Species Status Assessment Report
for the
Suwannee Alligator Snapping Turtle
(*Macrochelys suwanniensis*)

Version 1.1

Suwannee Alligator Snapping Turtle
Photo credit: Christopher Coppola

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EXECUTIVE SUMMARY

The alligator snapping turtle (*Macrochelys temminckii*) was petitioned in 2012 by the Center for Biological Diversity (CBD) for listing under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531-1543) (Act) with a subsequent petition on September 1, 2015 to request the U.S. Fish and Wildlife Service (Service) to consider additional distinct species based on Thomas et al. (2014, entire). The Service acknowledged the alligator snapping turtles occurring in the Suwannee River basin as a separate species (*Macrochelys suwanniensis*) during the *M. temminckii* recommendation team meeting on October 15, 2019. This Species Status Assessment (SSA) serves as a compilation of the best available scientific information about *M. suwanniensis* 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 Service's forthcoming decision on whether *M. suwanniensis* warrants protection under the Act. Much of the data in this document is based on research completed before the genetic work in 2014 (Thomas et al. 2014, entire) and is presented at the genus or species level depending on location of research.

Turtles within the *Macrochelys* genus are the largest freshwater turtles 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 meters) 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) with nesting occurring from April to July (Reed et al. 2002, p. 4). 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). *Macrochelys spp*. display sexual dimorphism with males being distinctly larger than females and having a greater anterior-to-vent tail length.

Macrochelys spp. 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. *Macrochelys spp*. 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. *Macrochelys spp*. are opportunistic predators and foragers and consume a variety of foods. Fishes 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.

A table of individual, population, and species needs for the *M. suwanniensis* is below (Table EX1).

Table EX1. Individual, population, and species needs for *M. suwanniensis*.

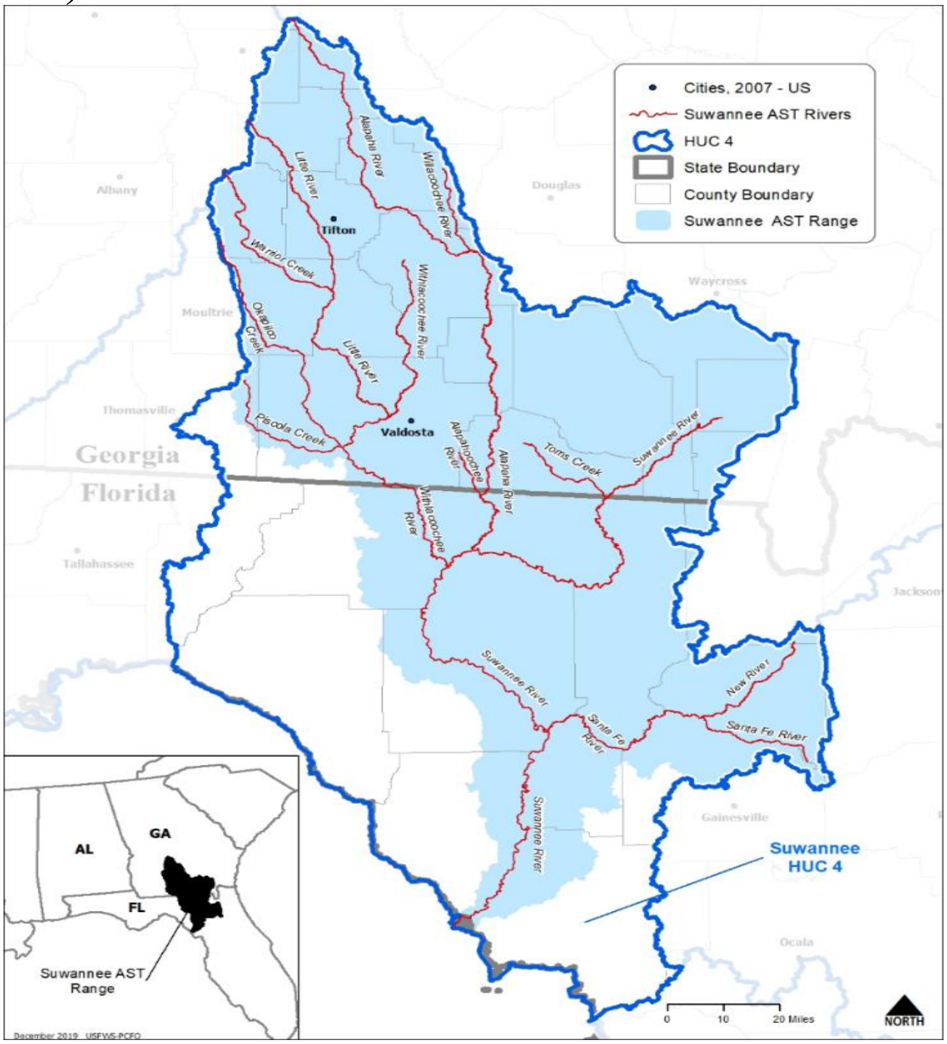
Life Stage	Need
Individual Needs	
Eggs	Temperatures 66 to 80° F (19 to 26.5° C) increasing to 79 to 98° F (26.1 to 36.5° C) as the season progresses
Eggs	Near shore areas (8 to 656 feet [2.5 to 200 meters] landward from the nearest water) with appropriate temperatures (see above)
Hatchlings	Shallow water and increased canopy cover
Juveniles	Found in small streams with mud and gravel bottoms (e.g., 8-18 inches [20-46 centimeters] 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 *M. suwanniensis* 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	This species exists as one population and thus redundancy is not applicable to our analysis of the species.
Representation	The concept of representation and representative units do not apply to this single basin endemic species.

Commercial and recreational harvesting in the last century resulted in minimal declines to *M. suwanniensis* (Enge et al. 2014, p.40). Commercial and recreational harvest is now prohibited in Florida and Georgia. Although regulatory harvest restrictions have decreased the quantity of *M. suwanniensis* being harvested, the species has 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 *M. suwanniensis* are: bycatch and hook ingestion associated with recreational fishing, habitat alteration, and nest predation. Climate change and disease may negatively influence the species, but the impacts of these drivers on the species are more speculative due to lack of information. Conservation measures implemented to preserve and improve riverine habitat, and incidentally *M. suwanniensis*, in Florida and Georgia include limiting deadhead logging and the removal of snags throughout the Suwannee River basin.

The range of the M. suwanniensis is limited to the Suwannee river basin (Suwannee, Withlacoochee, Alapaha, Santa Fe, New and Ichetucknee rivers) and constitutes one population (Figure EX1).

Figure EX1. *M. suwanniensis population occurs entirely within the Suwannee Hydrological Unit Code (HUC) 4.*



Current Conditions

To assess the current condition of *M. suwanniensis*, information was gathered from species experts about current abundance (our measure of resilience), current threats, and a comparison of the current and historical distribution. Currently, abundance is estimated at 2,000 individuals with approximately 76.2 turtles per 1,000 hectares (2,471 acres) of open water. Turtles are believed to be exposed to the threat of incidental hooking on recreational trot and limb lines, illegal harvest, and nest predation by native and exotic species.

Future Conditions

To assess future conditions and viability of the *M. suwanniensis*, we constructed a female-only, stage-structured matrix population model to project *M. suwanniensis* population dynamics over 50 annual time steps for the single population occupying the Suwannee River basin. 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 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, 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 *M. suwanniensis* 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 *M. suwanniensis* population declined to less than 5% of the abundance in year one of the simulations (e.g., starting abundance).

Resilience is expected to decline across the Suwannee River basin under all scenarios. We modeled scenarios that reflected uncertainty in the impact of threats on *M. suwanniensis* demography, and all scenarios produced mean growth rates indicating population decline. Predicted abundances were likely to virtually certain (i.e., had a >66% probability) of dropping below 5% of current abundances within 50 years. Time to quasi-extinction in the Suwannee River basin is between 32 and 42 years.

The future condition scenarios predict that resiliency will decline at a high rate within the Suwannee River basin. Where *M. suwanniensis* persist in the future, they are likely to be rare and not found in resilient groupings. The addition of conservation actions, or different assumptions about the impact of threats on *M. suwanniensis* demography, altered the time to quasi-extinction by about a decade at most, typically less.

Contents

ACRONYMS

Act	Endangered Species Act
C	Celsius
CBD	Center for Biological Diversity
CDD	Continuous Dry Days
CITES	Convention on International Trade in Endangered Species
CO_2	Carbon Dioxide
CPUE	Catch-Per-Unit-Effort
F	Fahrenheit
Ft	feet
GHG	Greenhouse Gas
M	meter
RCPs	Representative Concentration Pathways
Service	U.S. Fish and Wildlife Service
SSA	Species Status Assessment
TSD	Temperature-dependent sex-determination
TTN	Turtles per trap-night

CHAPTER 1 – INTRODUCTION AND ANALYTICAL FRAMEWORK

The Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) is a reptile from Georgia and Florida that is confined to the Suwannee River basin that flows into the Gulf of Mexico from Georgia and Florida. On July 11, 2012, we, the U.S. Fish and Wildlife Service (Service), 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 (*Macrochelys temminckii*) 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 *M. temminckii* should be considered a distinct species. During the Services' review of the most current information, the Suwannee alligator snapping turtle was found to be a distinct species, *M. suwanniensis,* and separate from the alligator snapping turtle, *M. temminckii.* A review of the status of the *M. suwanniensis* as its own species was initiated to determine if the petitioned action is warranted. Based on the results of the status review, the Service will issue a 12-month finding for *M. suwanniensis*. 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. Much of the information used in the SSA describes *Macrochelys spp*. however, when information was available specifically for *M. suwanniensis* or its habitat, that information was further analyzed in the report to determine the species' viability. This SSA Report documents the results of the comprehensive status review for the *M. suwanniensis* 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 (U.S. Fish and Wildlife Service [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 *M. suwanniensis*. 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 *M. suwanniensis*.

The objective of this SSA is to thoroughly describe the viability of the *M. suwanniensis* 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** (USFWS, 2016, entire).

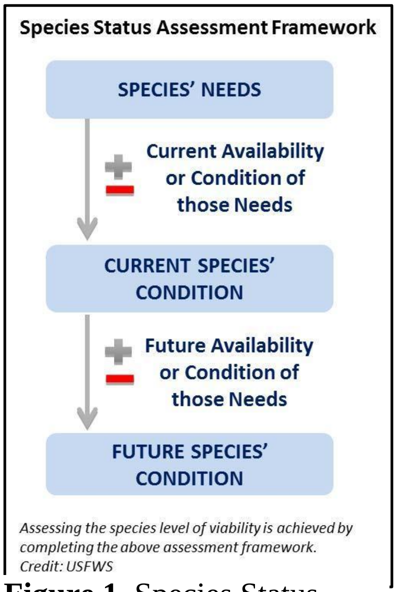


Figure 1. Species Status Assessment Framework

- **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. <u>Introduction</u>;
2. <u>Species Biology and Individual Needs</u>. 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 *M. suwanniensis* 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 *M. suwanniensis*. Importantly, this SSA Report does not result in, nor predetermine, any decisions by the Service under the Act. In the case of *M. suwanniensis*, the SSA Report does not determine whether *M. suwanniensis* 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 *M. suwanniensis*.

CHAPTER 2 – SPECIES BIOLOGY AND INDIVIDUAL NEEDS

In this chapter, we provide biological information about the *M. suwanniensis*, 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

M. suwanniensis 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 (*C. serpentina*), 2) South American snapping turtle (*C. acutirostris*), and 3) Central American snapping turtle (*C. rossignonii*). The nomenclatural history of the *M. suwanniensis* 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, Lovich 1993, p. 562.1-562.2). 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 (*M. temminckii*) across much of the southeastern United States until a recent analysis of variation in morphology and genetic structure described two new species of alligator snapping turtles: the

Apalachicola alligator snapping turtle (*M. apalachicolae*) and the Suwannee alligator snapping turtle (*M. suwanniensis*; Thomas et al. 2014, entire).

Three genetically distinct lineages of *Macrochelys* were identified morphologically, with *M. suwanniensis* being the most distinct (Thomas et al. 2014, p. 161). The carapace of *M. suwanniensis* can be differentiated by the presence of a large, lunate caudal notch, whereas *M. temminckii* and *M. apalachicolae* have narrow, triangular or U-shaped caudal notches that are more difficult to differentiate from each other. The skulls of *M. temminckii* and *M. apalachicolae* have large, globular squamosal projections, whereas the skulls of *M. 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 *M. temminckii* identified: 1) a western clade including populations from the Trinity River to Pensacola Bay (retained as *M. temminckii*), 2) a central clade from the Choctawhatchee River to the Ochlockonee River (corresponding to *M. apalachicolae*), and 3) an eastern clade restricted to the Suwannee River (corresponding to *M. 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 *M. apalachicolae* from *M. 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 River basin 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 River basin 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 (*M. apalachicolae*) from the western population (*M. temminckii;* Folt and Guyer 2015, p. 449-450). In addition, there are seven rivers between the Suwannee River basin 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 *M. suwanniensis (*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) *M. temminckii* and 2) *M. suwanniensis*. The Turtle Taxonomy Working Group concurred with the recognition of two species since Folt and Guyer (2015, entire) reconsidered published data, critiqued the methods of Thomas et al. (2014), and provided evidence to support the distinction of *M. suwanniensis* (Rhodin et al. 2017, p. 26). For the purpose of this SSA report, we describe the species using the scientific name, *Macrochelys suwanniensis* rather than the common name, Suwannee alligator snapping turtle to avoid confusion with alligator snapping turtle (*M. temminckii*). We also used information regarding *M. temminckii* and *Macrochelys spp*. in general in sections of the report that relate to life history where information was not available specific for *M. suwanniensis* and is noted throughout the document.

2.2 Species Description

The genus *Macrochelys* includes the largest freshwater turtle species in size found in North America. These turtles are highly aquatic and somewhat secretive. They are primitive in appearance and are characterized by a large head with an acute, sharp squamosal projection, long tail, and an upper jaw with a strongly hooked beak. They have muscular legs and webbed toes with long, pointed claws. *M. suwanniensis*'s carapace has a large, lunate caudal notch and three keels with posterior elevations on the scutes. Their dark brown carapace often has algal growth that adds to their 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 grayish-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).

2.3 Range and Distribution

Historical distribution records of *M. suwanniensis* are sparse, with most trapping occurring near easy access areas to streams and rivers (i.e. road crossings and boat launches). Allen and Neill (1950, p. 1) report *M. suwanniensis* in Florida from the Suwannee River in Dixie and Levy counties and from the Santa Fe River in Suwannee, Bradford, and Alachua counties. Reports of *M. suwanniensis* occurring in the Ocklawaha River, Marion Co. may be escapees from the Ross Allen's Reptile Institute at Silver Springs, Marion Co. (Moler, 1996, p.1; Florida Fish and Wildlife Conservation Commission, 2015, p.2), but a voucher specimen exists from 1916 (AMNH 8287). Reports of *M. suwanniensis* from the Okefenokee Swamp have occurred since 1912 (Allen and Neill, 1950, p.1; De Sola and Abrams, 1933, p. 11; Wright and Funkhouser, 1915, p. 111, and S. Aicher 2020, personal communication), but they are sparse and *M. suwanniensis* have not been found in any of the other waterways draining from the swamp.

Research indicates the current occupied bodies of waters are (Figure 2):

Florida – Suwannee River, Hunter Creek, Rocky Creek, Santa Fe River, New River, Ichetucknee River, Cow Creek, Alapaha River, and Withlacoochee River (Enge et al. 2014, p. 19 – 20; Johnston et al. 2015b, p. 75 – 76; Jackson and Thomas 2018, entire; K. Enge 2020, pers. communication)

Georgia – Suwannee River, Withlacoochee River, Alapaha River, Alapahoochee River, Willacoochee River, Little River, Tom's Creek, Warrior Creek, Okapilco Creek, and Piscola Creek (Stevenson, 2019, entire).

