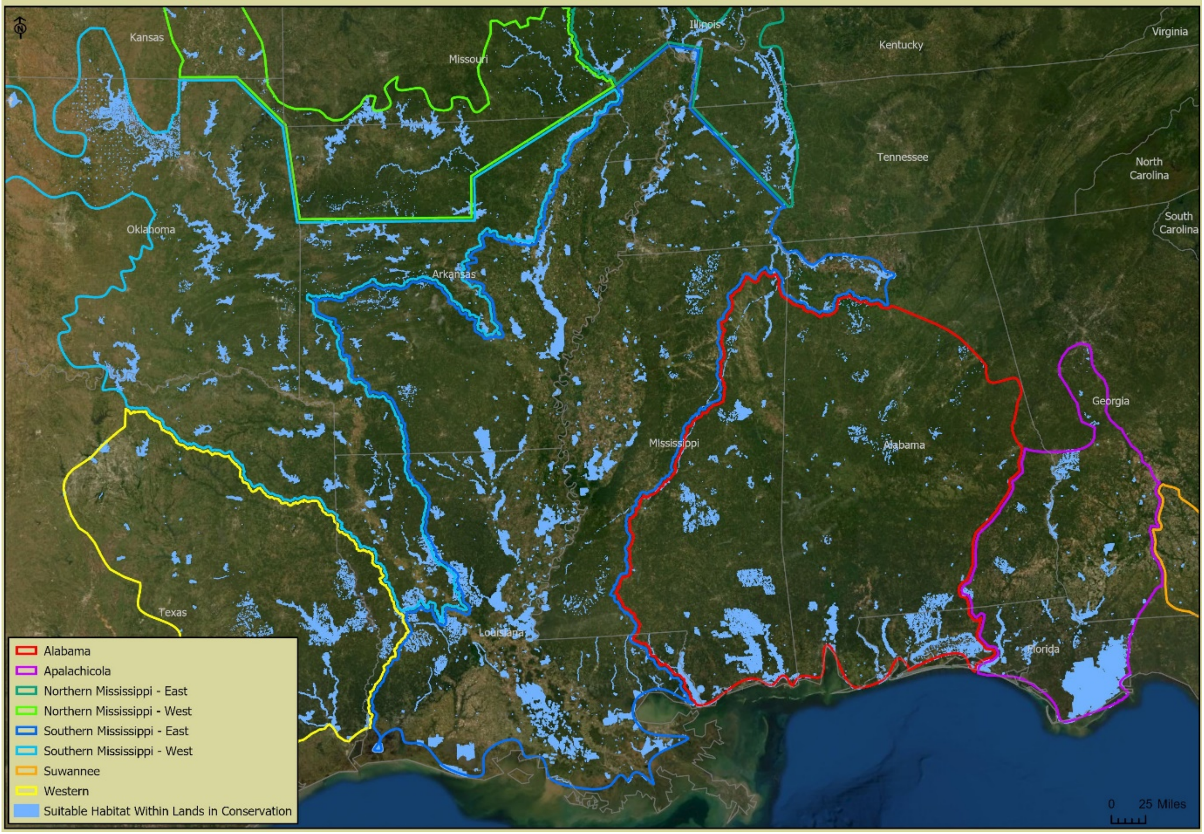

Figure A17. Suitable alligator snapping turtle habitat on conservation lands within the Southern Mississippi – East Unit.

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%**

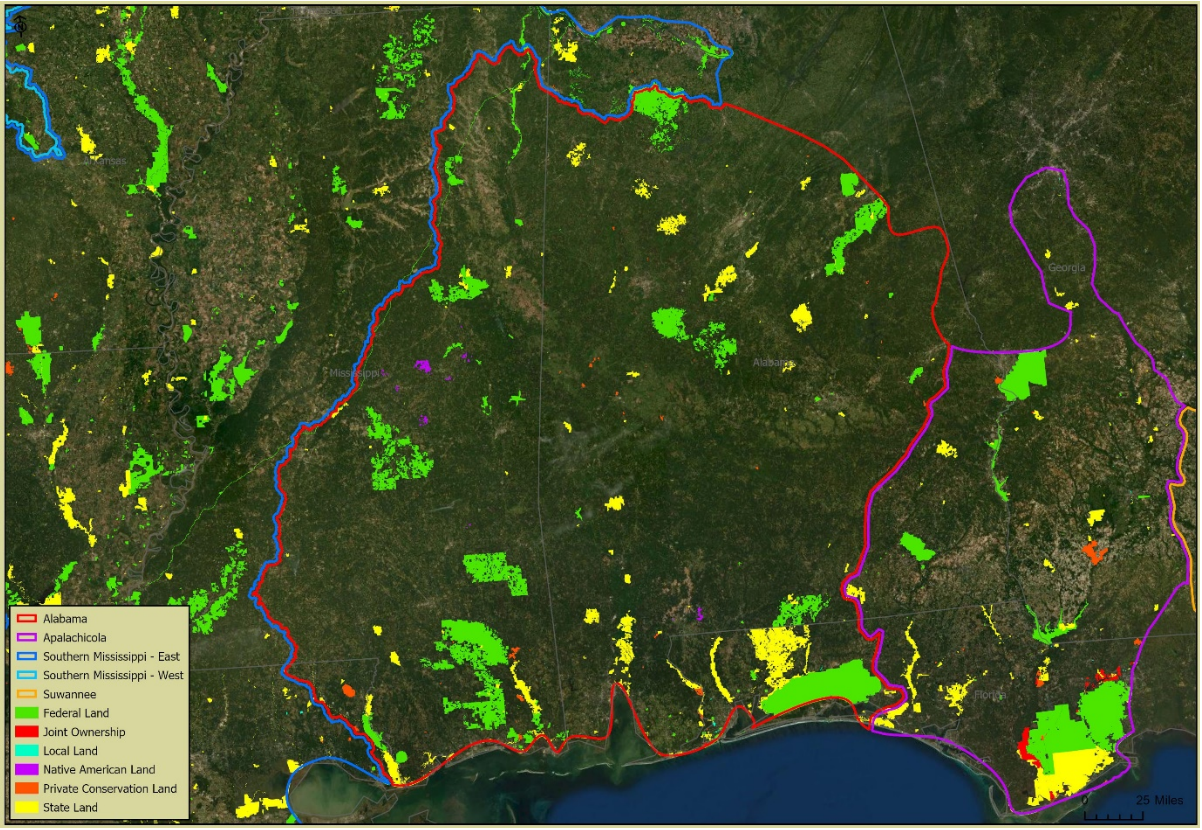

Figure A18. Lands in conservation within the range of the alligator snapping turtle in the Alabama Unit.

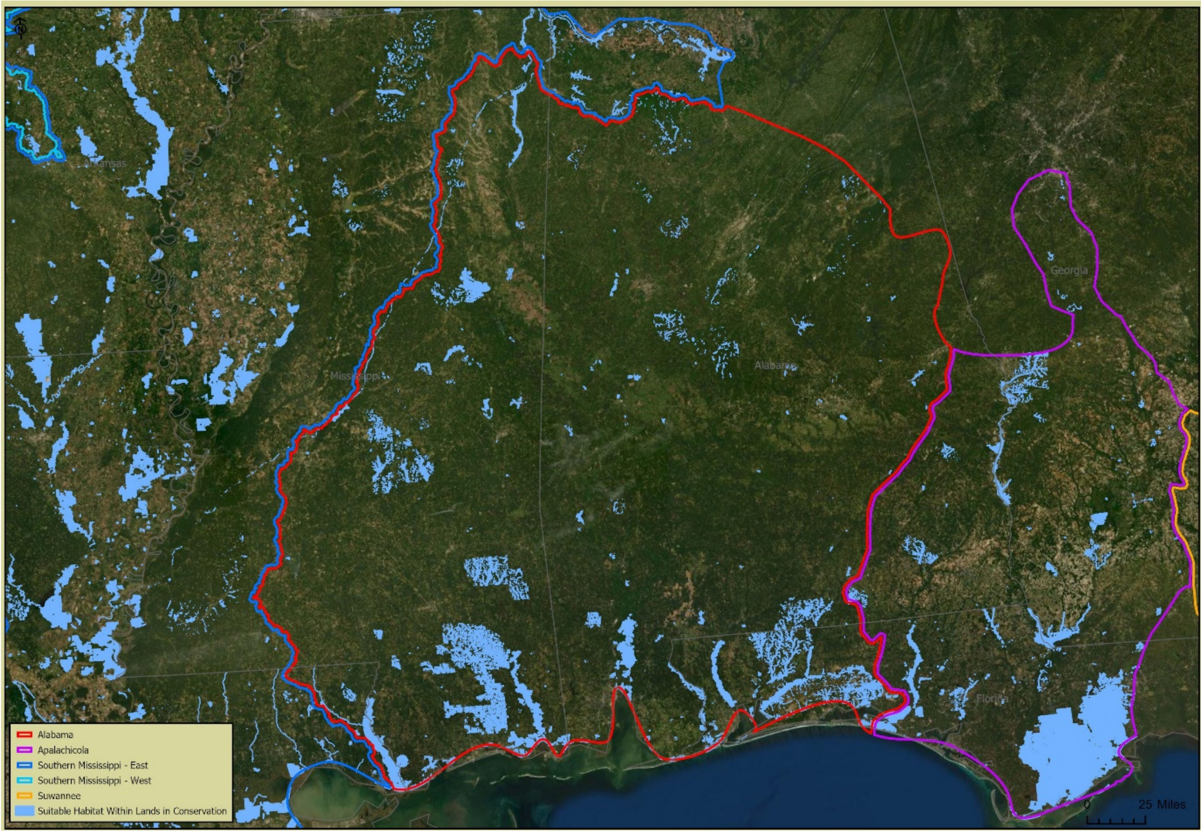

Figure A19. Suitable alligator snapping turtle habitat on conservation lands within the Alabama Unit.