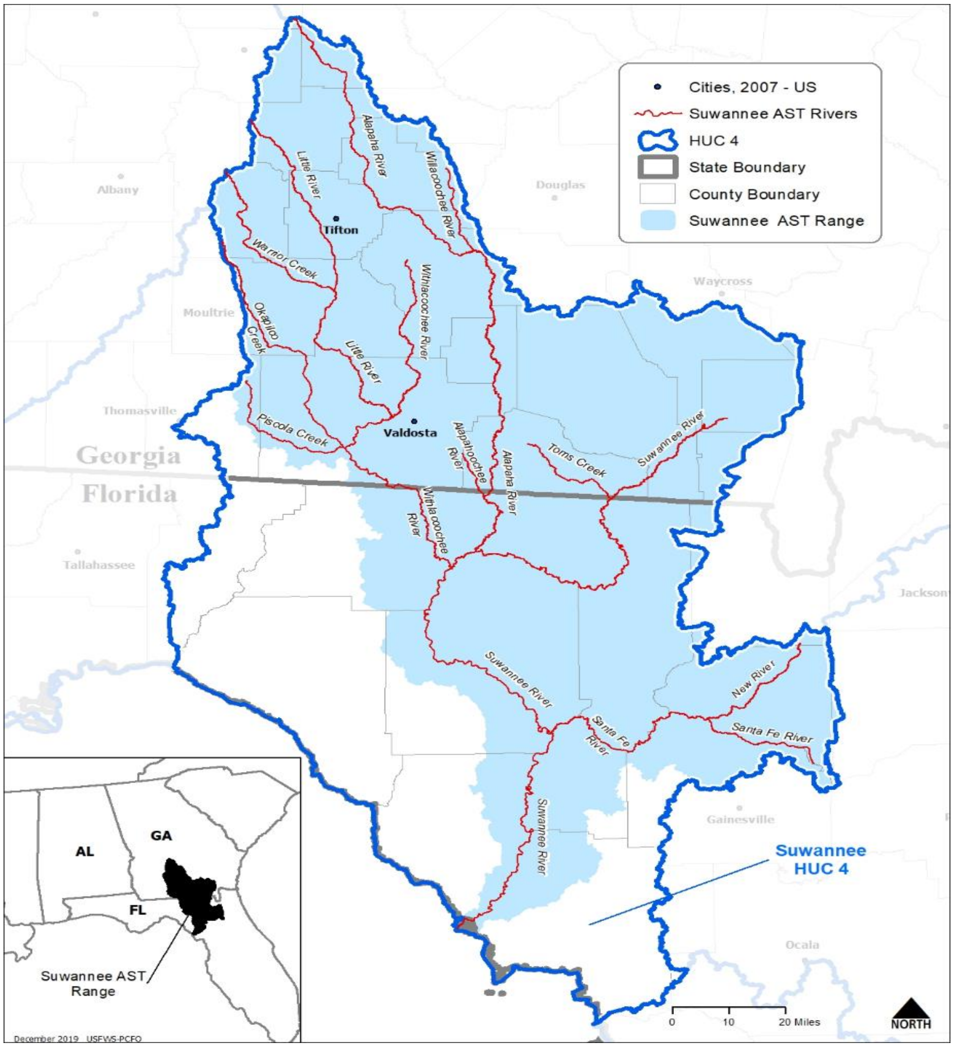


Figure 2. Range distribution of *M. suwanniensis* in Florida and Georgia.

2.4 Habitat

Macrochelys spp. 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, springs, bayous, canals, swamps, lakes, reservoirs, ponds, floodplains during flooding, and oxbows (a lake that forms when a meander of a river is cut off; Ernst and Lovich 2009, p. 141). *Macrochelys spp*. 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).

In Florida, habitat has been identified as floodplain swamp forests comprised of bald cypress and tupelos associated with close association with numerous flooded channels (Ewert et al. 2006, p.61, Ewert and Jackson 1994, p. 3 – 4). The Suwannee River basin waterways are fed by artesian springs and spring runs which may provide additional thermally stable refugia or optimal habitat (Enge et al. 2014, p. 39). In the upper Suwannee River, M. suwanniensis were observed or trapped in Hunter and Rocky creeks, which are small, blackwater tributaries (K.

Enge 2020, pers. communications). The amount of suitable habitat available to *M. suwanniensis* within its range and a description of how those numbers were derived is presented in Appendix E.

Barnacles have been observed growing on shells of *M. temminckii.* in Dog River, about 2 mi upriver from Mobile Bay, which implies a certain level of salt tolerance (Jackson and Ross 1971, p. 188). In addition, *M. temminckii* have been documented on Tyndall Air Force Base (Lane and Mitchell 1997, p. 6) where the individual(s) would have needed to transverse through brackish water from a river to the coastal military installation.

2.5 Diet and Feeding

Macrochelys suwanniensis is an opportunistic scavenger and consumes a variety of foods. Fish comprise a significant portion of the *M. suwanniensis* diet; however, crayfish, mollusks, smaller turtles, insects, snakes, birds, and vegetation (including acorns) have also been reported (Elsey 2006, p. 448-489; Elbers and Moll 2011, entire). *M. suwanniensis* also consume fruit of the common persimmon (*Diospyros virginiana*) and may function as a seed disperser (Johnston et al. 2015a, p. 59–60; Elbers 2010, entire). *Macrochelys spp*. are the only turtle species that have a predatory lure (a small, worm-like appendage on the tongue. 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). Ernst and Lovich (2009, p.148) describe four phases to feeding behavior: waiting, luring, attack, and handling. Success in each phase increases 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 low reproductive output, present levels of nest predation may be detrimental to *Macrochelys spp*. 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).

Macrochelys spp. 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*) (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 declines in hatching success. *Macrochelys spp*. 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 *Macrochelys spp*.

2.7 Movement and Behavior

Macrochelys spp. 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 (Ernst and Lovich 2009, p. 141), and most reports consist of a single observation (Carr et al. 2011, p. 3; Ewert 1976, p. 154; and Thomas 2009, p.336). Home range for adult *M. suwanniensis* in the Suwannee River has had limited research. Results are shown in Table 1.

Table 1. Results from radio telemetry studies on *M. suwanniensis* in the Suwannee River.

Study	**State**	**Site**	**Sex**	***n***	**Mean Minimum Linear Home Range (m)**
Enge et al. 2014, p. 39	Florida	Suwannee River	Male	9	3,986 ± 8545
			Female	5	2,061 ± 355
			Juvenile	2	2,745 ± 1,930

2.8 Life Cycle and Reproduction

Macrochelys spp. sexual maturity is achieved in 11-21 years for males and 13-21 years for females (Figure 3) (Ernst and Lovich 2009, p. 144; Reed et al. 2002, p. 4). Mating takes place underwater (Ernst and Lovich 2009, p. 144) and has been observed in captive *M. temminckii* from February to October. 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, *C. serpentina*; Reed et al. 2002, p. 4). Clutch sizes for *Macrochelys spp*. have been reported from across the species' range (9-61 eggs, with a mean of 27.8 (Ernst and Lovich 2009, p. 145). Two clutches of *M. suwanniensis* in the wild had 43 and 47 eggs (Jackson and Thomas 2018, entire), and six clutches from captive *M. suwanniensis* had a mean of 24.5 eggs (range 16–44) (Allen and Neill 1950, entire). Reproductive output also varies substantially among females but generally is positively correlated with body size (Reed et al. 2002. p. 4). Larger (older) females probably produce more eggs than recently matured females (Ernst and Lovich 2009, p. 145). 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 short distances 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). *Macrochelys spp*. 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). *Macrochelys spp*. nests have been observed approximately 8-656 feet (2.5 to 200 meters) landward from the nearest water (Ernst and Lovich 2009, p. 145).

Internal temperature of nests in Florida were between 66 and 80° Fahrenheit (F) (19-26.5° Celsius [C]) initially and increased 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 begins in April and extends through May (Ernst and Lovich 2009, p. 145; Carr et al. 2010, p. 87). Holcomb and Carr (2011*a*, p. 225) estimated the incubation period was 98-121 days and estimated emergence of hatchlings was 0.5 – 22 days.

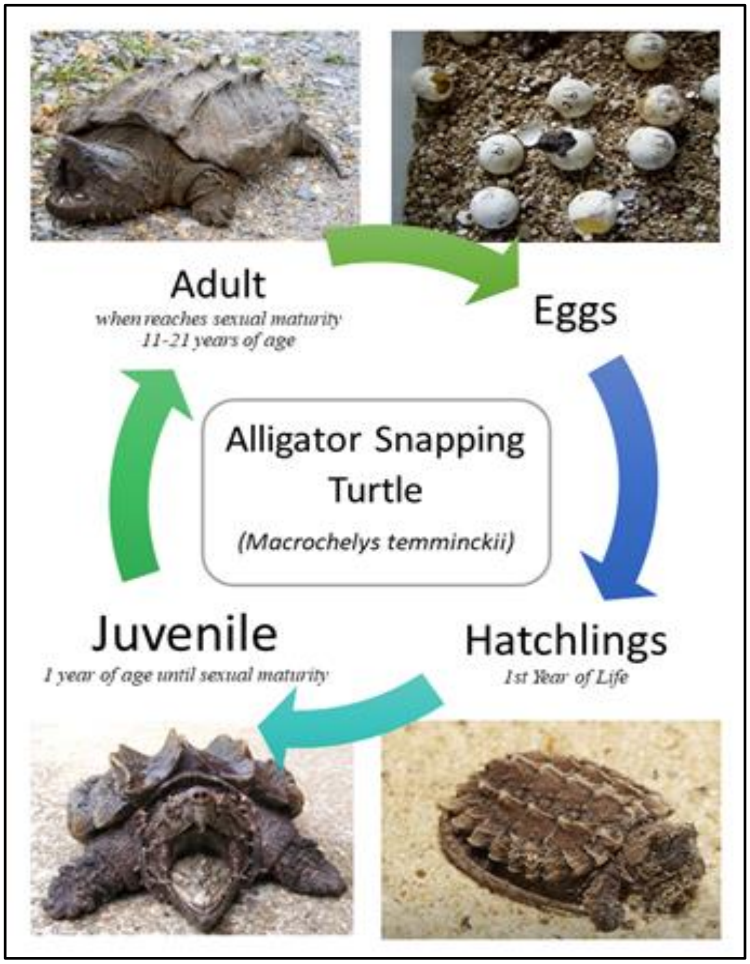


Figure 3. *Macrochelys* life cycle. Photo credits: Eva Kwiatek (top left), US Fish and Wildlife Service (top right), Indiana DNR (bottom left), Kory Roberts (bottom right).

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 spp*. (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). Population modeling of *M. suwanniensis* in the Suwannee River in Florida indicated an estimated survival of 0.98 for adults (T. Thomas, 2020. pers. communication).

Rate of survivorship of juveniles is estimated at only about 5%, with most mortality occurring in the first two years of life (Ernst and Lovich 2009, p. 150). In a non-declining population of *Macrochelys spp*., however, juvenile apparent survival has been reported as 0.86 (Folt et al. 2016, p. 27). Mean generation time for *Macrochelys spp*. 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). In captivity, 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 when it died (Ernst and Lovich 2009, p. 147).

Growth data are also scarce for wild *Macrochelys spp*. Annual weight growth rate has been reported as 5.3% in males and 5.2% in females, with males growing significantly faster than females (Ernst and Lovich 2009, p. 146). Growth is rapid until maturity (11-13 years of age), slowing after 15 years of age (Dobie 1971, p. 654). Immature *M. suwanniensis* in the Santa Fe River were observed to grow 13.3–19.1 mm carapace length (CL) / year, suggesting approximately 20 years of growth are required to attain sexual maturity (Johnston et al. 2012, p. 474). 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 factors. 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). Estimates of this index for *M. suwanniensis* in the Santa Fe River (Johnston et al. 2015b, p. 78) are -2.0 by mass (34 kg male/17 kg female) and -1.2 by length (53.1 cm CL male/42.4 cm CL female).

M. temminckii 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 for *M. suwanniensis* in Florida (3.5:1; Enge et al. 2014, p. 32 and Thomas 2013, p. 41), but it varied among sections of the river. A 1:1 adult sex ratio was reported for *M. suwanniensis* in the Santa Fe River (Johnston et al. 2015b, p. 76).

A ratio of *Macrochelys spp*. juveniles to adults has been reported at 1:4 in Georgia (Jensen and Birkhead 2003, p. 29). 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 *Macrochelys spp*. (Folt et al. 2016, p. 29). A 1:3 ratio was reported for *M. suwanniensis* in the Santa Fe River (Johnston et al. 2015b, p. 78–79).

Catch-Per-Unit-Effort (CPUE) can be used as an indirect measure of the abundance of a species. For the *M. suwanniensis*, 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). Table 2 lists CPUE reported in the Suwannee River basin.

Table 2. Catch per Unit Effort reported for *M. suwanniensis* in the Suwannee River basin.

River	**State**	**Catch-Per-Unit-Effort**	**Reference**
Suwannee River	Florida	0.263	Moler 1996, p.10
Santa Fe River	Florida	0.09	Moler 1996, p.10
Suwannee, Little, Withlacoochee, and Alapaha Rivers	Georgia	0.05	Jensen and Birkhead, 2003, p. 29
Suwannee River	Florida	0.22	Enge et al. 2014, p. 19
Suwannee River	Florida	0.217	Thomas et al. 2013, p. 40
Upper Santa Fe River	Florida	0.181	Johnston et al. 2015b, p. 75
Lower Santa Fe River	Florida	0.078	Johnston et al. 2015b, p. 75
Alapaha, Alapahoochee, and Willacoochee Rivers	Georgia	0.19	Stevenson, 2019, p. 12
Little River and Warrior Creek	Georgia	0.28	Stevenson, 2019, p. 12
Suwannee River, Cypress, Suwannoochee, and Tom's Creeks	Georgia	0.04	Stevenson, 2019, p. 12
Withlacoochee River, Okapilco, and Piscola Creeks	Georgia	0.17	Stevenson, 2019, p. 12
Suwannee, Alapaha, and Withlacoochee Rivers	Georgia	0.05	Stegenga, 2019, p. 9

2.10 Summary of Species Biology and Individual Needs

Macrochelys spp. are the largest species of freshwater turtle in North America (Ernst and Lovich 2009, p. 138) and are 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. 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 meters) 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 April to May (Reed et al. 2002, p. 4). 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).

M. suwanniensis 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. *M. suwanniensis* 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. *M. suwanniensis* are opportunistic scavengers and consume a variety of foods. Movements can be highly variable but are generally a few to hundreds of feet per day.

The individual needs of *M. suwanniensis* are summarized in Table 3.

Table 3. *M. suwanniensis* 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; also exhibits TSD-2	Survival, Sheltering	Ernst and Lovich 2009, p. 16, 146, Enge et al. 2014, p. 4
Eggs/ Nests	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, p.151, Ewert et al. 2006, p. 63 – 65, Powders 1978, p.155, Trauth et al. 2004, n.p., Enge et al. 2014, p. 4
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, Enge et al. 2014, p. 4
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, Thomas 2013, p. 62, Johnson et al. 2015, p. 77, Stevenson 2019, p.3
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, Thomas 2013, p. 42
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 *M. suwanniensis*, including illegal intentional harvest, bycatch, habitat alteration, nest predation, climate change, disease, and conservation measures (Figure 4).

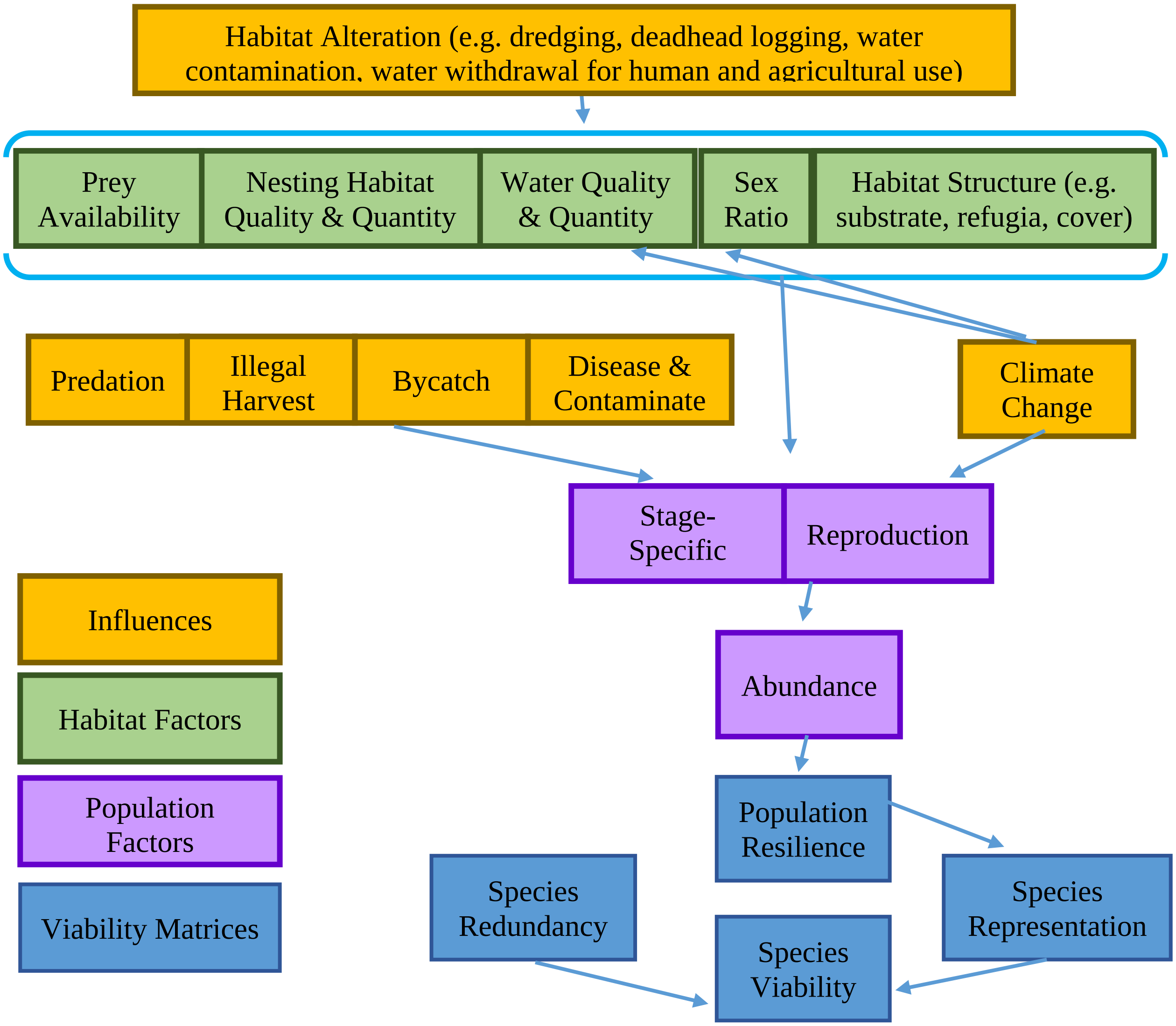


Figure 4. Simplified influence diagram illustrating how various impacts influence habitat and population factors that in turn influence the resilience of populations and viability of the species.