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%**

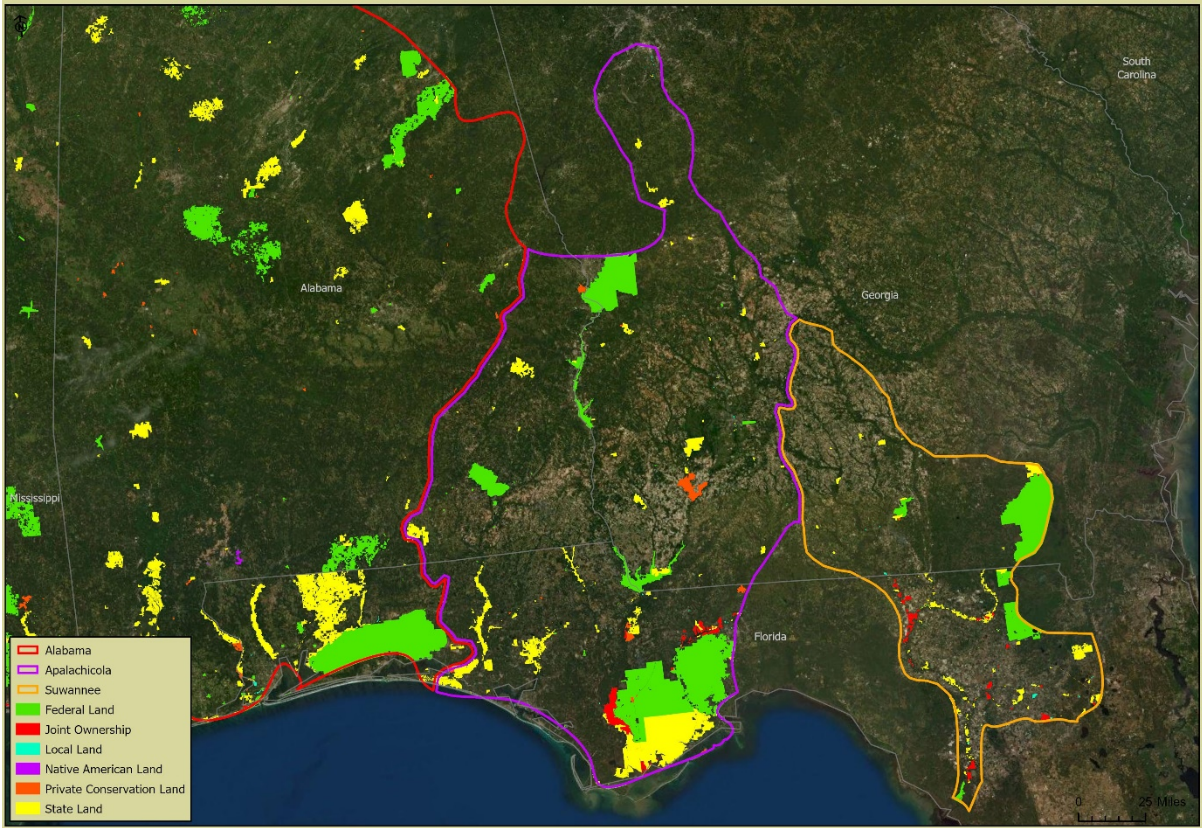

Figure A20. Lands in conservation within the range of the alligator snapping turtle in the Apalachicola Unit.

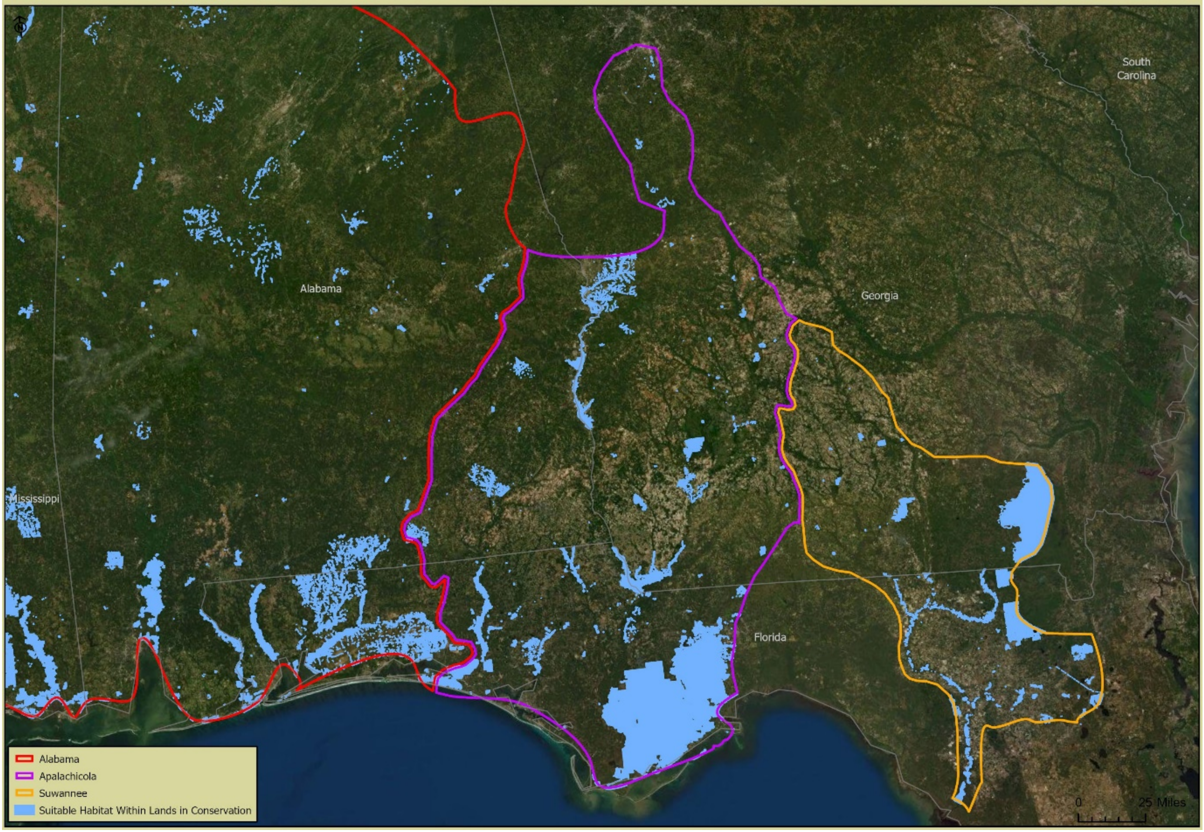

Figure A21. Suitable alligator snapping turtle habitat on conservation lands within the Apalachicola Unit.

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%**

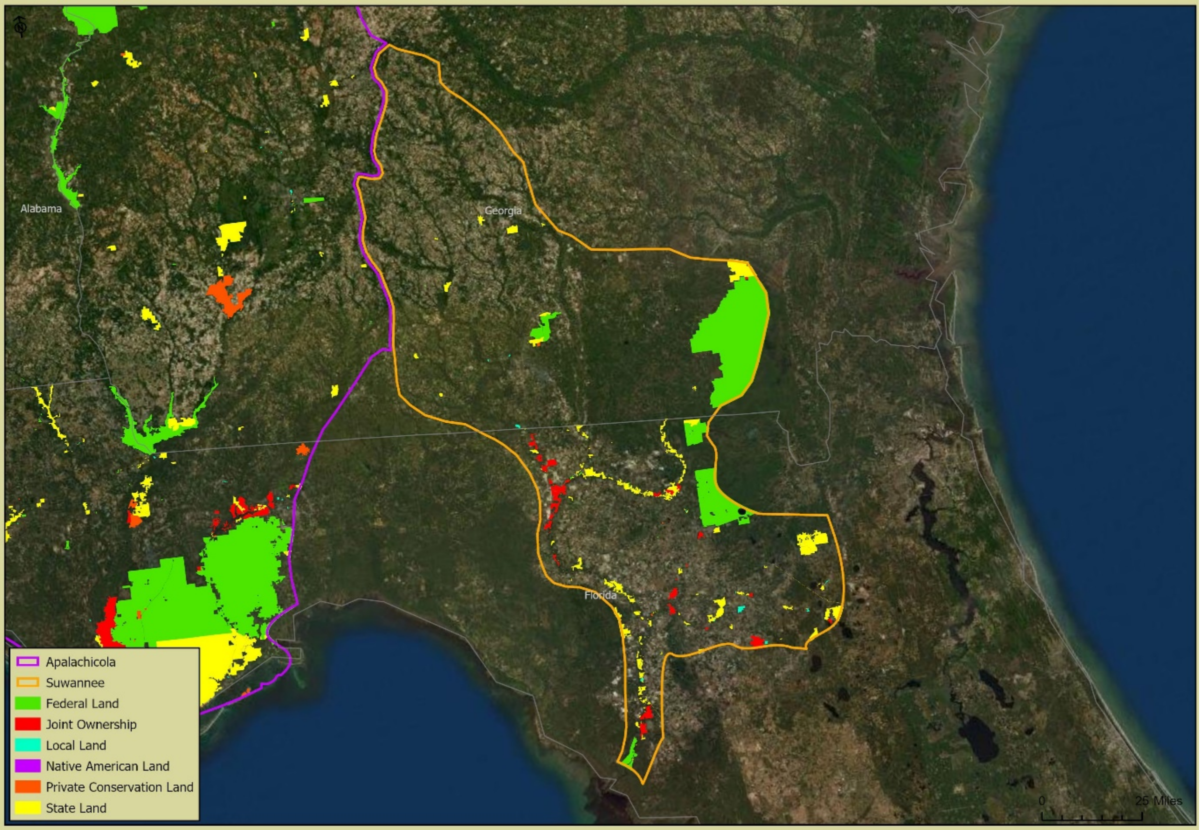

Figure A22. Lands in conservation within the range of the alligator snapping turtle in the Suwannee Unit.

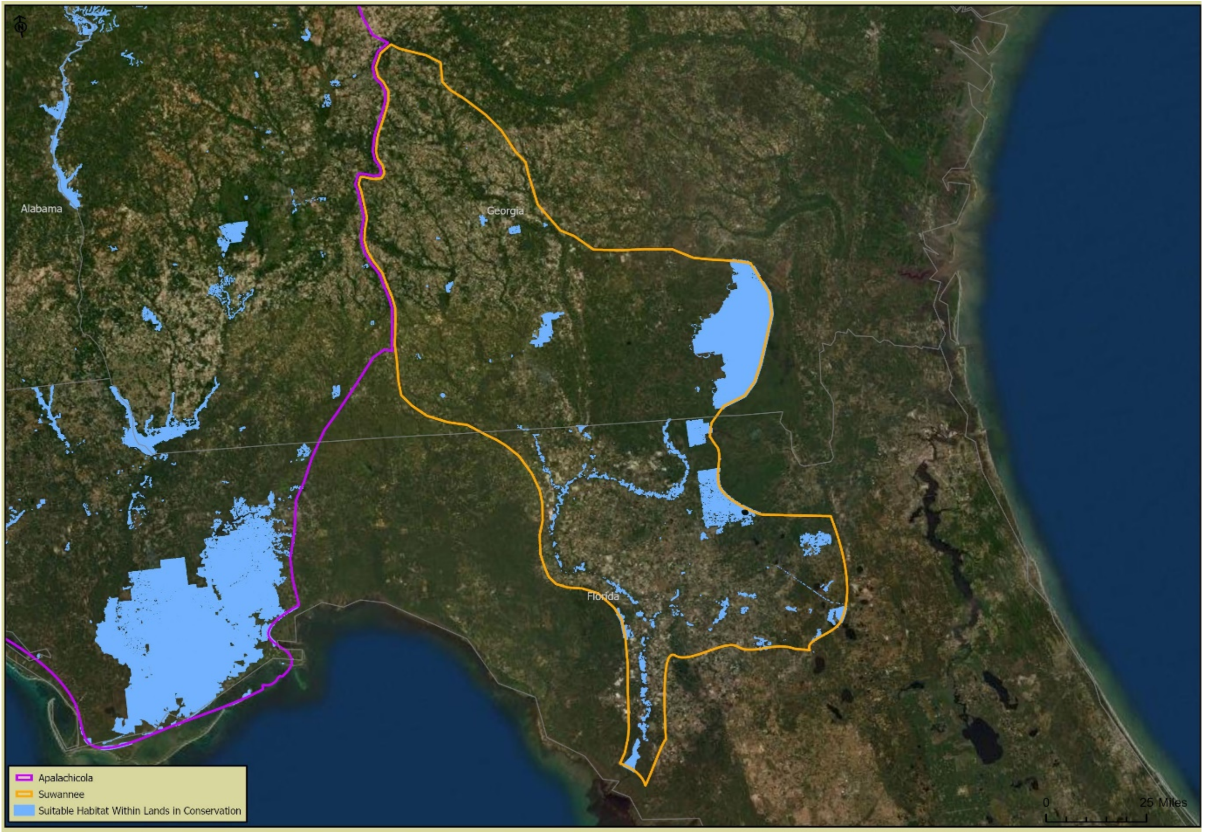

Figure A23. Suitable alligator snapping turtle habitat on conservation lands within the Suwannee Unit.

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%**

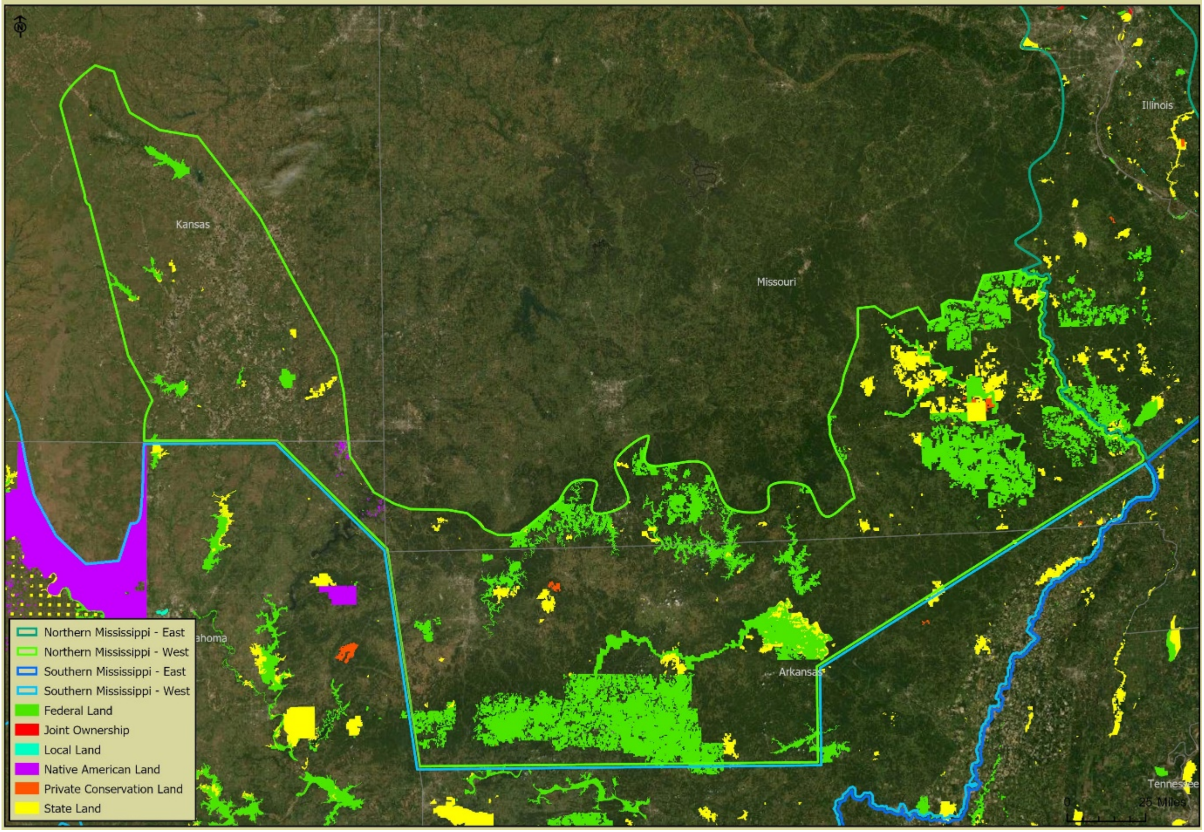

Figure A24. Lands in conservation within the range of the alligator snapping turtle in the Northern Mississippi – West Unit.

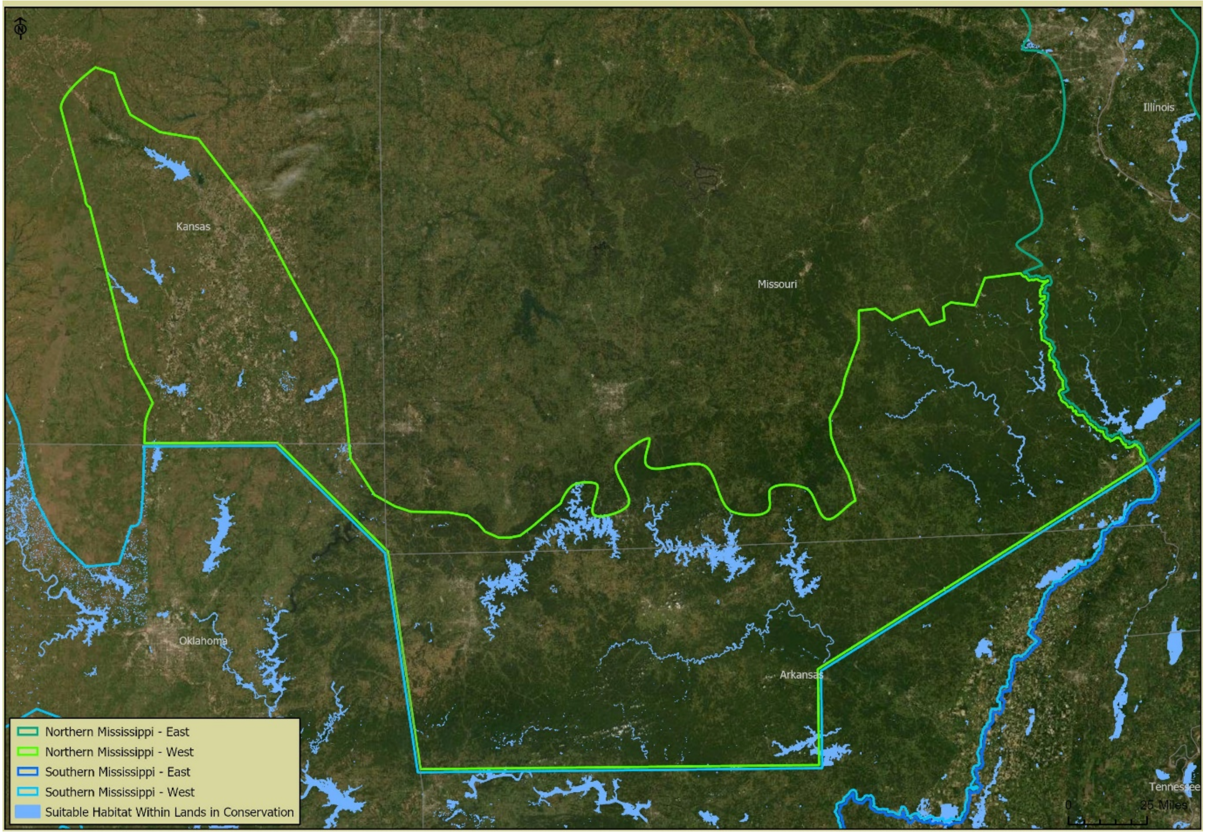

Figure A25. Suitable alligator snapping turtle habitat on conservation lands within the Northern Mississippi – West Unit.

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%**

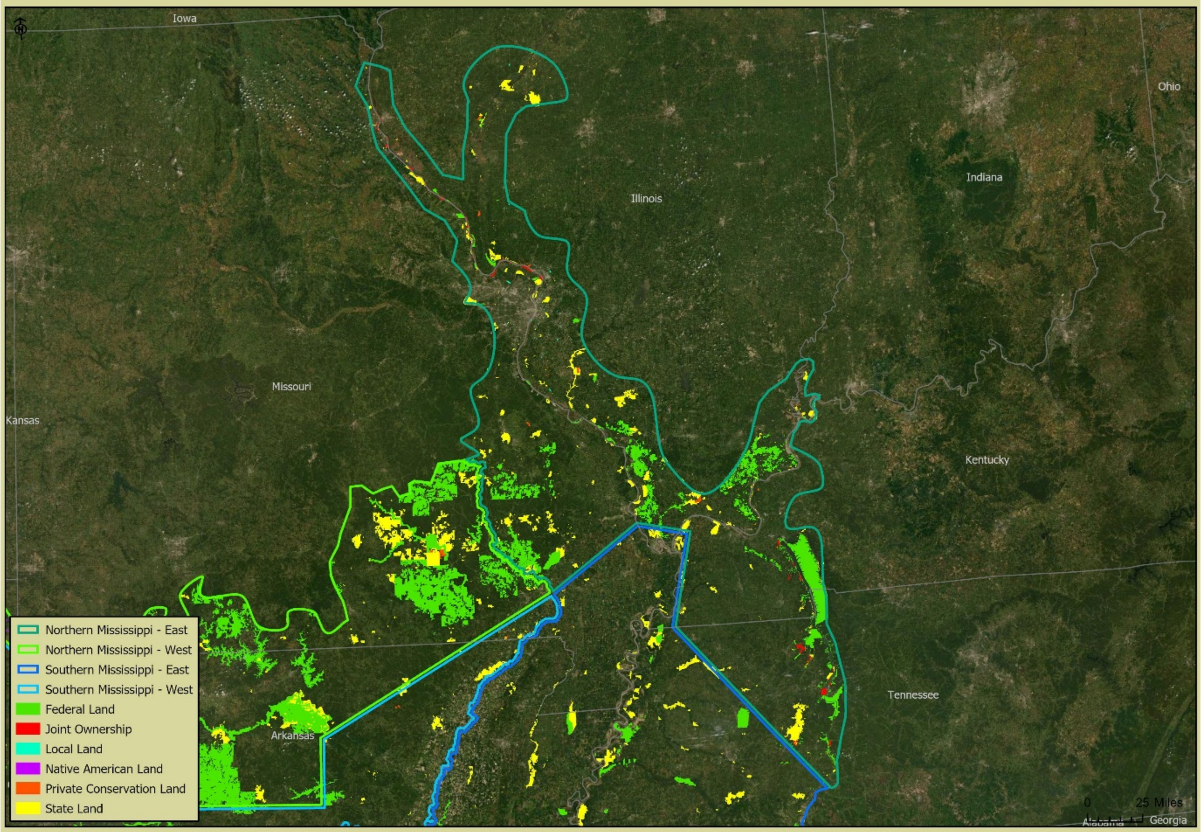

Figure A26. Lands in conservation within the range of the alligator snapping turtle in the Northern Mississippi – East Unit.