3.1 Harvest

3.1.1 Commercial Harvest

Extensive commercial harvest in the last century resulted in significant declines to many *Macrochelys spp*. populations across the species' range (Enge et al. 2014, p. 4) and reached its peak in the late 1960s and 1970s when the meat was sold for turtle soup.

Commercial harvest impacts to *M. suwanniensis* is unknown, but thought to be less due to the remoteness of the Suwannee River. Florida prohibited the commercial harvest of all *Macrochelys spp*. in 1972 and Georgia prohibited all harvest in 1992.

3.1.2 Recreational Harvest

Recreational harvest of *Macrochelys spp*. was allowed in Georgia prior to 1992 at which time the state listed the species as threatened. In 1992 Florida started to allow the possession of one (1) turtle for recreational harvest but ended this in 2009 when the state listed *Macrochelys spp*. as a Species of Special Concern and possession became illegal.

3.1.3 Impacts of Harvest

Because of *M. suwanniensis*'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). *M. suwanniensis* 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 *M. suwanniensis* 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 for *M. temminckii* in Georgia, 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).

3.1.4 International Trade and Illegal Harvest

In 2006, *M. temminckii* was listed under Convention on International Trade in Endangered Species (CITES), as an Appendix III species to allow for better monitoring of exports. Prior to that listing, up to 23,780 *M. temminckii*/ year were exported from the U.S. Since the CITES listing, up to 43,718 live *M. temminckii* have been identified as "specimens taken from the wild" leaving the U.S. in a single year (USFWS 2018). No CITES-permitted farms occur in Florida or Georgia.

Illegal Harvest

There is some evidence of illegal harvest, as well. For instance, in 2017, three men were convicted of collecting 60 large *M. temminckii* in a single year in Texas and transporting them across state lines violating the Lacey Act (Eastern District of Texas Department of Justice, 2017, entire). While several closed cases involving *Macrochelys spp*. 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.

Illegal poaching of many turtle species is an issue globally. *M. temminckii* are exported for the pet trade and also as a source of food.

3.2 Bycatch

M. suwanniensis can be killed or harmed incidental to other fishing and recreational activities. Threats include capture as bycatch associated with recreational 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.

M. suwanniensis 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 *M. suwanniensis* 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 10; 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). Damage caused by boat propellers can also injure *M. suwanniensis* and cause extensive damage to their carapaces, though effects on population demographic rates are unknown (Enge et al. 2014, p. 41).

3.3 Habitat Alteration

M. suwanniensis 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 water withdrawal for agriculture, 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 *M. suwanniensis* based on their habitat needs, but actual impacts of these processes on *M. suwanniensis* have not been quantified.

3.4 Nest Predation

As described in Chapter 2, nest predation rates for *Macrochelys spp*. 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 *Macrochelys spp*. (Pritchard 1989, p. 69). Predation by fire ants was the suspected cause of nest failure in seven of 16 *M. temminckii* naturally incubated nests (in contrast to artificial nests) at Black Bayou Lake in Louisiana (Holcomb 2010, p. 51). Beyond nest failure, some *M. temminckii* 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 *M. temminckii* nests investigated in Louisiana was infested by the phorid fly *Megaselia scalaris*, the first documentation of infestation by fly larvae in 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 *M. suwanniensis* in several ways, including loss of habitat to sea level rise for individual turtles near the coast, 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).

Macrochelys spp. exhibit TSD, 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. 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 *M. suwanniensis* 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.

3.7 Disease and Health

Chaffin et al. (2008, entire) captured and assessed the health of 97 free-ranging *Macrochelys spp*. across nine sites in northwestern Florida and southwestern Georgia between 2001 and 2006. Assessed 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 *M. temminckii* tested from Pataula Creek, Georgia. None were showing symptoms, and *M. temminckii* 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 *C. serpentina* (Hopkins et al. 2013, p. 2418-2419), but the levels of mercury detected in *Macrochelys spp*. were low relative to those detected in the *C. serpentina* study and are unlikely to have very large effects on reproduction. More direct exposure to environmental mercury that leads to higher mercury levels in *Macrochelys spp*. would be expected to impact reproduction as well as other aspects of health.

3.8 Conservation measures

Species Protections

The FWC directs staff to evaluate all species listed as Threatened or Species of Special Concern as of September 1, 2010, as required by rule 68A-27.0012 Florida Administrative Code. Since the original 2010 biological status review, Thomas et al. (2014, p. entire) described 2 new species of alligator snapping turtle based upon genetic and skeletal differences, necessitating new biological status reviews of all species. During the 2017 biological assessment, it was determined by the biological review group that *M. suwanniensis* was distinct and warranted listing as Threatened based upon IUCN Red List criteria (FWC, 2017. p.3). This determination lead to the development of a Species

Action Plan (SAP) for Florida's alligator snapping turtles. The SAP includes all *Macrochelys spp*. due to their similarity in appearance, vulnerability to deliberate human take, incidental take with fishing gear, pollution, riverine habitat alteration, and nest predation (FWC 2018, p.iii). The objectives of the SAP include: Habitat Conservation and Management, Population Management, Monitoring and Research, Rule and Permitting Intent, Law Enforcement, Incentives and Influencing, Education and Outreach, and Coordination with Other Entities (FWC 2018, p.10-27).

Georgia listed *Macrochelys spp.* as threatened in 1992. In the State's Wildlife Action Plan, the Department of Natural Resources indicate they intend to conduct genetic, taxonomic and reproductive studies of high priority species (GDNR 2015, p. D-5)

State and Federal Stream Protections (Deadhead Logging)

Deadhead logging is the removal of submerged cut timber from a river or creek bed and banks. The structural diversity and channel stabilization created by instream woody debris has been found to be essential in providing habitat for spawning and rearing aquatic species (Bilby 1984, p. 609 and Bisson et al. 1987, p. 143). Wallace and Benke (1984, p. 1651) reported that snag or woody habitat was the major stable substrate in southeastern Coastal Plain sandy-bottom streams and a site of high invertebrate diversity and productivity. Wood enhances the ability of a river or stream to utilize the nutrient and energy inputs and has a major influence on the hydrodynamic behavior of the river (Wallace and Benke 1984, p. 1643). Florida allows deadhead logging with the proper permits from Florida Department of Environmental Protection and Georgia is not currently processing permits.

State and Federal Stream Protections (Buffers & Permits)

A buffer is a strip of trees, plants, or grass along a stream or wetland that naturally filters out dirt and pollution from rain water runoff before it enters rivers, streams, wetlands, and marshes (Southern Environmental Law Center 2014, p. 2). Loss of riparian vegetation and canopy cover result in increased solar radiation, elevation of stream temperatures, loss of allochthonous (organic material originating from outside the channel) food material, and removal of submerged root systems that provide habitat for fish and macroinvertebrates (Allan 2004, p. 266-267). The Georgia Erosion and Sediment Control Act restricts disturbance and trimming of vegetation within a 25 foot (7.62 m) buffer adjacent to creeks, streams, rivers, saltwater marshes and most lakes and ponds and the Georgia Planning Act require some local governments to adopt a 100 foot (30.48 m) buffer. The Florida Surface Water Improvement and Management plan addresses statewide non-point source pollution impacts to waterbodies on a landscape scale and partners' federal, state, local government, and the private sector to restore damaged ecosystems and prevents pollution from storm water runoff.

Section 401 of the federal Clean Water Act (CWA) requires that an applicant for a federal license or permit provide a certification that any discharges from the facility will not degrade water quality or violate water-quality standards, including state-established water quality standard requirements. Section 404 of the CWA establishes programs to regulate the discharge of dredged and fill material into waters of the United States.

Permits to fill wetlands, to install, replace, or remove culverts, to install, repair, replace, or remove bridges, or to re-align streams or water features are issued by the U.S. Army Corps of Engineers under Nationwide, Regional General Permits or Individual Permits include:

- Nationwide Permits are for “minor” impacts to streams and wetlands, and do not require an intense review process. These impacts usually include stream impacts under 150 feet (45.72 m), and wetland fill projects up to 0.50 acres (0.2 hectare). Mitigation is usually provided for the same type of wetland or stream impacted, and is usually at a 2:1 ratio to offset losses and make the “no net loss” closer to reality.
- Regional General Permits are for various specific types of impacts that are common to a particular region; these permits will vary based on location in a certain region/state.
- Individual permits are for the larger, higher impact and more complex projects. These require a complex permit process with multi-agency input and involvement. Impacts in these types of permits are reviewed individually and the compensatory mitigation chosen may vary depending on project and types of impacts.

3.9 Summary of Factors Influencing Viability

Historically, commercial and recreational take in the last century resulted in significant declines of the *M. suwanniensis* population in some Georgia streams. Commercial and recreational harvest is now prohibited in Florida and Georgia. 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 *M. suwanniensis* are: 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.

Conservation measures implemented by Florida and Georgia have prohibited the commercial and recreational harvest of *Macrochelys spp.* and have implemented state regulations to protect and minimize impacts to rivers, streams and riparian areas. Waterway protections limit the amount of woody vegetation removed, water removal for municipal and agricultural use, and water quality. Corps of Engineers permit applicants must comply with Section 401 and 404 of the CWA and state regulations when they receive a permit to discharge dredged and fill material into waters of the United States.

CHAPTER 4 – POPULATION AND SPECIES NEEDS AND CURRENT CONDITION

In this chapter, we first describe population and species needs, describe how we collected information to assess resilience, and then summarize the current resilience and redundancy for the single basin endemic population of *M. suwanniensis*.

4.1 Population Needs

For the population to persist, the needs of individuals (Table 3) 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 *M. suwanniensis* to survive, find mates, and reproduce while avoiding inbreeding depression. To persist, the population 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 the single population of *M. suwanniensis*.

4.2 Species Needs

For the species to be viable, *M. suwanniensis* require redundancy and representation of resilience within the population. Redundancy of resilience within the population distributed across the species' range is necessary to buffer the species against the effects of catastrophic events on any section of inhabited waterway. 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 the population that contributes to the ability of the species to respond and adapt to changing environmental conditions over time. Maintaining resiliency 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 *M. suwanniensis* is described in the following section.

4.5 Current Condition Methods

To assess the current condition of *M. suwanniensis*, we surveyed species experts in Florida and Georgia 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 *M. suwanniensis* and impacts of influencing factors on their population (the elicitation questionnaire can be found in Appendix A). The questionnaire was sent to 3 species experts after they viewed a webinar explaining the types of questions they would encounter and how their responses would be used. Collectively, the experts had many decades of experience working in the Suwannee River basin.

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

Because of the elusive nature of this aquatic turtle, estimating density or abundance is challenging. To obtain estimates of abundance, we used expert elicitation, using a 4-point elicitation procedure in a written questionnaire (Speirs-Bridge et al. 2010, p. 515). 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 *M. suwanniensis* in the Suwannee River basin. They were then asked to report how confident they were that their interval (lowest estimate to highest estimate) captured the actual number of *M. suwanniensis* (akin to a confidence interval). Finally, the experts were asked to describe how they generated their estimates.

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 where there is more structure (e.g. sunken logs, undercut banks), higher prey availability, 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 *M. suwanniensis* in the Suwannee River basin. Using the same 4-point elicitation format, we asked the species experts to estimate the extent of occupied area in the river basin where *M. suwanniensis* are exposed to each of the following threats: incidental hooking on trot and limb lines, commercial fishing bycatch, 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 the Suwannee River basin, as well as any conservation actions that are being implemented.

In addition to asking the expert team about the spatial extent of different threats in the Suwannee River basin, 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 5). We received usable responses from 2 experts, with varying numbers of questions answered by each. Due to the small number of species experts and variable answers received on the 4-point elicitation, we are providing the expert elicitation data received from the *M. temminckii* SSA to indicate the potential level of impacts the identified threats have on *M. suwanniensis*. *M. suwanniensis* comments were included in the evaluation of the 4-point elicitation data. 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.

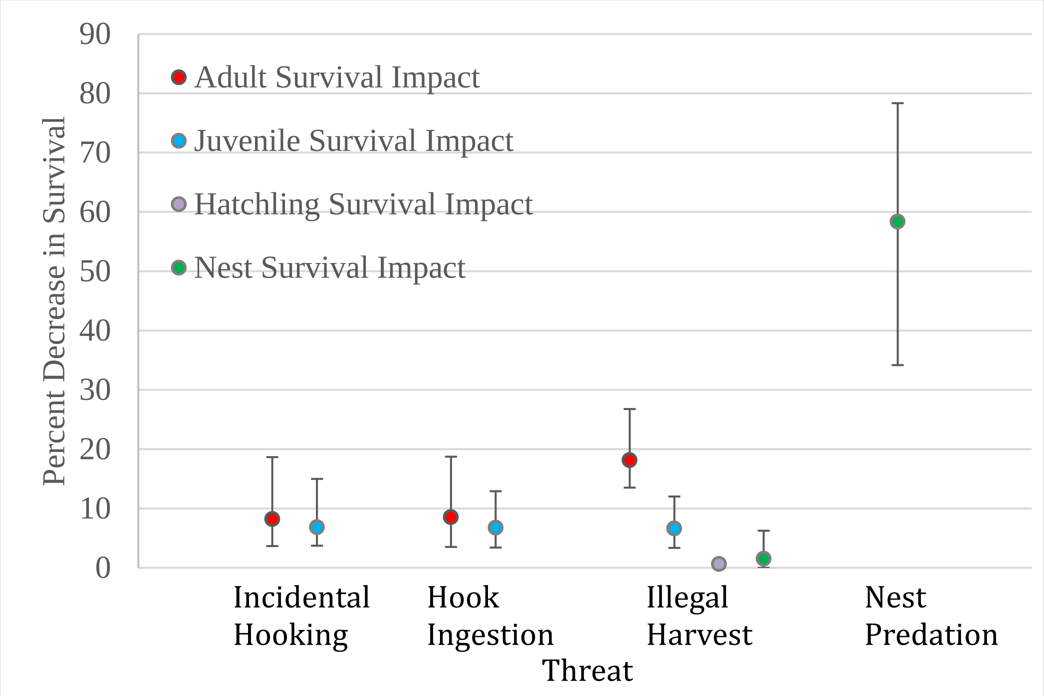


Figure 5. Expert-elicited magnitude of threats facing *Macrochelys spp*. in terms of the percent decrease to survival rates. Error bars indicate the average of lowest likely and highest likely estimates of impacts on survival, while circles indicate the average of most likely impacts on survival. The 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 *M. suwanniensis* by querying state biologists or those with access to the state's natural heritage program data. For each county in their state, we asked for the current and historical status, and the date of the last confirmed record of *M. suwanniensis*. 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), *M. suwanniensis* occupancy was classified as either occupied, not occupied, or unknown (Table 4). Respondents were also asked to describe, if known, the reasons behind any changes in occupancy status from historical to current.

Table 4. Definitions of Occupied, Not Occupied, and Unknown, for characterizing the current (since 2000) and historical (prior to 2000) distribution of *M. suwanniensis* by county.