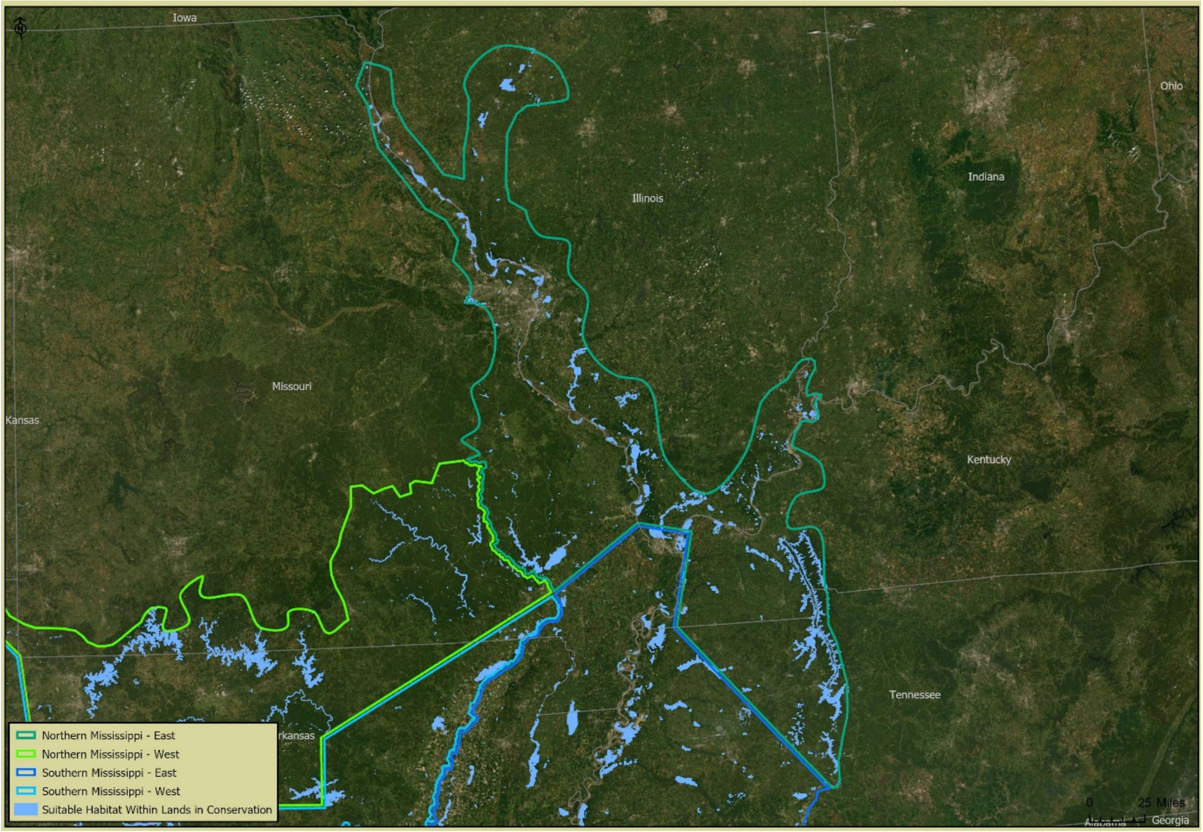

Figure A27. Suitable alligator snapping turtle habitat on conservation lands within the Northern Mississippi – East Unit.

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). Please record your responses in the attached excel sheet, not in this word document.

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 ($\mu.\text{p}_i$, CS_i) and process standard deviation ($\sigma_{\text{P}i}$) were drawn from the parameter's overall mean and variance distributions:

$$\mu.\text{p}_i \sim \text{beta}(\alpha, \beta)$$

$$CS_i \sim \text{lognormal}(x_1, x_2)$$

$$\sigma_{\text{P}i} \sim \text{normal}(\sigma_\text{P}, \sigma_\text{P} \times 0.05)$$

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 ($CV_\text{P} = 0.002$). Lastly, the iteration-specific means ($\mu.\text{p}_i$, CS_i) and standard deviations ($\sigma_{\text{P}i}$) were then used to create iteration-specific distributions from which baseline parameter values were then drawn for each timestep *t* within iteration *i*:

$$\mu.\text{p}_{i,t} \sim \text{beta}(\alpha_i, \beta_i)$$

$$CS_{i,t} \sim \text{lognormal}(x_{1i}, x_{2i})$$

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 $\theta_{\mathrm{SNP},p}$ and $\omega_{\mathrm{SNP},u}$. values were used for all scenarios. We chose to hold the SNP spatial extent ($\omega_{\mathrm{SNP},u}$) 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:

$$\theta_{p,a,s,i,t} \sim \mathrm{beta}(\alpha_{a,p,s}, \beta_{a,p,s})$$

$$\omega_{a,u,s,i,t} \sim \mathrm{beta}(\alpha_{a,u,s}, \beta_{a,u,s})$$

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 ($\varphi_{\mathrm{N,Alabama},c}$): (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 ($h_{\mathrm{J,i,t}}$) and adults ($h_{\mathrm{A,i,t}}$) were drawn from Poisson distributions and placed in the **hsd** vector:

$$h_{\mathrm{J}} \sim \text{Poisson}(30)$$

$$h_{\mathrm{J}} \sim \text{Poisson}(12)$$

$$\boldsymbol{hsd}_{u,s,i,t} = \begin{bmatrix} h_{J,u,s,i,t} \\ h_{A,u,s,i,t} \end{bmatrix}$$

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 ($\mathbf{D}_{\mathrm{SOME,\ ALAB,\ APAL}}$ and $\mathbf{D}_{\mathrm{NOME}}$) 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.

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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 ($PJ = \varphi J \times (1 - \gamma J)$); (2) juvenile growth, the probability of surviving as a juvenile and transitioning to the adult stage ($GJ = \varphi J \times \gamma J$); (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 ($\mathbf{D}_{\text{SOME, ALAB, APAL}}$) and Northern Mississippi – East ($\mathbf{D}_{\text{NOME}}$).

Southern Mississippi – East

Scenario	P_J	G_J	P_A	F_A
DETH	0.811 ± 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
$\mathbf{D}_{\mathrm{SOME,ALAB,APAL}}$	0.843	0.017	0.950	0.229
$\mathbf{D}_{\mathrm{NOME}}$	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 ($\mathbf{D}_{\mathrm{SOME,ALAB,APAL}}$) and Northern Mississippi – East ($\mathbf{D}_{\mathrm{NOME}}$).

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
$D_{\mathrm{SOME,\ ALAB,\ APAL}}$	0.1510	0.0244	0.8002	0.0244
D_{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

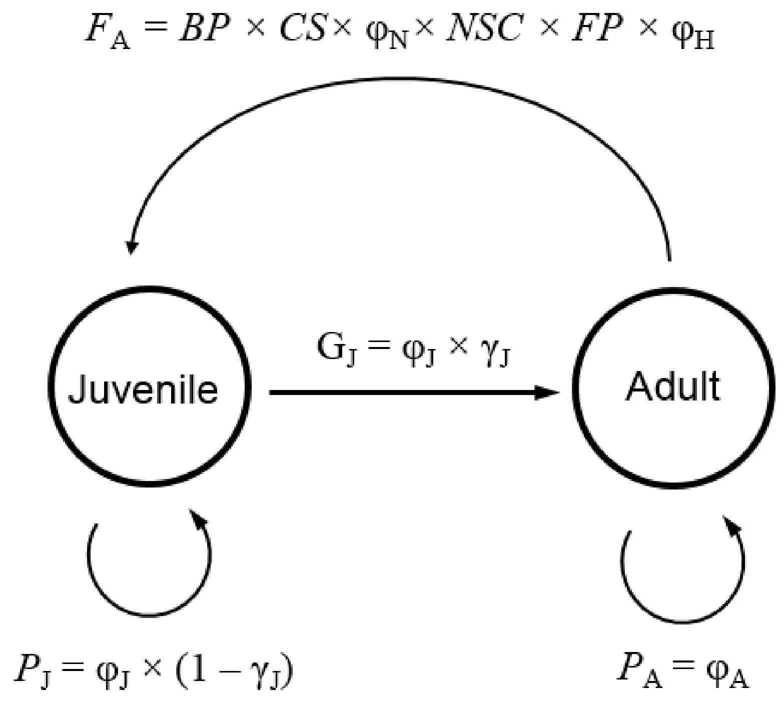

Figure E1. Alligator snapping turtle (*Macrochelys temminckii*) life cycle diagram for a female only two-stage prebreeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals), denoted by the J and A subscripts, respectively. 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.

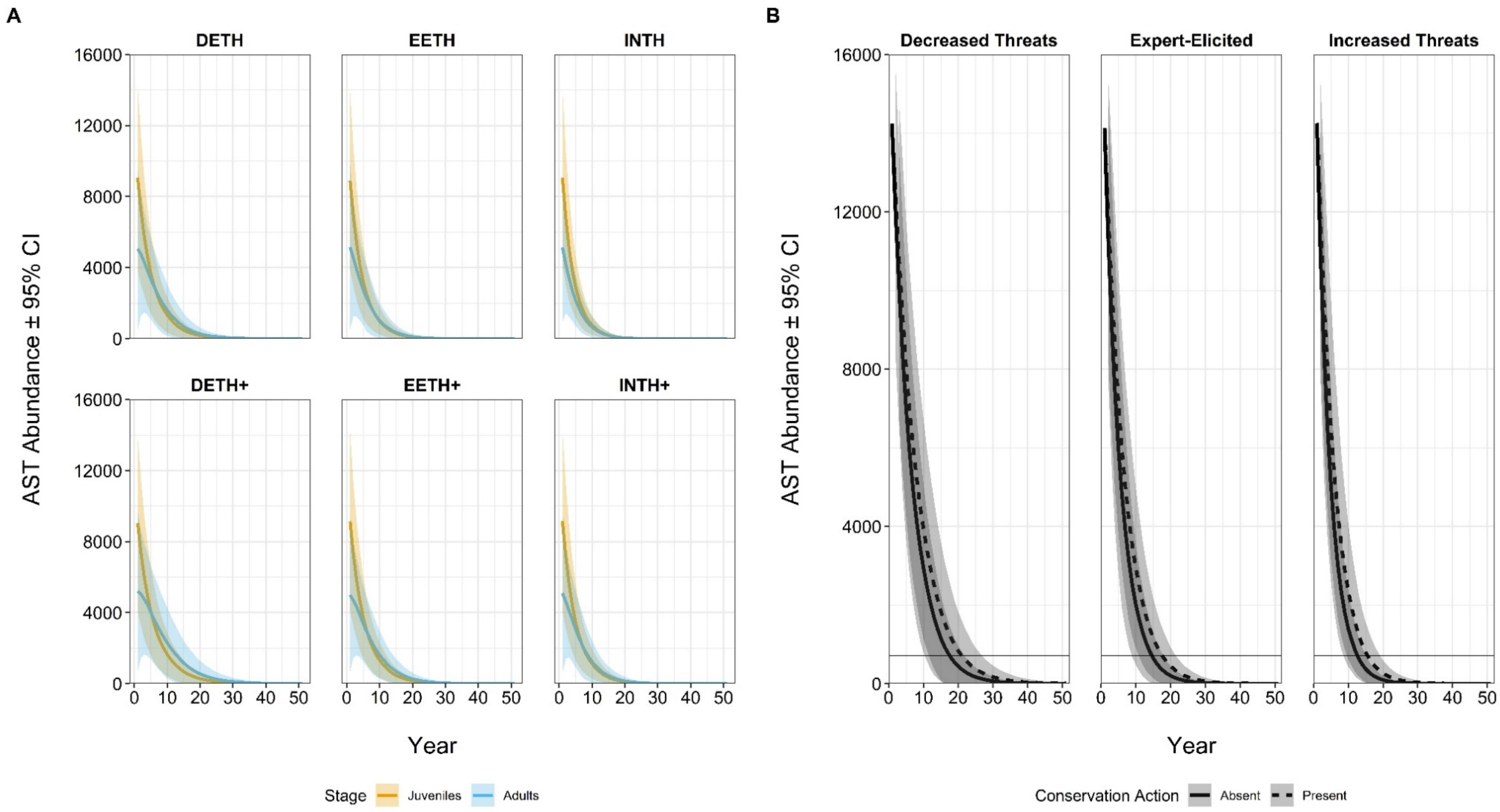

Figure E2. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Southern Mississippi – East analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations 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.

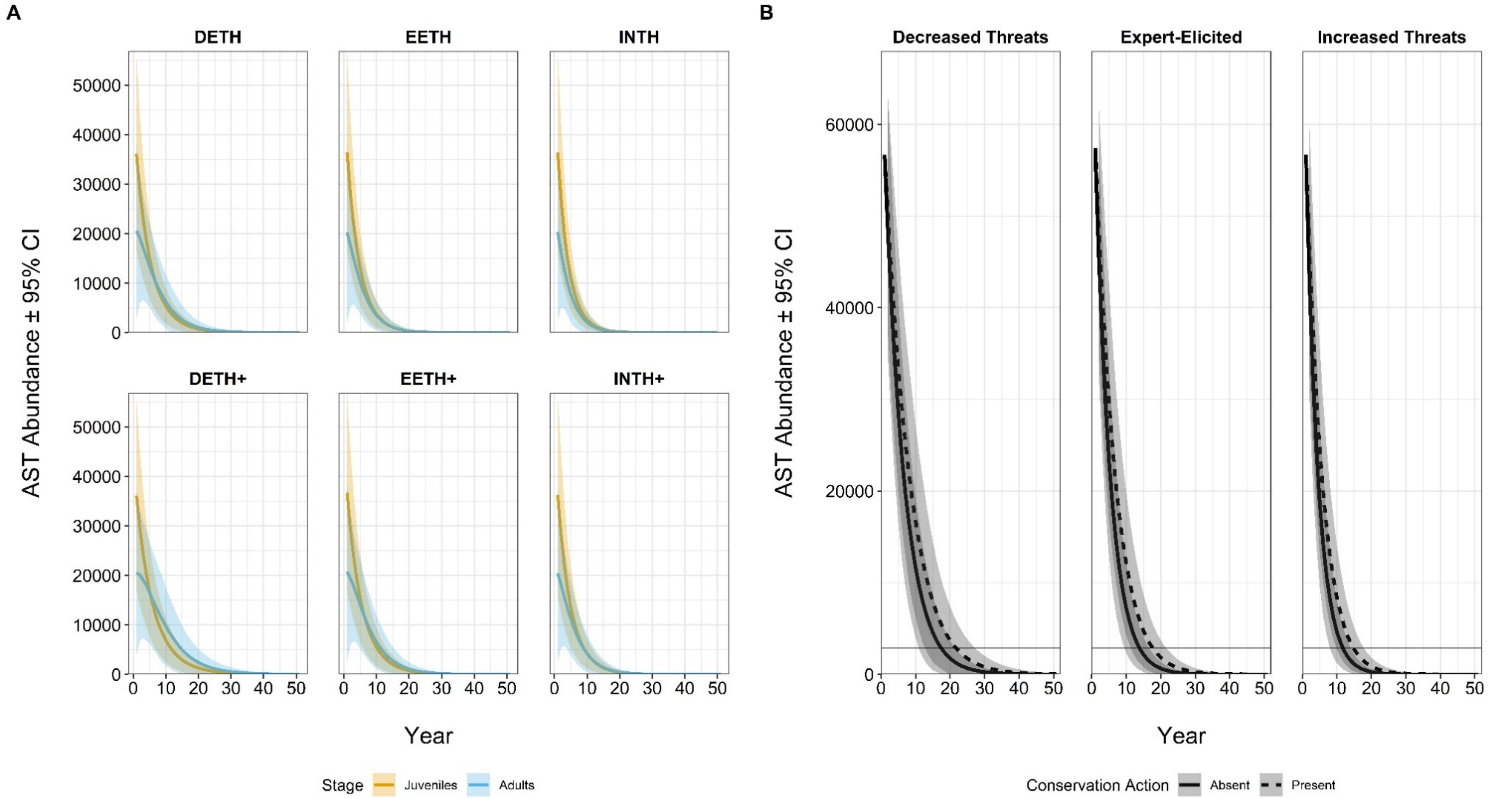

Figure E3. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Alabama analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the shaded 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.

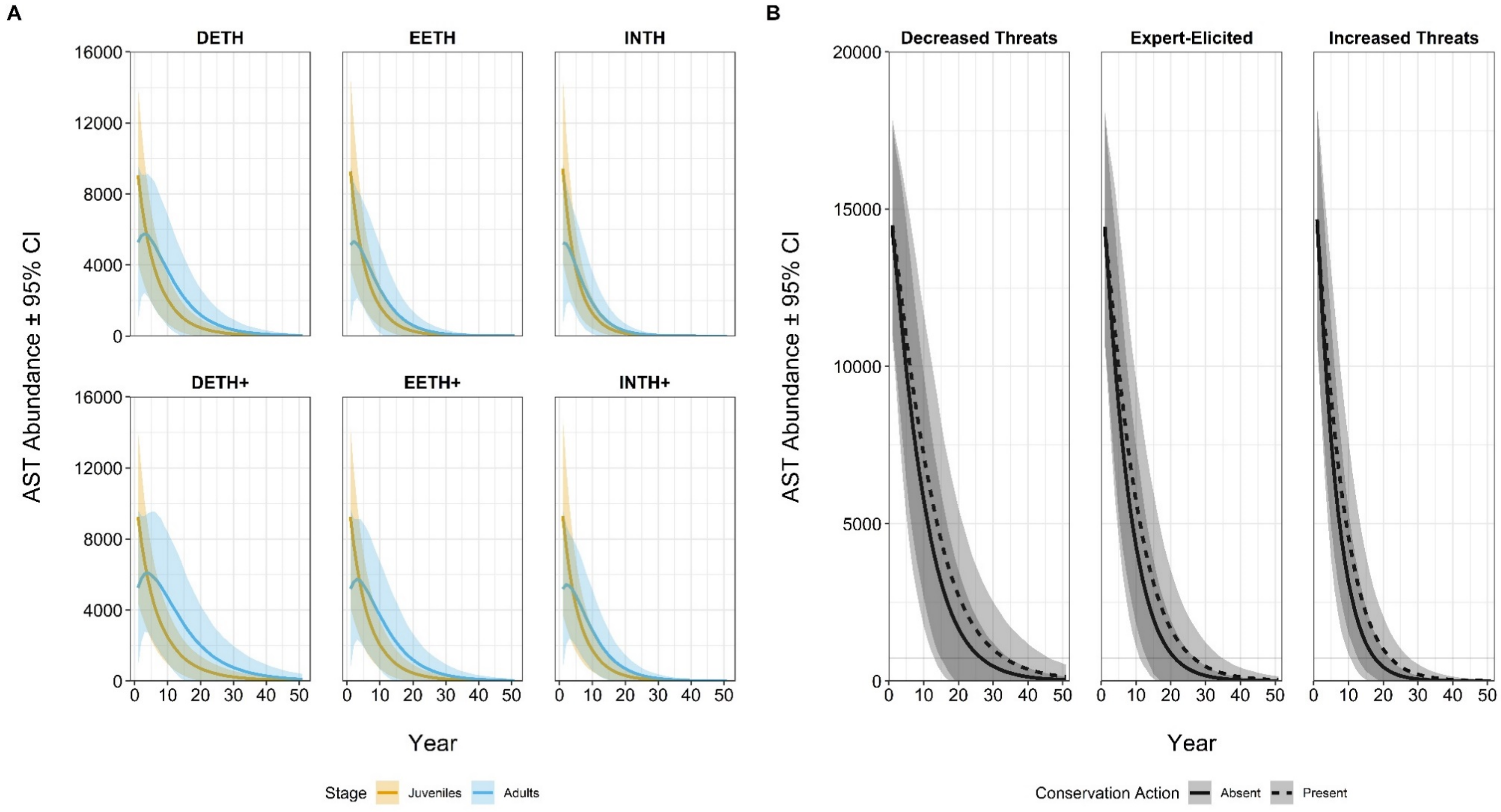

Figure E4. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Apalachicola analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations and the 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.