Occupancy	**Current**	**Historical**
Occupied	Signifies that *M. suwanniensis* are known or presumed to occur in this county now. In the absence of very recent records, currently occupied counties will include those with *M. suwanniensis* records since 2000, provided that there is no evidence that the species has been extirpated since those most recent records	Signifies that *M. suwanniensis* are known or believed to have occurred in this county prior to 2000
Not Occupied	Signifies that *M. suwanniensis* 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 *M. suwanniensis* occurred in this county prior to 2000
Unknown	Signifies uncertainty in the current occupation of this county by *M. suwanniensis*. 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 *M. suwanniensis*

4.6 Current Condition Results

This section reports the current abundance, current threats and conservation actions, and comparison with the historical range for *M. suwanniensis*. All the information came from expert elicitations for *Macrochelys spp*. unless otherwise specified.

The Suwannee River basin (Figure 1) encompasses parts of southern Georgia and northern Florida. Main water bodies that currently or historically supported *M. suwanniensis* include the Suwannee River, Santa Fe River, New River, Alapaha River, Little River, and Withlacoochee River. The Suwannee River experiences longitudinal changes in water chemistry from the low-nutrient acidic blackwater at the head to the saline delta (Ceryak et al. 1983, p. 46). Tidal variation is particularly evident during low-flow condition and can extend to 43 km (26.7 miles) upstream from the mouth. Woody debris, undercut banks and large rocks found throughout the river are important habitat during low water levels (Enge et al. 2014, p. 10).

Current Abundance

Current abundance of *M. suwanniensis* in the Suwannee River basin is estimated to be 2,000 individuals, with 50% confidence that the true abundance is between 500 and 5,000. These estimates were based on extrapolating localized experience to the entire basin (two experts responding).

Current Threats and Conservation Actions

Threats in the Suwannee River basin include:

- **Incidental hooking**, which is estimated to affect 50% of the species' range in the Suwannee River basin (average bounds between 30 and 75%, average 55% expert confidence that the true value lies within their specified bounds). The expert elicitation data indicates a reduction of juvenile and adult survival by 6 – 9%. This is similar to Steen and Robinson's (2017) determination of 3 – 11% chance of mortality from hook ingestion.

- **Illegal harvest**, which is estimated to affect 32.5% of the species' range in the Suwannee River basin (average bounds between 20.5% and 55%, with an average expert confidence of 55% that the true value lies within their specified bounds)

- **Nest predation**, which is estimated to affect 7% of the species' range in the Suwannee River basin (60% confidence that the true value lies between 5 and 10%)

- **Habitat alteration** from woody debris removal was also identified as a threat to individuals in this population. Florida allows deadhead logging with the proper permits from Florida Department of Environmental Protection and Georgia is not currently processing permits.

M. suwanniensis are listed in Florida and Georgia as threatened. *Macrochelys spp*., is listed as an Appendix III species under CITES which requires annual reporting for all exported individuals.

Comparison with Historical Range

Recent surveys have confirmed minimal change in the known historical range of *M. suwanniensis* (Figure 6). Areas in Georgia with historical and current unknowns/ not occupied occur in the upper Alapaha and Suwannee Rivers. The upper Alapaha River is shallow and difficult to access. Turtles may occupy this stretch of the river, but are difficult to trap and/ or find during visual surveys. Multiple surveys in the upper Suwannee River, near the outflow of the Okefenokee Swamp, have failed to capture *M. suwanniensis*, leading biologists to suggest that they may occur at such a low density that the turtles are very difficult to detect (Jensen and Birkhead 2003, p. 32).

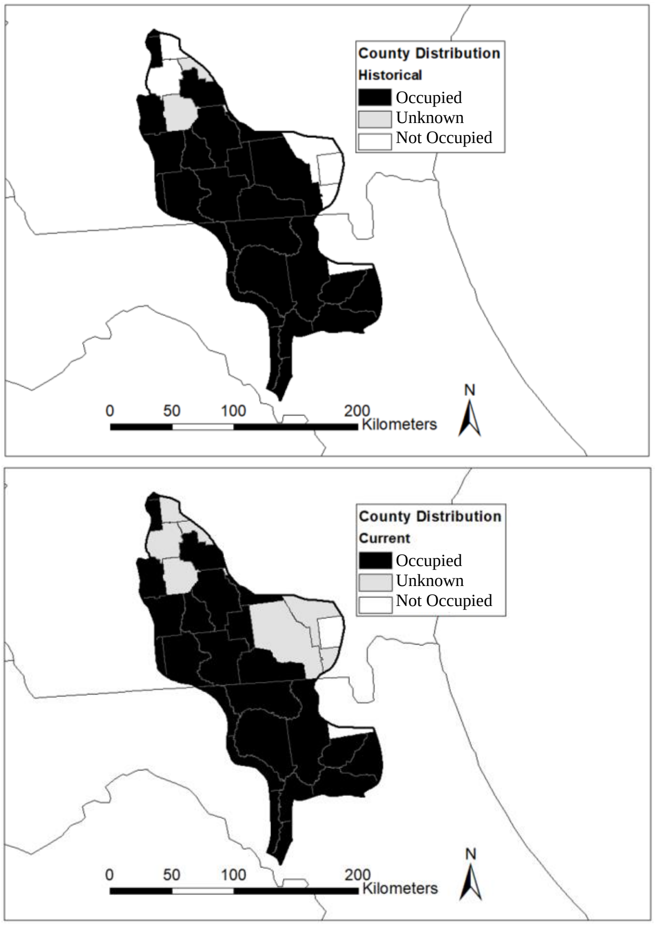


Figure 6. Historical and current distribution of *M. suwanniensis* in the Suwannee River basin.

4.7 Current Condition Overall Results

In this section, we summarize the above results to describe the current resilience, redundancy, and representation for *M. suwanniensis*.

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.

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. Population depletions occurred 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 Florida and Georgia. There are limited data available describing how populations have responded to reduced harvest pressure.

Estimated abundance of *M. suwanniensis* in the Suwannee River basin is estimated to be 2,000 alligator snapping turtles, with a true abundance is between 500 and 5,000 bars show higher confidence of species experts in their estimates.

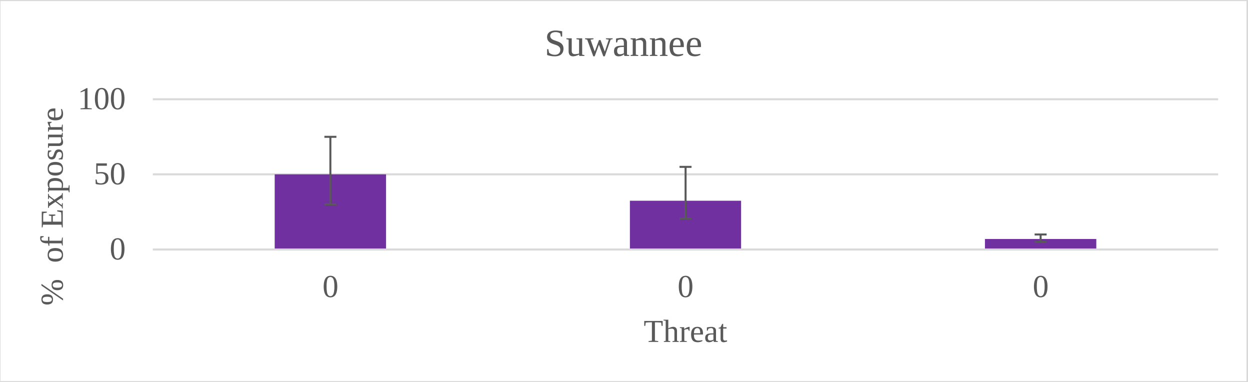


Figure 7. Percent of *M. suwanniensis* exposed to expert elicitation threats within the Suwannee River basin.

Figure 7 displays the spatial extent of threats to *M. suwanniensis* in the basin. These threats were identified during the expert elicitation process and are described below with their percent ranges:

- Incidental hooking (i.e. recreational trot and limb lines, fishing tackle, etc), with estimates of the percentage of turtles exposed to the threat ranging from 30% to 75%,
- Illegal harvest or poaching across the basin ranges from 20% to 55%. Illegal harvest may occur for human consumption, pet trade, or international export, and
- Nest predation by native and exotic species that varies from 5% to 10%.

Although Florida and Georgia have regulations governing recreational trot and limb lines, some fishermen choose not to check their lines once every 24 hours and/ or do not mark their lines with the harvester's name and address while being used or possessed in or upon the waters. In Florida, harvesters are limited to 25 hooks. Anything over 25 hooks is considered commercial fishing. In theory, if harvesters do not mark their hooks, they could have many more than 25 hooks in the waters without having to get the necessary state permits and law enforcement would have difficulty making a case against the harvesters. Florida regulations do allow for areas to be removed from lawful use of trotlines, bush hooks, and/ or set lines. This would reduce incidental hooking related deaths.

Both states could increase their law enforcement efforts to reduce illegal harvest, poaching, and export of *M. suwanniensis* and other freshwater turtles. In addition, the federal government can propose an amendment to have *Macrochelys* spp. moved from

Appendix III to Appendix I or II in CITES. This would restrict all export of *M. suwanniensis*.

Nest protection measures to minimize impacts from predators include:

- Predator exclusion devices with self-releasing opening for hatchling turtles,
- Predator trapping/ removal, and
- Fire ant insecticide applications in nesting areas.

4.7.2 Current Representation

Representation refers to the breadth of diversity within and among population(s) of a species, which allow it to adapt to changing environmental conditions. Even though *M. suwanniensis* is reported to have minimal loss in historical distribution, it is a single basin endemic species. Thus, the concept of representation and representative units do not apply to our analysis of this species.

4.7.3 Current Redundancy

Redundancy refers to the number and distribution of resilient population(s) across a species' range, which provides protection for the species against catastrophic events that impact entire populations. Within the core of their range however, *M. suwanniensis* 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 and hook ingestion from recreational fishing. The current and historical status of each county within the *M. suwanniensis* range, can be found in Appendix B. We assessed the entire basin as a single population and thus redundancy is not applicable to our analysis of this species.

This concludes the assessment of the current condition of *M. suwanniensis* across its 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 resilience, representation, and redundancy across the range of the species, we used a tiered approach of life history and ecology.

Current Resilience

To assess the current condition of *M. suwanniensis*, information was gathered from species experts about current abundance (our measure of resilience), current threats, and a comparison of the current and historical distribution. The Suwannee River basin has an estimated 2,000 individuals with an estimated low of 500 and high of 5,000.

M. suwanniensis 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%. 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. Estimates of the extent of nest predation are estimated to be low for *M. suwanniensis*.

Resilience increases with abundance and density; where there are more individuals, populations will have a greater ability to withstand stochastic demographic and environmental events. Therefore, as a result of the historical and ongoing threats, as described above, the overall current conditions (resiliency) encompasses a single population with an estimated abundance of 2,000 turtles across most of its historical range in Georgia and Florida. This will serve as useful baseline conditions against which to compare future resilience in the next chapter of this SSA.

Current Representation

The concept of representation and representative units do not apply to our analysis of this species.

Current Redundancy

We assessed the entire basin as a single population and thus redundancy is not applicable to our analysis of this species.

CHAPTER 5 – FUTURE CONDITIONS

In this chapter, we describe the methods used to project *M. suwanniensis* population 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 C.

5.1 Future Projection Model

We constructed a female-only, stage-structured matrix population model (Caswell 2001, p. 33) to project *M. suwanniensis* population dynamics over annual time steps for 50 years for the Suwannee River basin. We based our model on the peer reviewed and published model in Folt et al. (2016, p. 24) and updated the model to reflect new guidance on 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 *Macrochelys spp.* life cycle (Figure 8) 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 is a standard time frame for models of this type, and patterns in the output were apparent within less than 50 years (i.e., no additional information was gained by running the model for longer).

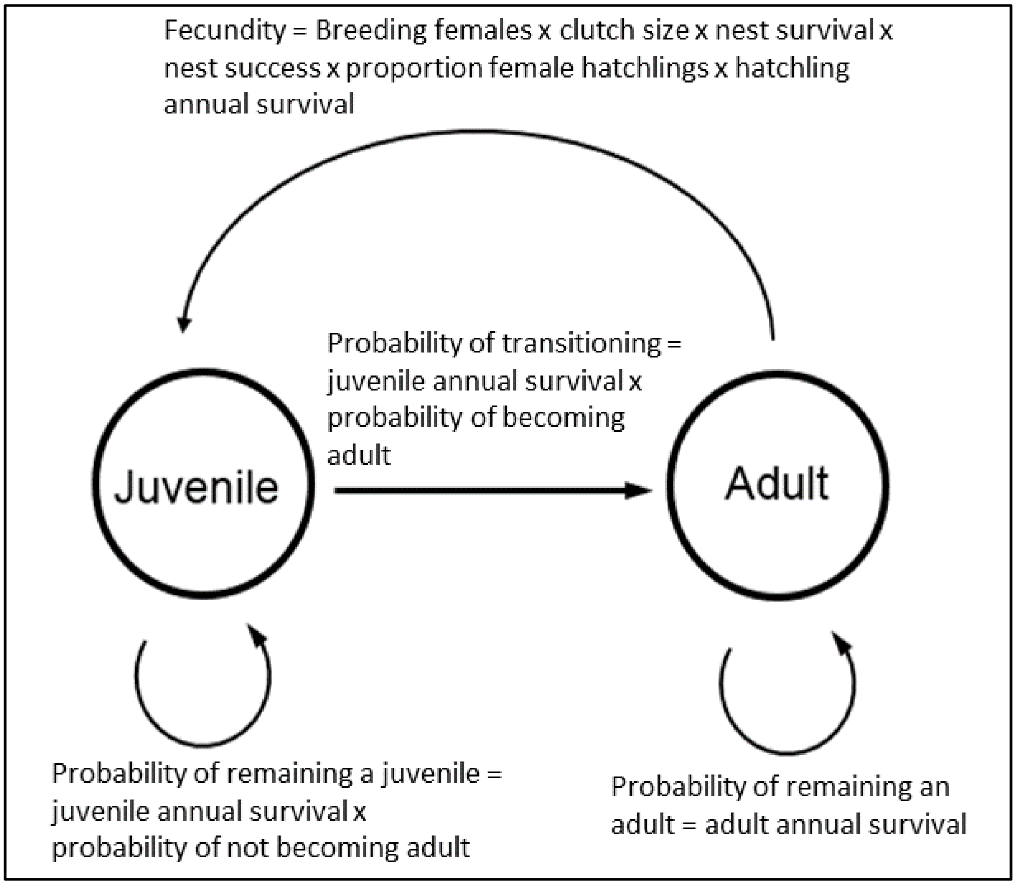


Figure 8. *Macrochelys spp.* life cycle diagram for a female only two-stage pre-breeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals). At each time step, juveniles can remain in their current stage, which is the product 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 *Macrochelys spp.* 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 *Macrochelys spp.* or the closely related *C. serpentina*, with information gaps filled in using expert elicitation (further details about how values were derived in Appendix C). 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 *Macrochelys spp*. 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)^{C}	0.860	0.02772	0.010532	Folt et al. 2016	Spring Creek, Georgia
Juvenile survival Northern Mississippi - East Unit^{C}	0.730	0.03542	0.010822	Dreslik et al. 2017	Illinois
Juvenile to adult transition probability	0.020	0.01112	0.008892	Tucker and Sloan 1997	Louisiana

Adult survival	0.950	0.01742	0.009692	Folt et al. 2016	Spring Creek, Georgia
Proportion of females that breed annually	0.980	0.01122	0.008942	Dobie 1971	Southern Louisiana
Clutch Size	33.200	10.00002	5.000002	Weighted average [b]; Folt et al. 2016 (SD)	Multiple
Nest survival	0.130	0.02692	0.010372	Ewert et al. 2006	Lower Apalachicola River, Florida
Nest success	0.723	0.03582	0.010972	Ewert et al. 2006	Lower Apalachicola River, Florida
Proportion of female hatchlings	0.500	0.04002	0.010902	Expert opinion	–
Hatchling survival to one year	0.150	0.02852	0.010602	Expert opinion	–

[a] Demographic parameter mean, sampling variance, and process variance values apply to all modeled *M. temminckii* SSA analysis units and the Suwannee River basin except for juvenile survival (φJ), which used different values for the *M. temminckii* SSA 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 C2.

[C] From the 2019 alligator snapping turtle (*M. temminckii*) SSA.

SD = standard deviation

Table 6. Threat-specific percent reductions (mean ± standard deviation) to *Macrochelys spp.* survival parameters, derived from remote expert elicitation among a team of taxon experts. These quantities were assumed to remain constant across the *Macrochelys spp.* range. 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	–

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 C).

During the expert elicitation process, we asked all participants to provide an estimate of total abundance within their geographical area(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 the Suwannee River basin assuming a stable stage distribution. Projections were modeled 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, scenario, and iteration combination (See Appendix C for more details).

5.1.2 Model Scenarios

We projected future conditions for *M. suwanniensis* under five different scenarios, across which the levels of threats and conservation actions varied. Species experts identified five 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), illegal collection (i.e., poaching; influenced hatchling, juvenile, and adult survival), and subsidized nest predators (influenced nest survival).