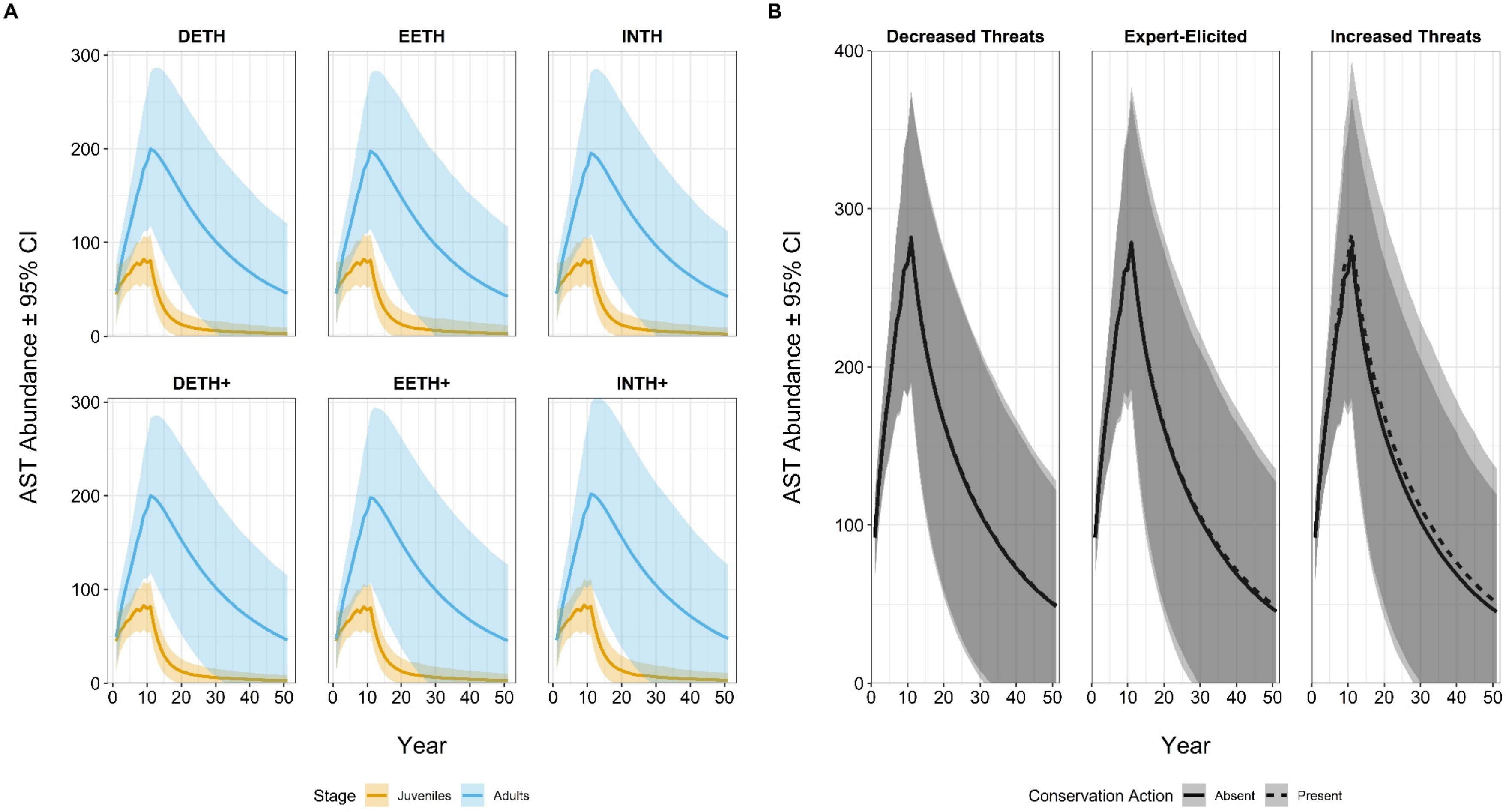

Figure E5. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) mean stage class-specific (A) and total (B) abundance (females only) over a 50-timesteps within the Northern Mississippi – East analysis unit. The curved lines depict the mean abundance trajectory across 500 stochastic iterations 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.

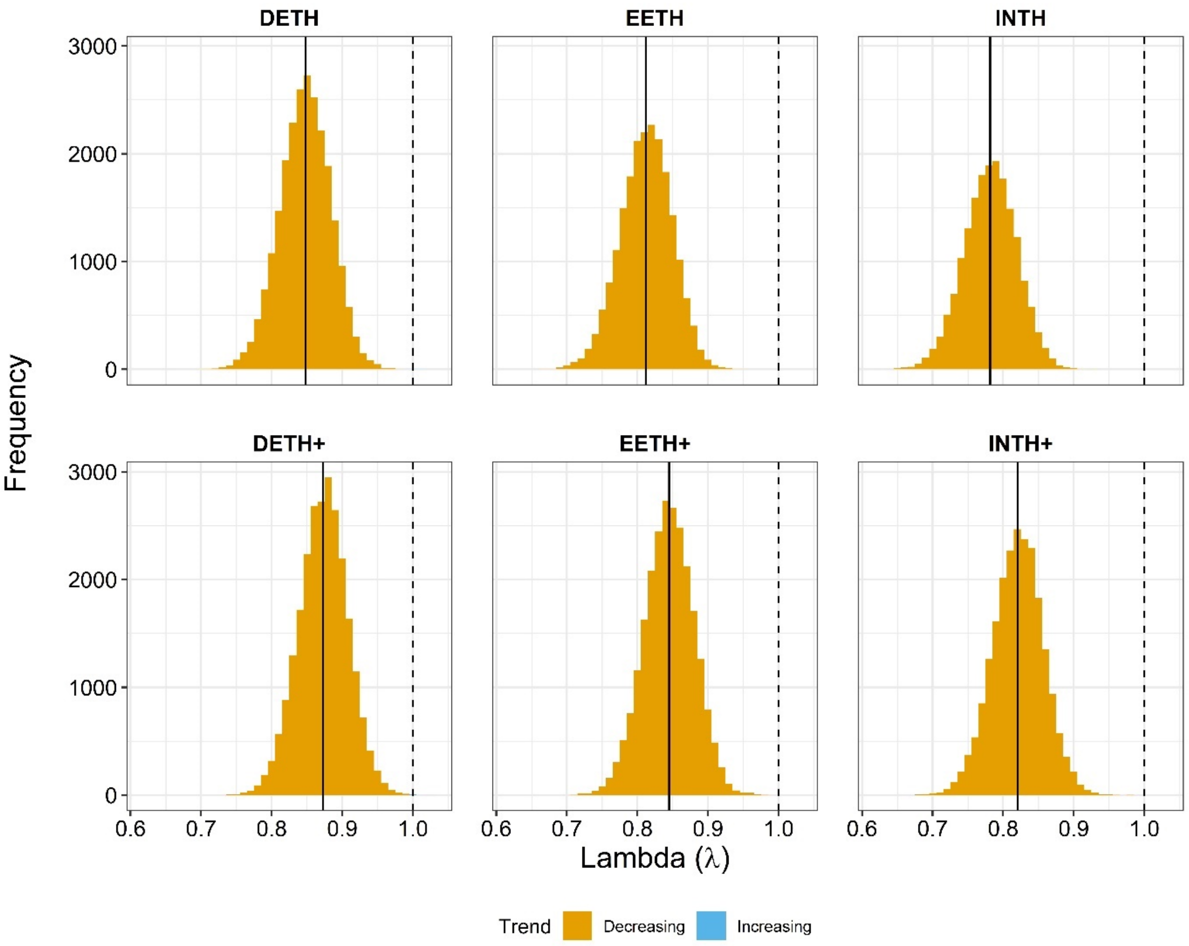

Figure E6. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Southern Mississippi – East analysis unit. Each panel 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).

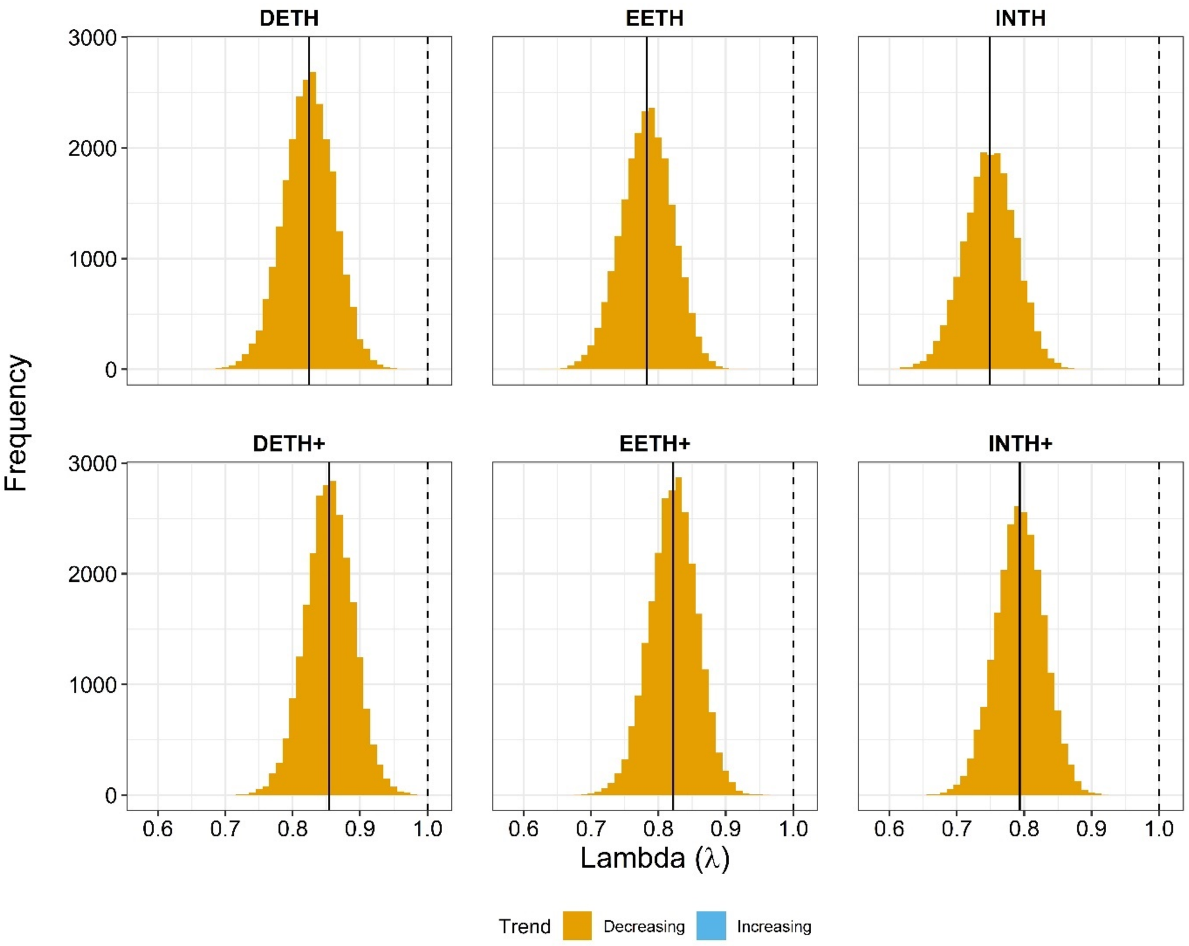

Figure E7. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Alabama analysis unit. Each panel represents a different 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).