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 geographical area of expertise. Thus, reductions to survival rates attributed to each threat were assumed to be the same across the Suwannee River basin. 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 *M. suwanniensis* encounters that threat would vary within the basin. 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 the basin.

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 were nest predation and legal collection, 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 basin 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 habitat threats include but are not limited to: increased enforcement of state laws or law enforcement presence to reduce poaching or bycatch on illegally set trot or limb lines, prohibit recreational fishing or certain gear (e.g., trotlines, hoopnets) in the Suwanee River basin, and management actions that reduce the densities of nest predators. In addition to habitat modification, long term female population augmentation can be implementing by head-starting and captive breeding programs by federal, state, and non-governmental organizations. 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 *M. suwanniensis*. 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: *Increase 25%* ● Spatial extent of threats: *Expert-elicited*	***"Increased Threats + "*** ● Impact of threats: *Increased 25%* ● Spatial extent of threats: *Reduced 25%*

The effect of habitat loss was incorporated into the adult fecundity element in the transition matrix where its effect depended on total abundance.

Habitat Loss

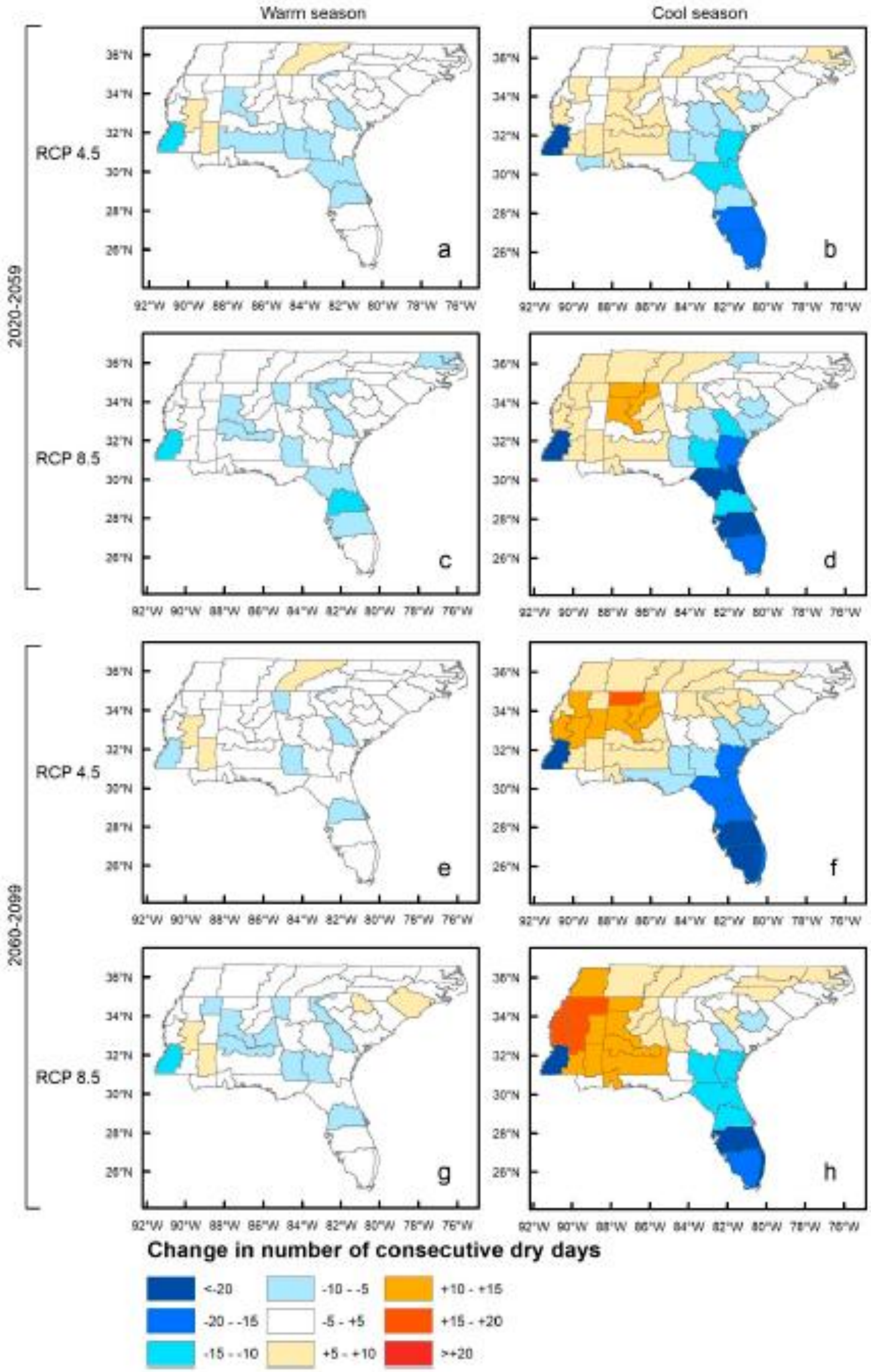
We asked the species expert team to list habitat loss mechanisms within geographical area of expertise. After adjusting for linguistic differences among responses (e.g., "desnagging" and "removal of large woody debris" were two answers that reflected the same mechanism), we summarized the number of unique habitat loss mechanisms within the Suwannee River basin and calculated the mean across experts. We imposed a population ceiling (i.e., carrying capacity) that was annually reduced by a habitat loss rate that equaled the mean number of unique threats in the basin, divided by 100. The initial population ceiling was determined based on the summarized expert elicitation values for the maximum possible number of *M. suwanniensis* currently within the Suwannee River basin, after adjusting for sex ratios and presence of hatchlings in the estimate. Thus, the population ceiling 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 *M. suwanniensis* 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 C Figure 13).

Climate Change

An increase in variation of temperature, consecutive dry days (CDD), and sea level rise may affect *M. suwanniensis* by altering the sex ratio of hatchlings and altering available habitat along the coast. Estimates of the effects of climate change using available climate models typically lack the geographic precision needed to predict the magnitude of effects at a scale small enough to discretely apply to the range of a given species. However, data on recent trends and predicted changes for southeastern United States (Keellings and Engstrom, 2019, entire and National Oceanic and Atmospheric Administration, 2019, n.p.) provide some insight for evaluating the potential impacts of climate change to *M. suwanniensis*.

Keellings and Engstrom (2019, pg. 2) used thirty-two localized constructed analog downscaled spatial resolution Global Climate Models to run near future (2020-2059) and future (2060-2099) warm/ cold season temperature and CDD models for the southeastern region of the United States. Warm season is defined as April – October and cold season is November – March. Two representative concentration pathways (RCPs) were used to represent different greenhouse gas (GHG) emissions trajectories: rapid GHG emission growth RCP 8.5 (8.5 W/m^2 radiative forcing, -1370 ppm CO_2) and medium GHG emission growth with stabilization RCP 4.5 (4.5 W/m^2 radiative forcing, -650 ppm CO_2). The CDD models analyze meteorological drought even though drought may be multifaceted within the Suwannee River basin (i.e., potable and agricultural water withdrawal and water lose through evaporation). The two models encompass the 50-year annual time step female-only, stage-structured matrix population model. The observed (1950-2005) lengths of 20 – year return period of CDD for warm season is 60 – 70 days and cold season is 60 – 90 days. Results of the near future (2020-2059) model show the greatest reduction (< - 20 days) in CDD occur in the cold season with GHG emission of 8.5 and the greatest reduction in CDD in the future (2060-2099) occur in the cold season with GHG emission of 4.5 (< -20 to -16 [Figure 9]). These models indicate more frequent rain events during the cold season in the future but do not model annual precipitation or the impacts of human demands. The more frequent cold season rains may provide the aquifers feeding the Suwannee River basin much needed water to offset irrigation withdrawals. This would help in maintaining aquatic habitat and food sources for *M. suwanniensis*.

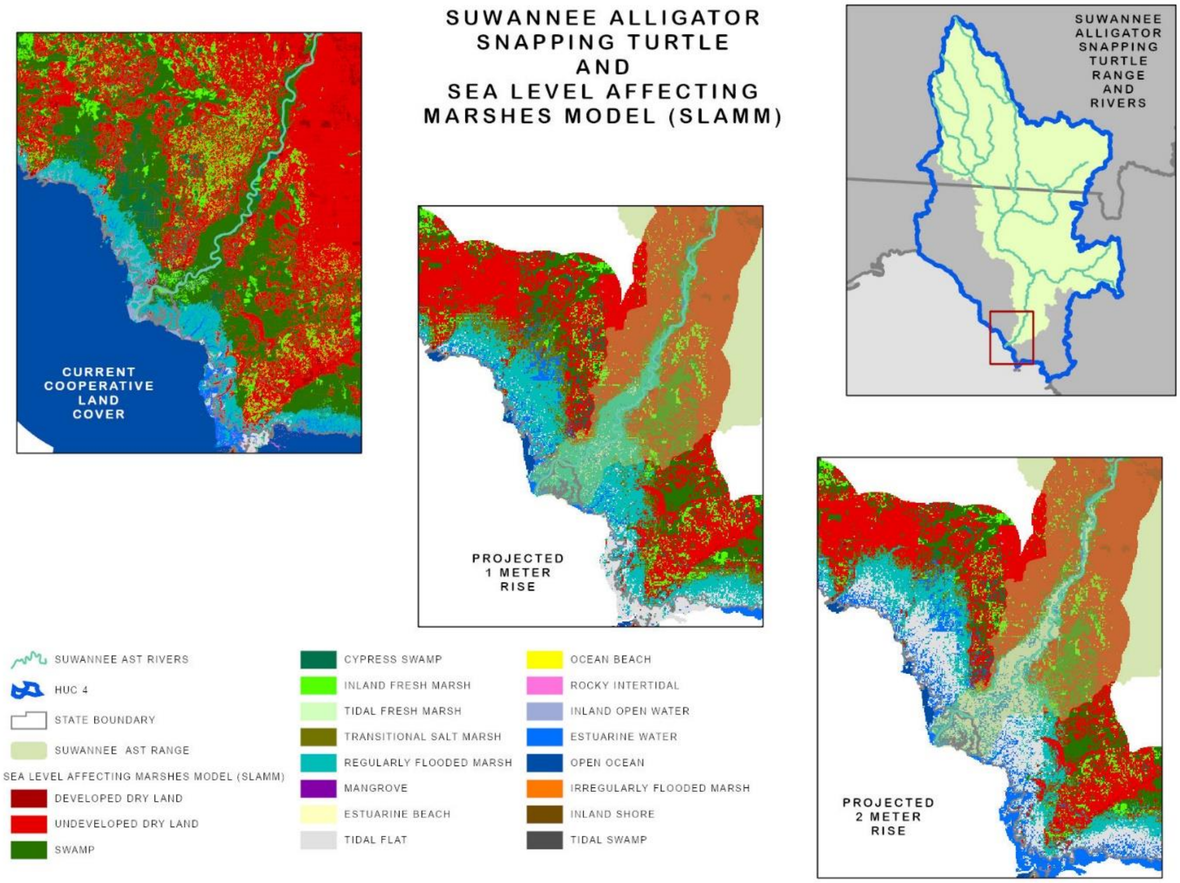
Figure 9. Keellings and Engstrom (2019, pg. 6) absolute changes in length of 20-year return period consecutive dry days in modeled future periods versus observation.



The Sea Level Affecting Marshes Model (SLAMM) was used to simulate the change of habitat from dry to at or below the mean high water line (MHWL) within the lower

Suwannee River basin. Analysis area is outlined in red on the map located in the upper righthand corner of Figure 10. Within the focal area, there is approximately 11,946 acres (4,834 hectares) or 0.19% of the area at or below the MHWL. With a 1 meter rise in sea level the amount of area that is at or below the MHWL grows to approximately 18,011 acres (7,288 hectares) or 0.29% of the area and with a 2 meter rise, approximately 28,360 acres (11,476 hectares) or 0.45% of habitat will change from dry to wet. Even with the projected 2 meter sea level rise, there should be adequate dry habitat for nesting and aquatic habitat for each of *M. suwanniensis'* life stages.

Figure 10. Change of habitat from dry to at or below the mean high water line (at 1 and 2 meter rise) using SLAMM for the Suwannee River coastal area.



We have no information leading us to believe *M. suwanniensis* is being impacted by the current level of consecutive dry days, sea level rise, or that they will not be able to make moderate changes through time. Therefore, we did not include climate change impacts into the stage-structured matrix population model.

5.1.3 Model Structure Summary and Limitations

Values for *M. suwanniensis* initial abundances, demographic parameters, threats, and conservation measures were acquired from literature and expert elicitation, as well as measures of error or uncertainty that were also incorporated into the stochastic model structure. For the Suwannee River basin, 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, 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 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 *M. suwanniensis*.

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 *M. suwanniensis*, 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 the basin. 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 *M. suwanniensis* biology, demography, response to and prevalence of threats, and how these vary across the range of the species.

5.2 Future Modeling Results

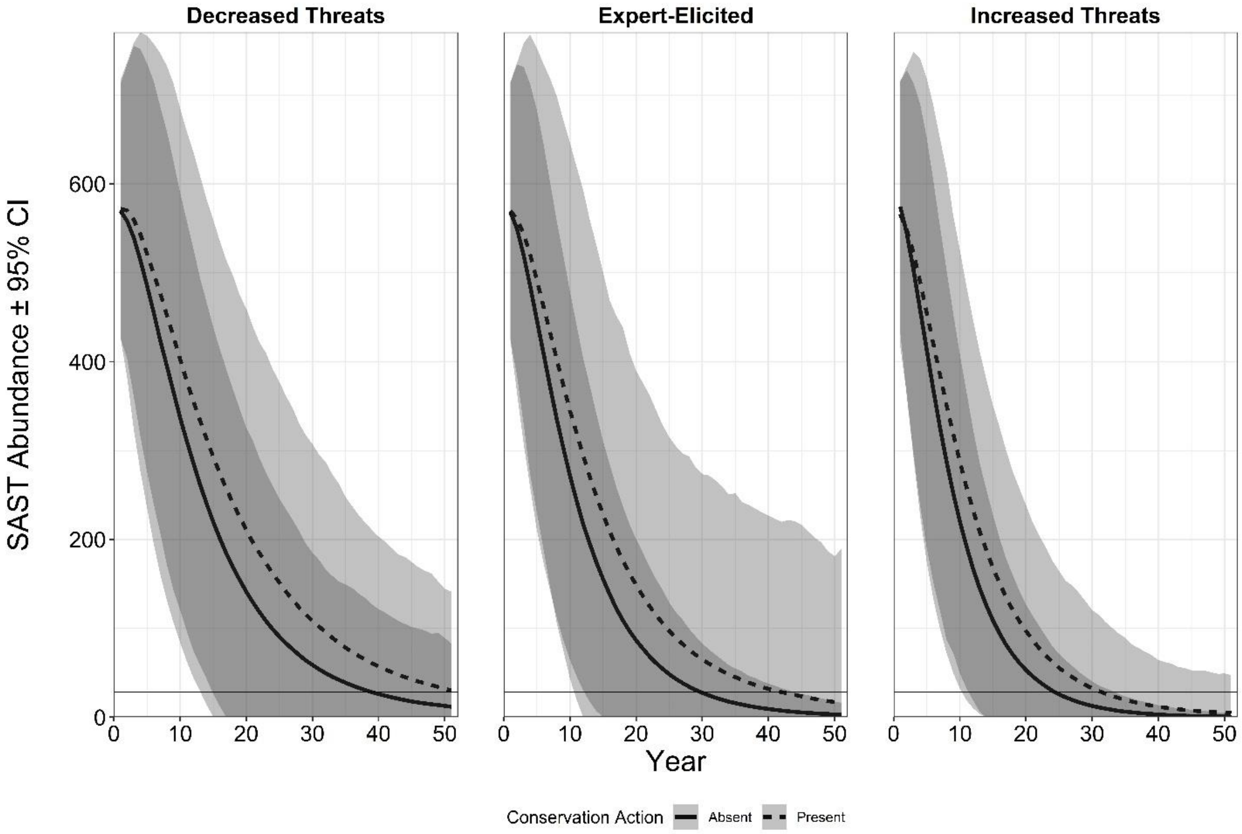
We derived a series of summary statistics to evaluate *M. suwanniensis* trends in abundance and evaluate potential variation and alternate scenarios. Here we define an extinction event as the total population (juveniles + adults) declining to zero individuals, whereas a decline to less than 5% of the starting population size was considered quasi-extinction. For each scenario combination, we estimated extinction and quasi- extinction 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 C). For the iterations in which abundance reached extinction or quasi-extinction, 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 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 extinction or quasi- extinction 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, we highlight the time to extinction or quasi- extinction only for those scenarios where extinction or quasi-extinction 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

M. suwanniensis abundances were predicted to decline over the next 50 years in all scenarios (Figure 11). Predicted declines were more rapid the higher the threat level (mean λ = 0.90, 0.87, and 0.85 for Decreased Threat, Expert-Elicited Threat, and Increased Threat scenarios, respectively, and mean λ = 0.92, 0.90, and 0.88 respectively for Decreased Threat +, Expert-Elicited Threat +, and Increased Threat + scenarios; Appendix C Table C5). Compared to initial abundances, mean abundance in the first five to ten years was predicted to increase modestly by 7-10% in the decreased threats scenarios, decline by 4-5% under expert-elicited threats scenarios, and decline by 17-21% under increased threats scenarios. Declines in abundance were predicted to be more drastic by halfway through the simulation; after 25 years mean abundance was predicted to decline by 67-69% under decreased threats scenarios, 78-80% under expert-elicited threats scenarios, and 87-89% under increased threats scenarios, with the lower bounds of these ranges from the scenarios with conservation actions and the upper bound from the scenario without conservation actions for each threat level. By the end of the 50-year simulation, abundances were predicted to decline 95-100% across all six scenarios compared to initial abundances (See Appendix D for mean abundances at each time step).