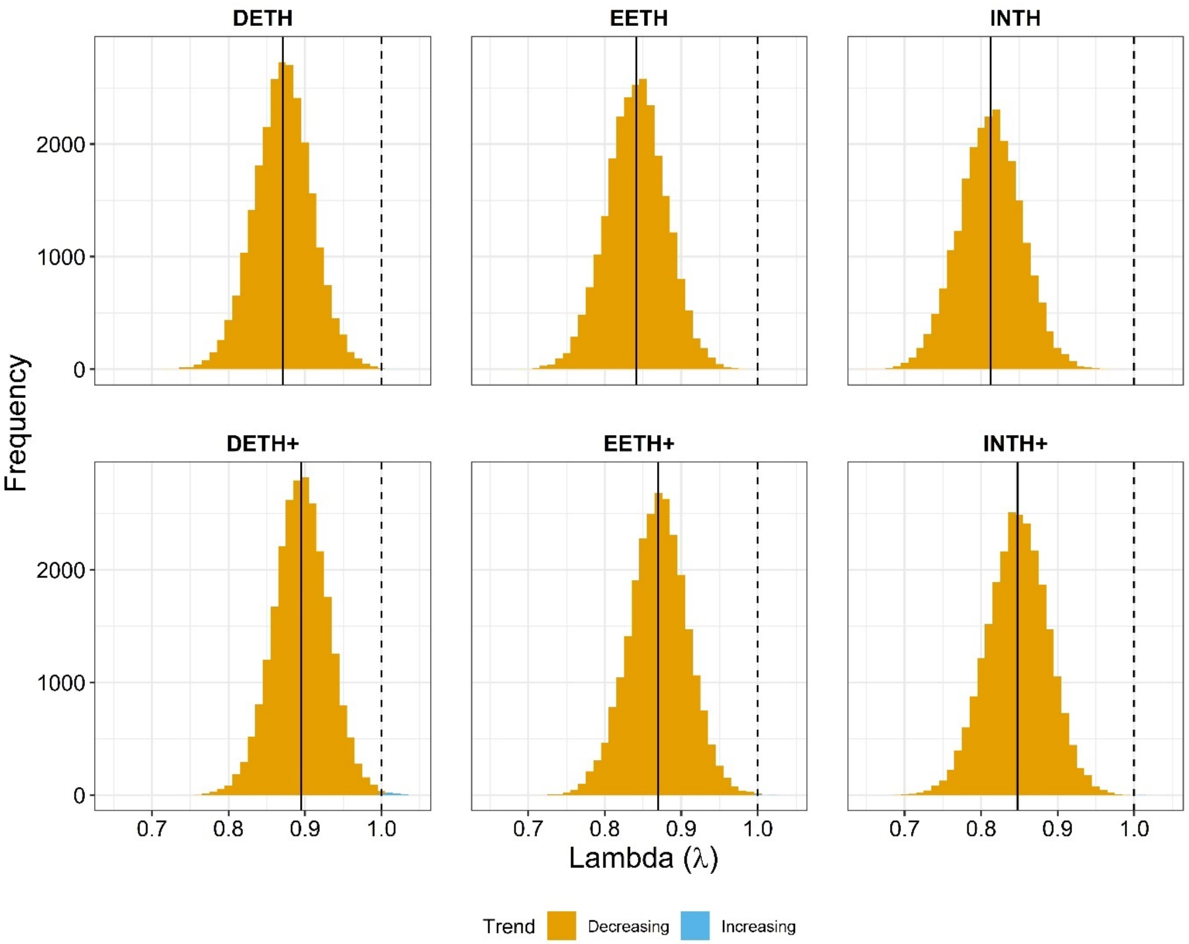

Figure E8. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Apalachicola analysis unit. Each panel represents a 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).

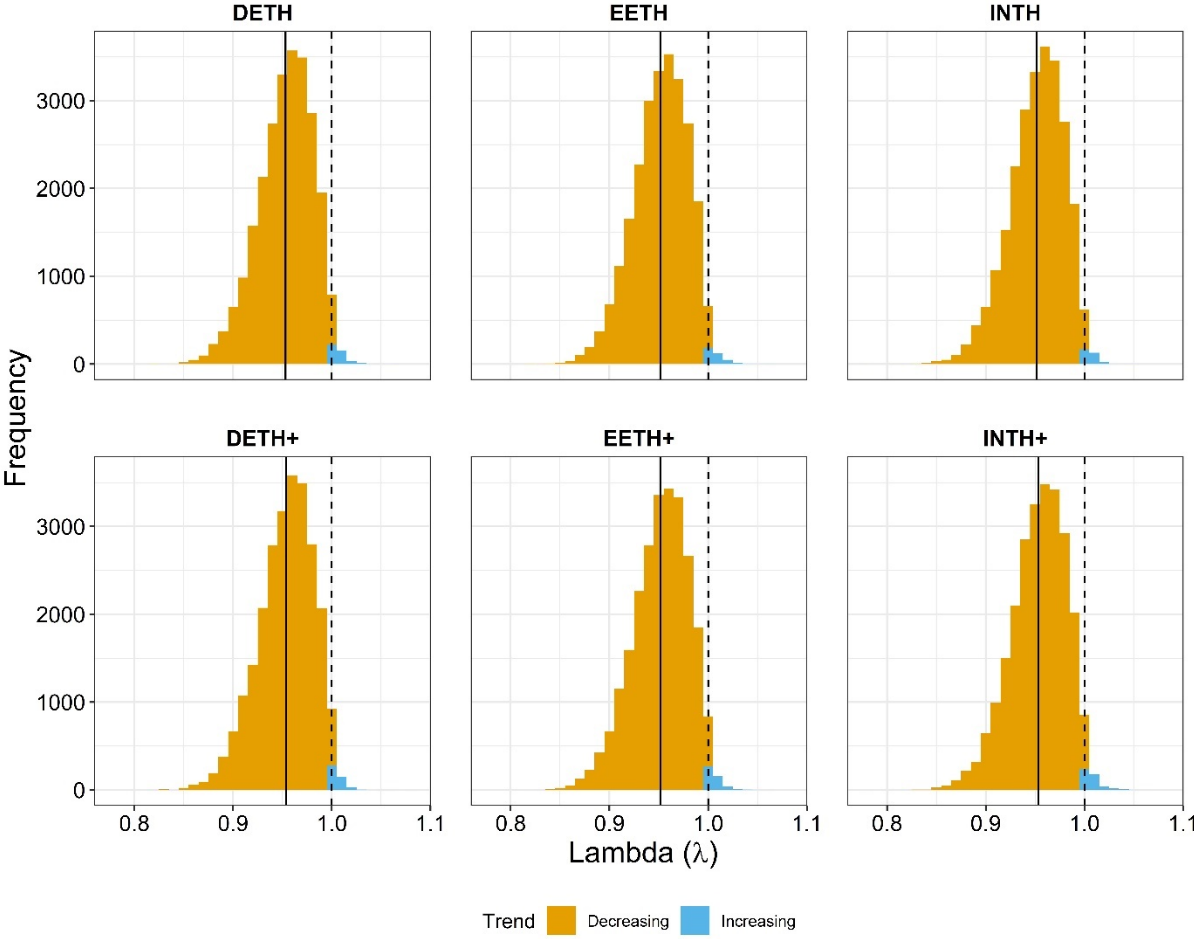

Figure E9. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices ($\mathbf{A}_u$) used to project alligator snapping turtle (*Macrochelys temminckii*) population dynamics of the Northern Mississippi – East analysis unit. Each panel 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).