Figure 11. Simulated *M. suwanniensis* total abundance (females only) over a 50-year period. The lines in each panel depict the mean abundance trajectory across 500 stochastic simulations for each of six scenarios and the shaded areas reflect the 95% confidence intervals (CI). The three panels indicate the scenario's threat level: decreased (DE), expert-elicited (EE), or increased (IN). The lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the quasi-extinction threshold (<5% of total abundance in Year 1) is given by thin flat line.



Though abundance declined in all scenarios, the species was very likely to become quasi-extinct in the basin within 50 years in all scenarios where conservation actions were absent and when conservation measures were present in the expert-elicited and increased threat levels. The species was likely to become quasi-extinct when the threat levels were decreased and conservation measures are present and to become extinct when the threat levels are increased and conservation measures are absent. The species is about as likely as not to become extinct when the expert-elicited threat level does not have conservation measures present. The species is very unlikely to become extinct then the expert-elicited threat level coincides with conservation measures and when the threat levels are decreased with and without conservation measures (Table 9). While the species was not predicted with high likelihood to become completely extinct in the basin within 50 years, quasi- extinction was very likely to occur in both decreased threats scenarios (after an average of 35-40 years), very likely to occur in both expert-elicited scenarios (after an average of 28-35 years), and virtually certain in both increased threats scenarios (after an average of 24-30 years).

Table 9. Probability and time to extinction and quasi- extinction for *M. suwanniensis*. 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 extinction (Prob Ext) and quasi- extinction (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 abundance, respectively. For only those

replicates in which the population reached extinction or quasi- extinction, 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. The asterisk (*) indicates that only one simulation replicate went extinct within a scenario, meaning that a standard deviation for Time to Ext could not be calculated.

Threat Level	Prob Ext		Time to Ext		Prob Q-Ext		Time to Q-Ext	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.07	0.00	48.39 ± 2.71 (40, 51)	49.0 ± * (49, 49)	0.94	0.72	34.56 ± 6.61 (19, 51)	39.57 ± 6.45 (24, 51)
Expert-Elicited	0.40	0.06	46.16 ± 3.56 (35, 51)	48.28 ± 2.42 (42, 51)	0.99	0.92	28.48 ± 5.70 (16, 51)	34.51 ± 6.49 (20, 51)
Increased	0.81	0.30	43.21 ± 4.27 (31, 51)	47.09 ± 3.20 (36, 51)	0.99	0.98	23.85 ± 4.48 (15, 49)	29.50 ± 5.52 (17, 51)

5.3 **Future Condition Results**

In this section we summarize the results from above to describe the future resilience, redundancy, and representation for *M. suwanniensis*.

5.3.1 Future Resilience

Resilience refers to the ability of a population to withstand stochastic disturbances (e.g. demographic, environmental stochasticity, etc.). Abundance is central to resilience, as a small population may be more vulnerable to perturbations than a larger population. We modeled scenarios that reflected uncertainty in the impact of threats on *M. suwanniensis* demography and all threat levels (decreased, expert-elicited, and increased) produced mean growth rates (λ) indicating population decline. Time to quasi- extinction varied across scenarios, but in general, the Suwannee River basin is likely to reach this in 32 – 42 years (Table 9).

After 50 years, the mean female abundance was not predicted to exceed 30 individuals in any scenario (Table 10). 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 (Table 11).

Table 10. Initial and final (after 50 years) projected female-only abundances for *M. suwanniensis* under Decreased, Expert-Elicited, and Increased (IN) threats scenarios with and without conservation measures. 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. Quasi-extinction occurs when the female-only population decreases to 28 individuals.

Threat Level Scenario	Mean Beginning Abundance	Mean Abundance at Year 51	Quasi-Extinction
Decreased	572	12	28
Decreased w/ Conservation		30	28
Expert-Elicited	574	3	28
Expert-Elicited w/ Conservation		17	28
Increased	566	1	28
Increased w/ Conservation		5	28

Table 11. Initial and final projected *M. suwanniensis* abundances expressed as raw abundances and scaled to 1,000 hectares of open water in the basin. 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 from a probability distribution that incorporated uncertainty surrounding current abundance, and b) included females only, while current condition abundances included males and females.

Area	**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**
Suwannee	570	21.7	183	7.0	21	0.80

Resilience is expected to decline in the Suwannee River basin under all scenarios. Abundance is predicted to decline by more than 95%, resulting in drastically lowered abilities of *M. suwanniensis* to withstand stochastic events, if *M. suwanniensis* persist at all.

Future redundancy and representation for *M. suwanniensis* is expected to decline drastically over the next 50 years (Table 11). Where *M. suwanniensis* persist in the future, they are predicted to be rare and not found in resilient groupings. The addition of conservation actions, or different assumptions about the impact of threats on *M.*

suwanniensis demography may alter the time to quasi- extinction by about a decade at most, typically less. No scenarios resulted in stable or increasing population within the Suwannee River basin.

5.4 Summary of Future Conditions and Viability

For *M. suwanniensis* to maintain viability, it needs to have resiliency throughout its range that is 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 population needs 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 is expected to decline drastically across the Suwannee River basin under all scenarios. We modeled scenarios that reflected uncertainty in the impact of threats on *M. suwanniensis* demography, and all scenarios produced mean growth rates indicating population decline. Predicted abundances were likely to virtually certain (i.e., had a >66% probability, of dropping below 5% of current abundances within 50 years). *M. suwanniensis* will likely become very rare or disappear from the Suwannee River basin in 32-42 years.

This concludes our assessment of *M. suwanniensis* needs, current condition, and future condition. It is apparent that based on the current state of knowledge, *M. suwanniensis* 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 - 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 B - Current and Historical Range by State and County

By state, *M. suwanniensis* was historically and is still found in 2 states: Florida and Georgia.

Table B1. Current and historical occupied status for counties within the *M. suwanniensis* 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 *M. suwanniensis* are not shown.

State-County	Current	Historical	Last Record	Notes
FL-Alachua	Occupied	Occupied	2012	
FL-Baker	Unknown	Unknown	2018	
FL-Bradford	Occupied	Occupied	2011	
FL-Columbia	Occupied	Occupied	2012	
FL-Dixie	Occupied	Occupied	2014	
FL-Gilchrist	Occupied	Occupied	2014	
FL-Hamilton	Occupied	Occupied	2017	
FL-Jefferson	Unknown	Unknown	-	2000 newspaper photo purportedly from Aucilla River, unsuccessful trapping - likely distribution gap
FL-Lafayette	Occupied	Occupied	2014	
FL-Levy	Occupied	Occupied	2014	
FL-Madison	Occupied	Occupied	2012	
FL-Marion	Unknown	Not Occupied	-	2 museum records from Ocklawaha River in 1916 & 1955, species isn't thought to occur in St. Johns River drainage, may be introduced here
FL-Suwannee	Occupied	Occupied	2014	
FL-Union	Occupied	Occupied	2011	
FL-Taylor	Unknown	Occupied	2018	
GA-Atkinson	Occupied	Occupied	2018	
GA-Ben Hill	Unknown	Unknown	-	Upper Alapaha River
GA-Berrien	Occupied	Occupied	2018	
GA-Brooks	Occupied	Occupied	2018	
GA-Chariton	Unknown	Unknown	-	Upper Suwannee River - Okefenokee Swamp
GA-Clinch	Unknown	Occupied	-	
GA-Colquitt	Occupied	Occupied	2018	
GA-Cook	Occupied	Occupied	1998	
GA-Coffee	Unknown	Occupied	2019	
GA-Crisp	Occupied	Occupied	1989	
GA-Echols	Occupied	Occupied	2018	
GA-Irwin	Occupied	Occupied	2017	
GA-Lanier	Occupied	Occupied	1997	
GA-Lowndes	Occupied	Occupied	2018	
GA-Thomas	Occupied	Occupied	2006	
GA-Tift	Unknown	Unknown	-	Upper Alapaha River
GA-Turner	Unknown	Not Occupied	-	Upper Alapaha River

State-County	Current	Historical	Last Record	Notes
GA-Ware	Unknown	Not Occupied	-	Upper Suwannee River - Okefenokee Swamp
GA-Wilcox	Unknown	Not Occupied	-	Upper Alapaha River
GA-Worth	Occupied	Occupied	2014	

APPENDIX C - Future Condition Model Methods and Results

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OVERVIEW

Here we describe the analytical framework used to evaluate the current condition of Suwannee alligator snapping turtle (*Macrochelys suwanniensis*; hereafter SAST) populations across their range. We constructed a female-only, stage-structured Lefkovitch matrix model to project SAST population dynamics over 50 annual timesteps. We used the best available data from the literature to parameterize the population 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. 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, 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 fully stochastic projection model that accounted for parametric uncertainty in the demographic parameters, to predict future conditions of the SAST in four of the seven 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 SAST population declined to less than 5% of the population size in year one of the simulation within an analysis unit.

METHODS

Modeling Framework Origin

The modeling framework used here was originally developed for the *M. temminckii* Species Status Assessment (SSA), which contained the Suwannee Analysis Unit, comprising the entire distribution of *M. suwanniensis*. Following the decision to conduct a separate SSA for *M. suwanniensis*, the results for the Suwannee analysis unit were removed from the *M. temminckii* (hereafter AST) modeling appendix and placed here. The model description for this document has been streamlined to only include portions that are relevant to the *M. suwanniensis* population in the Suwannee analysis unit. For example, the description of the legal harvest collection function, which did not affect the Suwannee analysis unit (which is closed to legal collection), has been removed.

Expert Elicitation

We relied on expert elicitation to fill information gaps needed to project SAST 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, 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 attempted to back calculate 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 SAST population dynamics over annual timesteps in the Suwannee analysis unit. We based our model off the peer reviewed and published model in Folt et al. (2016, p. 24) and updated the model to reflect new 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 SAST's life cycle (Fig. D1) 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 (Fig. 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 SAST and 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" value and more appropriate for perturbation analyses that seek to isolate the effects of threats and stressors. In broader AST modeling framework, though we created separate projection matrices ($\mathbf{A}_u$) for each analysis unit *u*, all demographic parameters used to calculate the matrix elements were the same across all seven units, with the exception of φ_J. The SAST modeling framework used the same baseline projection matrix as all analysis units within the AST SSA, with the exception of Northern Mississippi – East. 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).

$$\boldsymbol{A_u} = \begin{bmatrix} P_{J,u} & F_A \\ G_{J,u} & P_A \end{bmatrix}$$

Here the subscript *u* is used to indicate differences in the projection matrices among analysis units in the broader framework, but is used sparingly hereafter due to the exclusive focus on the Suwannee analysis unit.

We incorporated stochasticity into our modeling framework by modeling each demographic parameter (summarized in Table C1) 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. No demographic parameter estimates for SAST were available, therefore, all the parameters reported hereafter are from AST (*M. temminckii*) unless otherwise stated. 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) 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.

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) in the broader AST modeling framework, 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 for AST reported by Folt et al. (2016, p. 26; μ: 0.95, σ: 0.017). 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 C2). Therefore, we constructed a weighted mean of clutch sizes reported across the AST's range (Table C2), 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 2001, 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. (2016, 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 **A** 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 or SAST 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 C1) 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 **A** matrix elements had existing estimates of σ_T^2 reported in the literature, we opted to generate σ_S^2 and σ_P^2 variance components that adhered to the criteria above to ensure model stability (i.e., avoid sampling negative values from probability distributions) and to treat parameters in a consistent manner. It is a common practice in simulation modeling to apply a coefficient of variation function when empirical estimates of variance are not available. The above formulas are only suitable for proportional parameters, therefore we implemented the desired variance partitioning ratio for clutch size (*CS*) by setting σ_S and σ_P to 10 and 5, respectively. Our decision to partition σ_T^2 along a 2:3 ratio for σ_S^2 and σ_P^2 explicitly assumed that there is greater uncertainty in the true mean parameter value (i.e., parametric uncertainty) rather than the amount of annual variation, which is more conservative, given the dearth of AST demographic parameter estimates.

Following the framework described in McGowan et al. (2011, p. 1402), we used μ and σ_S to generate distributions of the overall mean and variance for each parameter. For the overall mean, we used beta distributions for all survival rates (φ_H, φ_J, φ_A, φ_N), the proportion of juveniles transitioning to adults (γ_J), *BP*, *NSC*, and *FP*— i.e., proportional parameters (μ.p)— whereas *CS* (a whole number) was sampled from a lognormal distribution. For each iteration *i* of the simulation loop, a mean ($\mu.p_i$, CS_i) and process standard deviation (σ_{Pi}) were drawn from the parameter's overall mean and variance distributions:

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

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

$$\sigma_{Pi} \sim \text{normal}(\sigma_P, \sigma_P \times 0.05)$$

in which α and β are the beta distribution parameters which describe the shape of the distribution bounded between 0 and 1.0, x_1 and x_2 are the shape and scale parameters of lognormal distribution, for the overall mean distributions. We used a normal distribution (above) for the overall variance, which was used to draw iteration-specific process (temporal) variances (σ_{Pi}^2) to determine the amount of temporal variation in each demographic parameter. We verified before beginning our analysis that the error term of the normal distribution was small enough to avoid generating negative values. The variance parameter of the normal distribution (i.e., the variance of the variance) was set to 5% of the theoretical maximum based on the mean sampling process deviation (σ_P), determined in Eq. 2 ($CV_P = 0.002$). Lastly, the iteration-specific means ($\mu.p_i$, CS_i) and standard deviations (σ_{Pi}) were then used to create iteration-specific distributions from which baseline parameter values were then drawn for each timestep *t* within iteration *i*:

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

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

This hierarchical simulation structure (i.e., using embedded loops to replicate parameter uncertainty and temporal variability) is widely applied in population viability modeling (McGowan et al. 2011, p. 1402).

Future Condition Scenarios

Incorporating threat effects. — The expert team identified five potential threats that were likely to reduce stage-specific survival probabilities in the Suwannee analysis unit (summarized in Table C3): 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), 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 C1) was based on a study in which 40 of 46 AST 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 demographic parameter attributed to each threat and the spatial extent of each threat within their analysis unit(s) of expertise. Thus, reductions to demographic 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 SAST encounters the threat would vary among analysis units.

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 current and 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 ω_a and present as reducing ω_a 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 ω_a by 25%, relative to the summarized expert elicitation quantities for $\theta_{a,p}$ and ω_a. The only exception to this structure is SNP, in which the expert-elicited $\theta_{SNP,p}$ and ω_{SNP}. values were used for all scenarios. Note that only the means for $\theta_{a,p}$ and ω_a, and not the standard deviations, were adjusted across the different scenarios. We then used the means and standard deviations for $\theta_{a,p}$ and ω_a 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 $\omega_{a,s,i,t}$ was drawn for each simulation *i* and timestep *t*:

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

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

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 C3) in the Suwannee analysis unit (Table C3). Therefore, φ_N has four possible threat combination-specific *c* survival values ($\varphi_{N,c}$): (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 C3). 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 ($\varphi'_{p,s,i,t}$), that was weighted according to the probability of the specific threat combination *c* occurring ($\delta_{p,c,s,i,t}$). We treated each threat that could potentially occur as an independent trial in which the threat was either present with probability ($\omega_{a,s,i,t}$) or absent ($1 - \omega_{a,s,i,t}$), 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 Suwannee analysis unit, the CLI only (#2) combination probability would be calculated as follows, using the spatial extent values in Table C3:

$$\delta_{CLI\ only} = \omega_{CLI} \times (1 - \omega_{SNP}) = 0.300 \times (1 - 0.070) = 0.279$$

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 large influence on the overall weighted nest survival estimate (φ'_N), due to the low threat combination probability value (0.279) nearly having a third of the possible weight (out of four possible scenarios). Thus, for *c* total threat combinations, the weighted average of survival parameter (φ') *p* in scenario *s* iteration *i* in year *t* is given by:

$$\phi'_{p,s,i,t} = \sum\nolimits_1^c \delta_{c,p,s,i,t\times}\phi_{p,c,s,i,t}) \qquad \text{(Eq. 4)}$$

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

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, all of the expert-elicited abundance estimates the Suwannee analysis unit 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, 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 (*IA*). We initialized the starting population for each analysis unit assuming that the population was in a stable stage distribution (**ssd**), the corresponding eigenvector of the dominant eigenvalue of the projection matrix **A**.