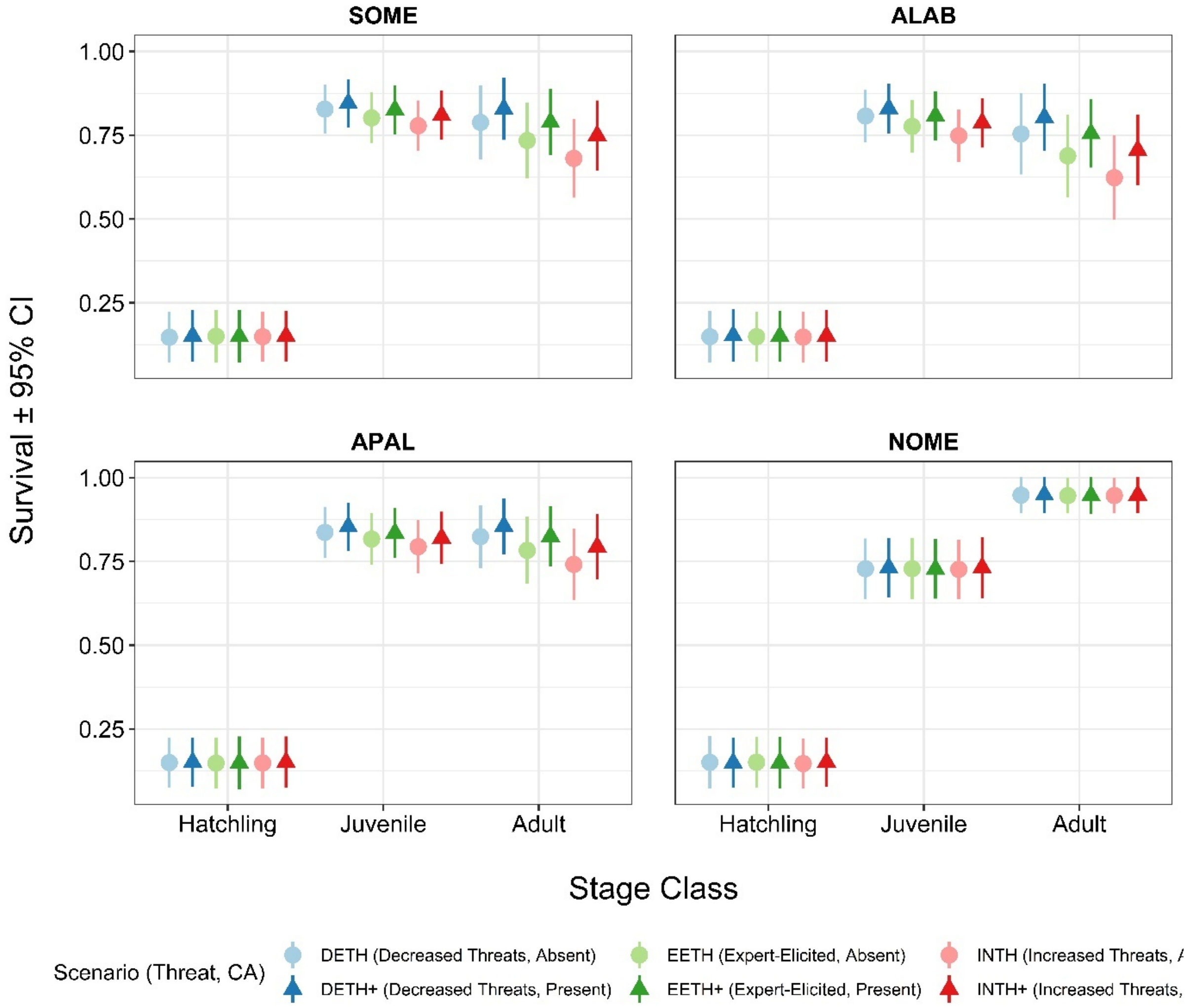

Figure E10. Mean stage class-specific alligator snapping turtle (*Macrochelys temminckii*; AST) survival parameters and their associated 95% confidence intervals (CI) for each analysis unit: Southern Mississippi – East (SOME), Alabama (ALAB), Apalachicola (APAL), and Northern Mississippi – East (NOME). 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).

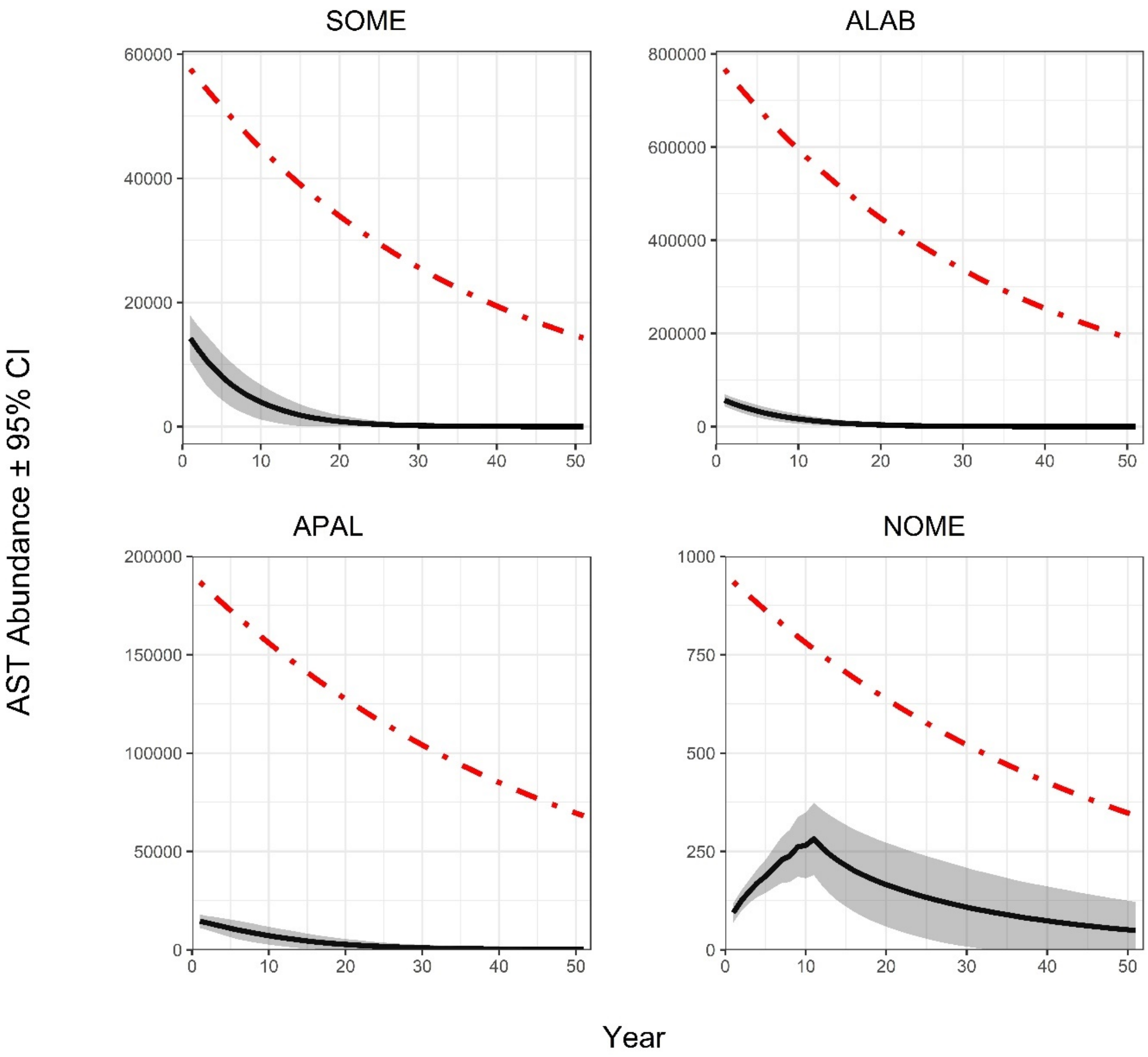

Figure E11. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) female total abundance over a 50-year period under the decreased threat with conservation action (DETH+) scenario for each analysis unit. Analysis unit abbreviations are listed above each panel and include: Southern Mississippi – 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.

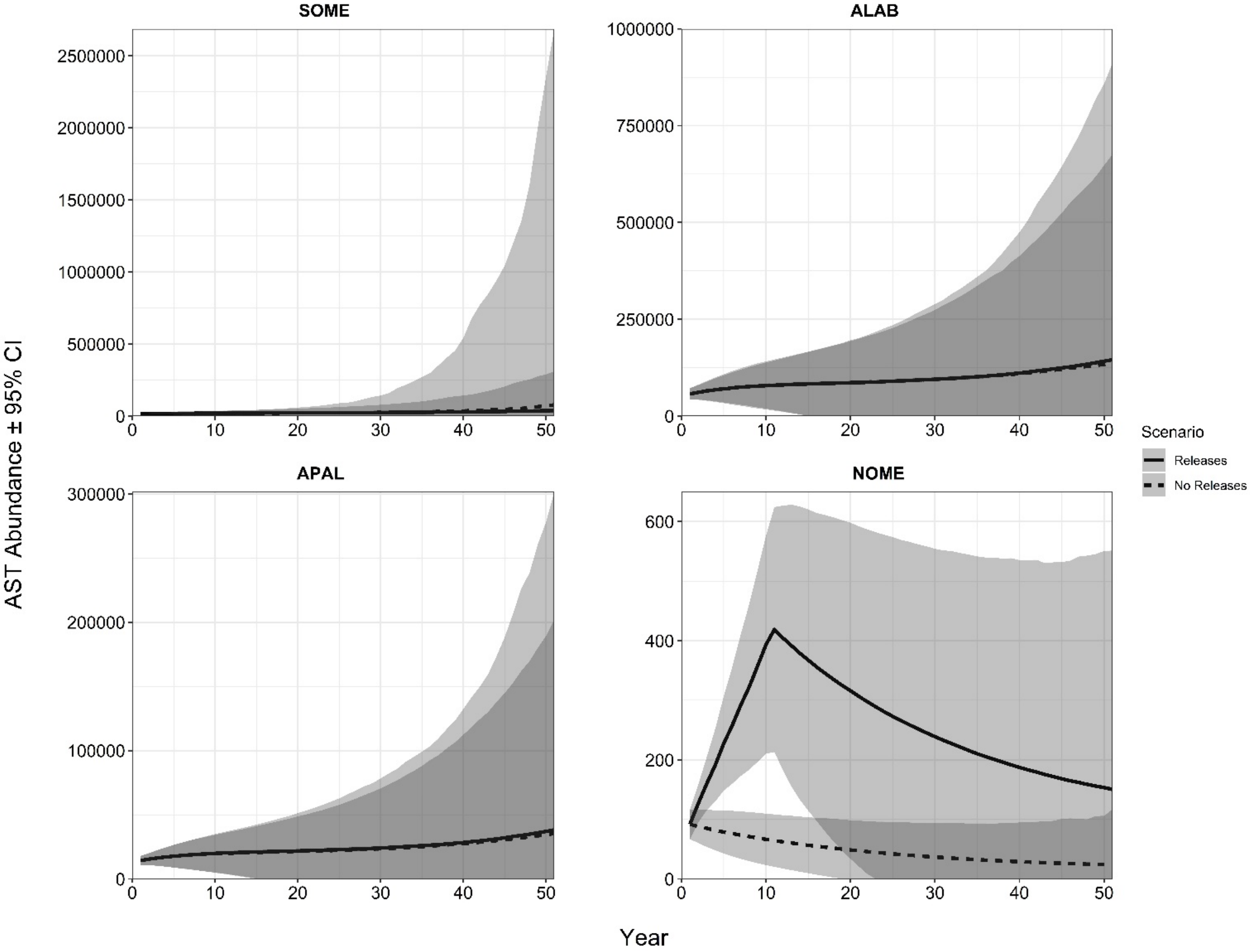

Figure E12. Simulated alligator snapping turtle (*Macrochelys temminckii*; AST) female total abundance over a 50-year period under the baseline scenario, with (solid line) and without (dashed line) head start and adult releases. The baseline scenarios used demographic parameters listed in Table E1, sampled 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