Next, we created a series of stochastic variables to generate stage-specific abundances at time t=1, that were unique to each scenario s, and iteration i combination. First, we converted *IA* to a Poisson-distributed stochastic variable ($N_{s,i}$) that was multiplied by an initial stage distribution ($\mathbf{isd}_{s,i}$) generated from a Dirichlet distribution to convert $N_{s,i}$ back to stage-specific abundances. We parameterized the Dirichlet distribution using the unit-specific stable stage distribution (**ssd**) multiplied by 10, to reduce the amount of variation.

$$N_{u,s,i} \sim \text{Poisson}(\text{IA}_{u,s,i})$$

$$\boldsymbol{isd}_{s,i} \sim Dirichlet(10 \times \overrightarrow{\boldsymbol{ssd}})$$

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 ($\mathbf{isr}_{j,s,i}$), 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.

$$\mathbf{isr}_{j,u,s,i} \sim \text{normal}(0.45, 0.45 \times (1-0.45) \times 0.20)$$

$$\begin{bmatrix} n_{J,s,i,1} \\ n_{A,s,i,1} \end{bmatrix} = N_{s,i} \times \boldsymbol{isd}_{s,i} \times \boldsymbol{isr}_{s,i}$$

Finally, we multiplied the three stochastic quantities to generate stage-specific initial abundances (t=1) for all scenario and iteration combinations ($n_{j,s,i,1}$).

Habitat Loss Function. — Habitat loss is believed to influence SAST demography by reducing the carrying capacity with the Suwannee analysis unit. Unlike the threat-specific parameter reductions, the effect of habitat loss was held constant across all future condition scenarios. 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 the Suwannee 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), which equaled the mean number of unique threats in the unit, divided by 100. The initial (i.e., t=1) population ceiling (PC_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:

$$PC_t = PC_1 \times (1 - \kappa)^t$$

(Eq. 5)

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 $\mathbf{A}_{s,i,s,t}$, which was then multiplied by the stage-specific abundance vector:

$$\begin{bmatrix} n_{Js,i,t+1} \\ n_{As,i,t+1} \end{bmatrix} = \begin{bmatrix} n_{Js,i,t} \\ n_{As,i,t} \end{bmatrix} \times \boldsymbol{A}_{s,i,t}$$

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.

Simulation Output Summary Statistics

Eigen Analysis. — For comparative purposes, we constructed a “deterministic” projection matrix (**D**) that used the baseline demographic parameter estimates in Table C1 to parameterize the matrix elements. The deterministic matrices were, thus, the closest approximation to reference (ideal) demographic conditions.
Additionally, 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 ($\mathbf{A}_{s,i,t}$) produced in the simulation framework, as well as the deterministic matrix.

Population Viability Analysis. — We derived a series of widely-used summary statistics to evaluate SAST population trends and viability, that could be compared across alternative future condition scenarios. In our simulations, we determined that an “extinction event” occurred if the total population (juveniles + adults) declined to zero individuals. Accordingly, extinction probability (p_{EX}) is the number of extinction events divided by the total number of iterations within the scenario (n=500). We then estimated the mean time to extinction (t_{EX}) for each scenario, by calculating the mean number years in which the population remained above zero across all iterations in which an extinction event occurred.

As common practice in population viability analysis, we also calculated the probability of quasi-extinction (p_{QX}) and time to quasi-extinction (t_{QX}) based on our pre-specified quasi-extinction threshold. For long-lived species such as the SAST, a functionally extinct population (e.g., one or two individuals) may persist for a long time before an extinction even occurs, thus, measures of quasi-extinction may be more informative for informing conservation decisions. We set our quasi-extinction threshold to 5% of the initial total population size (juveniles + adults in year t=1) to be consistent with existing Species Status Assessments for long-lived species (e.g., USFWS, 2015 p. 73), and to reflect the extensive uncertainty in SAST population dynamics, as our model does not account for changes in population dynamics that may occur when populations decline to very small numbers (Morris and Doak 2002, p. 43). 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 survival parameters ($\theta_{p,a}$) and their spatial extents (ω_a) within the Suwannee analysis unit are summarized in Table C3. Among the threats used in the model, the effect of SNP on φ_N was the largest overall reduction, followed by CLI on φ_A (Table C3). Within the Suwannee analysis unit, the combined BYR and HKI threats had the largest spatial extent (0.50 ± 0.12 SD), followed by CLI (0.30 ± 0.09), and SNP (0.07 ± 0.01) (Table C3). No Suwannee analysis unit experts reported commercial fishing bycatch (BYC) as a threat to the SAST population (i.e., spatial extent = 0.0). The mean number of habitat loss mechanisms identified by experts was 0.50 ± 0.71 SD.

SAST Population Viability

All future condition scenarios showed steep declines in abundance, that were mediated by conservation actions and reduced threat magnitude (Figs. C2–3). At the stage class level, all scenarios generally followed the same pattern. Adult abundance increased over the first ca. five years and then began to decrease, whereas juvenile abundance showed steep (relative to adults) declines over the entire time horizon (Fig. C3). The initial increase in adults likely reflects juveniles present at t=1 being recruited into the adult stage class, whereas the decline after year five indicates that an insufficient number of juveniles are surviving to recruitment into the adult stage class. The lack of juveniles could be attributed to low juvenile survival rates, low adult fecundity, or both. In the deterministic matrix (**D**), each adult female produced 0.23 juvenile females per year (F_A in Table C4), 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 (Table C3).

Projection matrix Eigen analysis. — The asymptotic population growth rates (lambda; λ) derived from the transition matrices (Table C5, Fig. C4) were consistent with the population declines we detected (Figs. C2–3), particularly in the absence of SAST releases. This is somewhat expected, given that the λs for the deterministic transition matrices, which reflect baseline demographic conditions in the absence of additional threat exposure implemented by the model, also indicated a population decline (**D**: 0.978; Table C5). As expected, mean λ was lower when conservation actions were absent, and declined with increasing threats (Table C5), ranging from 0.85 ± 0.04 SD (INTH) to 0.92 ± 0.04 (DETH+; overall: 0.89 ± 02 SD).

As consistent with many long-lived species, the elasticity analysis indicated that changes in adult survival (expressed via P_A) were most likely to produce changes in λ (Stearns 1992, entire). Matrix element elasticities followed the same pattern across all scenarios, in which P_A was the most elastic, followed by P_J, and G_J and F_A being equally inelastic

(Table C4). The relative elasticity of P_A varied negatively with threat level, and the absence of conservation actions. Interestingly, as survival rates were reduced in our analysis framework due to the increasing threat level (i.e., $\varphi_{LOTH} < \varphi_{MOTH}) < \varphi_{HITH}$), the elasticity of P_J, G_J, and F_A increased, while P_A elasticity decreased (Table C5).

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 (λ). In the deterministic matrices (**D**), juveniles comprised a larger proportion of the population than adults (Table C6). The proportion of adults in the SSD negatively varied with threat level (i.e., adults comprise the largest proportion of the population at decreased threat levels; Table C6), which is consistent with the elasticity analysis in that the elasticity of P_A is reduced with increasing threat levels (Table C5).

Extinction and Quasi-extinction Probability. — Probability of extinction (p_{EX}) ranged from 0.00 (DETH+) to 0.81 (INTH), with 0.27 ± 0.30 SD as the overall mean across all scenarios (Table C7). Time to extinction (t_{EX}) ranged from 49.0 years (DETH+, no SD) to 43.21 ± 4.27 years (INTH), with 47.02 ± 2.12 years as the overall average (Table C7). Probability of quasi-extinction (p_{QX}) ranged from 0.72 (DETH) to 0.99 (EETH, INTH), with 0.92 ± 0.10 SD as the overall mean across all scenarios (Table C7). Time to quasi-extinction (t_{QX}) ranged from 39.57 ± 6.45 years (DETH) to 23.85 ± 4.48 years (INTH), with 31.75 ± 5.56 years as the overall average (Table C7).

Synthesis

Drivers of SAST demographics. — The elasticity analysis indicated that under baseline conditions ("Deterministic" entries in Tables C4–6), conservation interventions to increase adult survival (contained in the P_A matrix element; Table C1) are likely to have the greatest proportional impact on SAST population trajectories (Table C4). Though all six of the future condition scenarios reduced the elasticity of P_A relative to the deterministic matrix (Table C4), P_A remained the most elastic parameter in all but two of the future condition scenarios (EETH, INTH). Under expert-elicited and increased threat conditions with conservation absent, the elasticity of P_J exceeded (INTH) or was equal to that of P_A (DETH), 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.

Within the stochastic simulation framework, we simulated conservation actions as a reduction in a threat's spatial extent (ω_a; bottom row in Table C3). Based on a comparison of survival rates for all stage classes and scenarios (Fig. C5), the conservation actions had increasing effectiveness (i.e., difference between circles and triangles for a given threat level in Fig. C5) with stage class (hatchlings, juveniles, adults). This pattern is likely a reflection of the threat-specific reductions to survivals ($\theta_{p,a}$; Table C3), in which the values were generally largest for adults. Additionally, the effectiveness of conservation actions also positively varied with threat level, particularly for adults. In other words, the largest improvements to adult survival with conservation action were observed at the high threat level (red points in Fig. C5).

In addition to the threats listed in Table C3, we also evaluated the effects of habitat loss. In all scenarios the SAST population declines outpaced the habitat loss rate (Fig. C6), meaning that the SAST population size never reached the population ceiling to trigger the density dependent response ($F_A = 0$).

Model limitations and weaknesses. — Our model was constructed to predict current and future conditions of the Suwannee alligator snapping turtle within the Suwannee analysis unit. While this model 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. suwanniensis* demographic studies required the use of data from closely related species (e.g., *M. temminckii, 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).

All of the baseline demographic parameters used in this study (Table C1) were from other *Macrochelys* spp. or expert-derived. No peer reviewed estimates of *M. temminckii* survival probabilities exist, and it is unclear if potential estimates from on-going studies would be representative of “baseline conditions”. Moreover, potential differences in non-threat demographic drivers (e.g., climate) between *M. suwanniensis* and *M. temminckii* remain poorly understood, thus we were unable to incorporate such effects into our model. While using demographic estimates from other species is imperfect, it is a common practice, particularly if coupled with the techniques used in our model to address uncertainty such as stochastic (rather than deterministic) simulation and parametric uncertainty.

Due to a dearth of demographic studies on *M. suwanniensis* and closely related species (*M. temminckii*), 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 C1). 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 A) 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. Moreover, we also used a weighted mean 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. Additionally, 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.

Further studies would be needed to comprehensively validate or "truth" the quantities that we elicited from expert. However, a recent paper by Steen and Robinson (2017, entire) examined the effects of hook ingestion on adult turtle mortality in multiple species, including *M. temminckii* and *suwanniensis*. Steen and Robinson (2017) report that among adult alligator snapping turtles that ingest a fish hook, a given individual has a 0.06 (95% Bayesian credible interval: 0.03–0.11, p. 1336) probability of dying from the ingested hook. If this probability is applied to the baseline adult survival estimate in C1 (0.95 – 0.06= 0.89) it equates to a 6% reduction in adult survival probability from 0.95 to 0.89.

Our approach to modeling the effects of hook ingestion differed from Steen and Robinson (2017, p. 1335) in that the percent reduction to adult survival due to hook ingestion (0.0824 in HKI column, Table C3) in our framework is conditioned on being in an area where the threat occurs (i.e., the spatial extent of HKI), rather than swallowing a hook. This distinction suggests that the percent reduction to adult survival due to HKI used by our study (0.0824, Table 3) is substantially higher than the 6% reduction in survival produced by the 0.06 conditional mortality probability reported by Steen and Robinson (2017, p. 1336). However, Steen and Robinson (2017, p. 1336) caution that their results are likely conservative, as their study estimated mortality rates from wild-caught turtles, meaning that they lived long enough to be encountered by the study, whereas a turtle that was severely injured from hook ingestion would be more likely to die faster and be missed by a mark-recapture study. While the comparison between our mortality attributed to hook ingestion by the experts in our study and that estimated by Steen and Robinson (2017, p. 1336) is imperfect due to methodological differences, we assert that the general similarity (8% vs. 6%) between the two studies inspires some degree of confidence in the expert-elicited quantities.

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 Suwannee analysis unit. 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 SAST 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.

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Table C1. Summary of Suwannee alligator snapping turtle (*Macrochelys suwanneinsis*; SAST hereafter) 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 ($P_{\mathrm{J}} = \varphi_{\mathrm{J}} \times (1 - \gamma_{\mathrm{J}})$); (2) juvenile growth, the probability of surviving as a juvenile and transitioning to the adult stage ($G_{\mathrm{J}} = \varphi_{\mathrm{J}} \times \gamma_{\mathrm{J}}$); (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 ($F_{\mathrm{A}} = BP \times CS \times \varphi_{\mathrm{N}} \times NSC \times \mathrm{FP} \times \varphi_{\mathrm{H}}$). 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]	**Mean (μ)**	**Sampling Var. (σ_S^2)**	**Process Var. (σ_P^2)**	**Source**	**Source Location**
P_{J}, G_{J}	φ_{J}	0.860	0.0277^2	0.01053^2	Folt et al. 2016	Spring Creek, Georgia
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; 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] 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.

[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 C2.

Table C2. Clutch sizes of Suwannee alligator snapping turtles (*Macrochelys suwanniensis*) used to compute a weighted mean in a stochastic population simulation. The mean values were weighted by the sample size 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 C3. Summary of threats effects applied to a Suwannee alligator snapping turtle (*Macrochelys suwanniensis*; hereafter SAST) population viability analysis. All quantities were derived using remote expert elicitation from a team of *Macrochelys* spp. experts. The first four rows reflect threat-specific percent reductions (mean ± standard deviation) to SAST survival parameters (Table C1), which were assumed to remain constant throughout the range of both SAST and the alligator snapping turtle (*M. temminckii*). The spatial extent (final row), describes the proportion of the SAST population exposed to each threat. The mean values for the threat-specific survival reductions listed here represent the "expert-elicited" scenarios (e.g., EETH, EETH+), whereas they were reduced or increased by 25% for the "decreased threat" and "increased threat" threat scenarios, respectively (Table 7). The spatial extents reported here represent "conservation absent" scenarios (DETH, EETH, INTH), and were reduced by 25% for the "conservation present" scenarios (DETH+, EETH+, INTH+). The dashes (–) indicate that the survival parameter was not exposed to the specific threat within the model (first four rows) or that the threat did not occur within the SAST's distribution (final row). In the former case, for example, hatchlings are likely too small to ingest hooks, so their survival rate was not reduced by HKI.

	Commercial Bycatch (BYC)	**Recreational Bycatch (BYR)**	**Hook Ingestion (HKI)**	**Illegal Collection (CLI)**	**Subsidized Nest Predators (SNP)**
Hatchling Survival (φ_H)	0.0001 ± 0.0007	–	–	0.0047 ± 0.0028	–
Juvenile Survival (φ_J)	0.0403 ± 0.0258	0.0579 ± 0.0205	0.0615 ± 0.0195	0.0565 ± 0.0191	–
Adult Survival (φ_A)	0.0630 ± 0.0361	0.0741 ± 0.0351	0.0824 ± 0.0322	0.1947 ± 0.0625	–
Nest Survival (φ_N)	–	–	–	0.0110 ± 0.01167	0.6075 ± 0.1154
Spatial Extent (ω)	–	0.500 ± 0.118[a]	0.500 ± 0.118	0.300 ± 0.087	0.07 ± 0.013

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[a] The expert elicitation questionnaire used a single question for the spatial extent of both BYR and hook ingestion (HKI) (Appendix A, Question 5), so we used the mean spatial extent for both threats in the model.

Table C4. Summary of Suwannee alligator snapping turtle (*Macrochelys suwanniensis*; SAST) projection matrix elements and elasticities from a stochastic simulation framework. The framework simulated SAST population dynamics for 50 years, under six different future condition scenarios with 500 iterations each. The six scenarios included decreased (DE-), expert-elicited (EE-), or increased (IN-) threat levels, with conservation action absent (-TH) or present (TH+) (Table 7). 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 values (± standard deviation) (top portion) and elasticities (bottom portion) for each element, averaged across all iterations and years, are given below, with their overall range (i.e., minimum and maximum values observed) in parentheses. For comparative purposes, we also report the matrix elements and elasticities for a deterministic analysis, in which the baseline demographic parameters (Table C1) are used, with no threat effects or stochasticity.

Scenario	P_J	G_J	P_A	F_A
		Projection Matrix Elements		
DETH	0.83 ± 0.04 (0.54, 0.956)	0.016 ± 0.017 (0, 0.281)	0.853 ± 0.042 (0.59, 0.968)	0.219 ± 0.119 (0.014, 0.99)
EETH	0.816 ± 0.04 (0.543, 0.954)	0.016 ± 0.017 (0, 0.277)	0.82 ± 0.045 (0.606, 0.947)	0.212 ± 0.123 (0.014, 1.202)
INTH	0.797 ± 0.041 (0.608, 0.928)	0.017 ± 0.017 (0, 0.145)	0.789 ± 0.048 (0.551, 0.934)	0.222 ± 0.12 (0.009, 1.244)
DETH+	0.846 ± 0.039 (0.68, 0.971)	0.017 ± 0.017 (0, 0.19)	0.876 ± 0.038 (0.643, 0.98)	0.219 ± 0.12 (0.011, 1.439)
EETH+	0.832 ± 0.039 (0.553, 0.952)	0.017 ± 0.017 (0, 0.219)	0.852 ± 0.041 (0.66, 0.966)	0.216 ± 0.119 (0.014, 1.277)
INTH+	0.817 ± 0.041 (0.551, 0.943)	0.016 ± 0.016 (0, 0.178)	0.828 ± 0.044 (0.624, 0.958)	0.21 ± 0.114 (0.011, 1.045)
Deterministic	0.843	0.017	0.950	0.229
		Elasticities		
DETH	0.392 ± 0.273 (0, 1)	0.024 ± 0.019 (0, 0.25)	0.56 ± 0.273 (0, 1)	0.024 ± 0.019 (0, 0.25)
EETH	0.478 ± 0.289 (0, 1)	0.026 ± 0.03 (0, 0.25)	0.47 ± 0.283 (0, 1)	0.026 ± 0.03 (0, 0.25)
INTH	0.513 ± 0.272 (0, 1)	0.039 ± 0.054 (0, 0.25)	0.409 ± 0.258 (0, 1)	0.039 ± 0.054 (0, 0.25)
DETH+	0.36 ± 0.256 (0, 1)	0.024 ± 0.017 (0, 0.115)	0.593 ± 0.258 (0, 1)	0.024 ± 0.017 (0, 0.115)
EETH+	0.402 ± 0.265 (0, 1)	0.025 ± 0.02 (0, 0.25)	0.549 ± 0.265 (0, 1)	0.025 ± 0.02 (0, 0.25)
INTH+	0.448 ± 0.278 (0, 1)	0.026 ± 0.027 (0, 0.25)	0.501 ± 0.274 (0, 1)	0.026 ± 0.027 (0, 0.25)
Deterministic	0.1510	0.0244	0.8002	0.0244

Table C5. Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) mean asymptotic population growth rates (lambdas; λ) derived from transition matrices in a stochastic simulation framework. We simulated six scenarios that 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+) (Table 7). For each scenario, we ran 500 replicates of SAST 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 deterministic projection matrix, which contained the baseline demographic parameter values (i.e., the raw values before adjustment for threat exposure) listed in Table C1.

	Conservation Action	
Threat Level	TH	TH+
Decreased	0.900 ± 0.039	0.920 ± 0.038
	(0.747, 1.106)	(0.780, 1.125)
Expert-Elicited	0.873 ± 0.040	0.899 ± 0.040
	(0.729, 1.065)	(0.739, 1.145)
Increased	0.852 ± 0.041	0.878 ± 0.041
	(0.691, 1.076)	(0.715, 1.076)
Deterministic	0.978	

Table C6. Stable stage distributions for simulated Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) populations. We derived mean stable stage distributions (± standard deviation) from the transition matrices across all timesteps (*n*=50) and iterations (*n*=500) for each analysis unit and scenario combination. The six scenarios included three different threat levels— decreased (DE), expert-elicited (EE), or increased (IN)— with conservation action absent (TH) or present (TH+) (Table 7). The mean proportion of each stage class within the stable stage distribution for the stochastic simulation framework are shown below, with the range (i.e., minimum and maximum values observed) values in parentheses. For comparative purposes, we also calculated the stable stage distributions for the deterministic projection matrix for units 1–7 and unit 8, which contained the baseline demographic parameter values (i.e., the raw values before adjustment for threat exposure) listed in Table C1.

Scenario	Stable Stage Distribution	
	Juveniles	Adults
DETH	0.754 ± 0.152	0.246 ± 0.152
	(0.113, 1)	(0, 0.887)
EETH	0.783 ± 0.152	0.217 ± 0.152
	(0.101, 1)	(0, 0.899)
INTH	0.785 ± 0.151	0.215 ± 0.151
	(0.218, 1)	(0, 0.782)
DETH+	0.742 ± 0.151	0.258 ± 0.151
	(0.101, 1)	(0, 0.899)
EETH+	0.756 ± 0.149	0.244 ± 0.149
	(0.167, 1)	(0, 0.833)
INTH+	0.773 ± 0.15	0.227 ± 0.15
	(0.154, 1)	(0, 0.846)
Deterministic	0.6275	0.3725

Table C7. Summary of Suwannee alligator snapping turtle (*Marcochelys suwanniensis*) population outcomes from six alternative future conditions scenarios. For each scenario, we ran 500 replicates of SAST population dynamics simulated for 50 years. 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+) (Table 9). For each scenario, we calculated the number of replicates in which the total population (both stage classes) declined to zero (extinction probability; p_{EX}) or less than 5% of the starting population size (quasi-extinction probability; p_{QX}). For the replicates in which the population reached extinction or quasi-extinction, 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 replicates) listed in parentheses below. The asterisk (*) indicates that only one simulation replicate went extinct within a scenario, meaning that a standard deviation for t_{EX} could not be calculated.

Threat Level	p_{EX}		t_{EX}		p_{QX}		t_{QX}	
	TH	TH+	TH	TH+	TH	TH+	TH	TH+
Decreased	0.07	0.00	48.39 ± 2.71	49.0 ± *	0.94	0.72	34.56 ± 6.61	39.57 ± 6.45
			(40, 51)	(49, 49)			(19, 51)	(24, 51)
Expert-Elicited	0.40	0.06	46.16 ± 3.56	48.28 ± 2.42	0.99	0.92	28.48 ± 5.70	34.51 ± 6.49
			(35, 51)	(42, 51)			(16, 51)	(20, 51)
Increased	0.81	0.30	43.21 ± 4.27	47.09 ± 3.20	0.99	0.98	23.85 ± 4.48	29.50 ± 5.52
			(31, 51)	(36, 51)			(15, 49)	(17, 51)

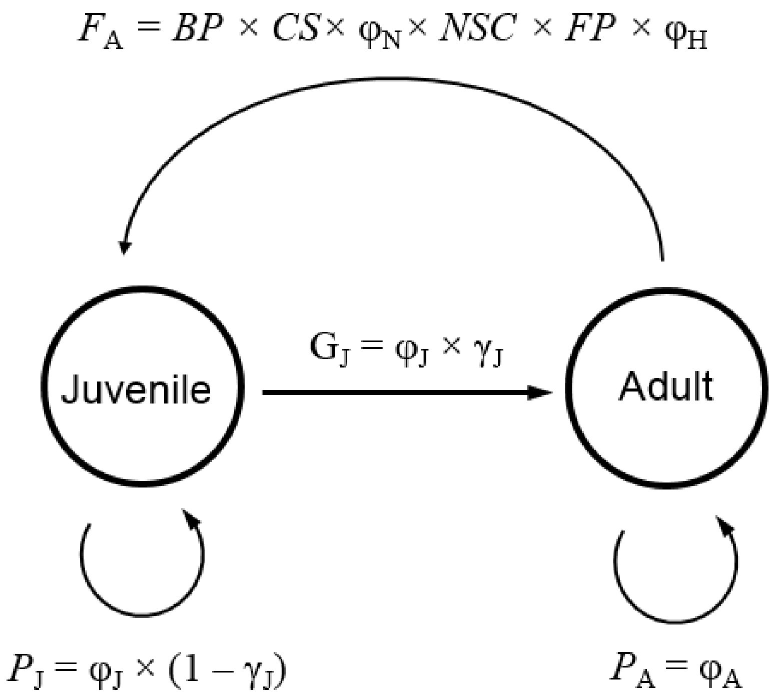

Figure C1. Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) life cycle diagram for a female only two-stage prebreeding matrix model. The open circles represent the two life stages, juveniles (immature individuals) and adults (breeding individuals), denoted by the J and A subscripts, respectively. 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 C1.

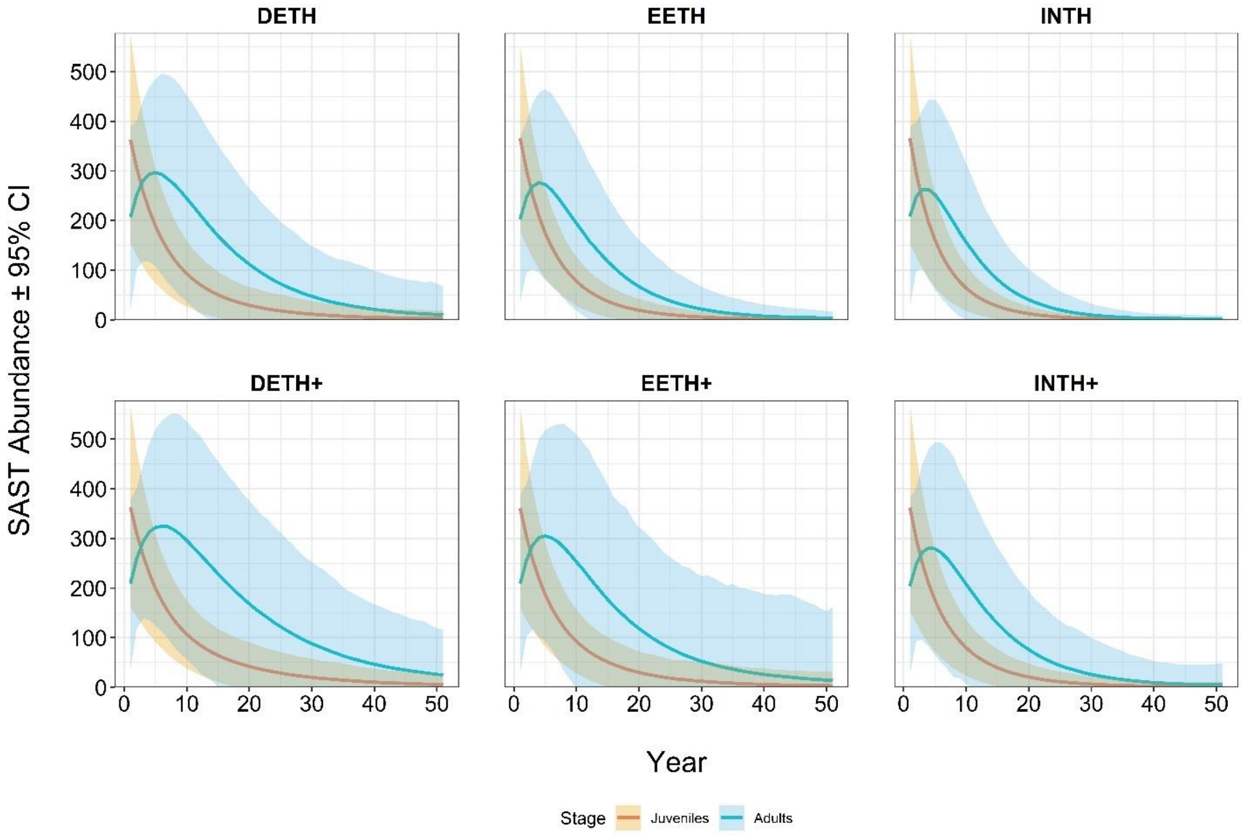

Figure C2. Simulated Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) stage class-specific abundance (females only) over a 50-year period. The juvenile stage is shown in orange and adults in blue. The solid lines depict the mean abundance trajectory across 500 stochastic simulations and the shaded areas reflect the 95% confidence intervals (CI). The bolded letters at the top of each panel reflect each of the six scenarios. The columns indicate the scenario's threat level (left to right): decreased (DE), expert-elicited (EE), or increased (IN). The top row indicates conservation action absent (TH) or present (TH+).

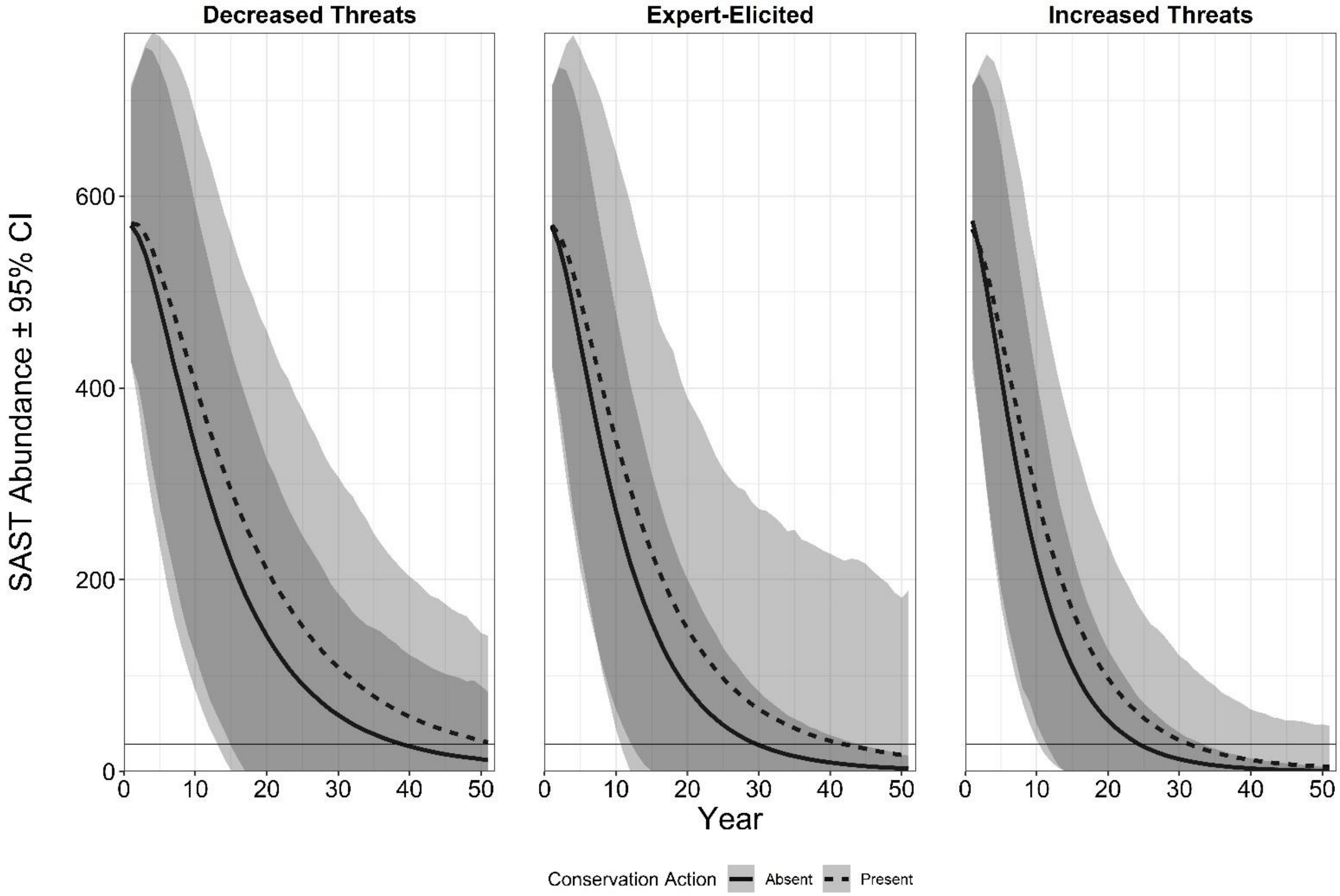

Figure C3. Simulated Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) total abundance (females only) over a 50-year period. The lines in each panel depict the mean abundance trajectory across 500 stochastic simulations for each of six scenarios and the shaded areas reflect the 95% confidence intervals (CI). The three panels indicate the scenario's threat level: decreased (DE), expert-elicited (EE), or increased (IN). The lines within each panel show the abundance trajectories for the conservation action absent (TH; solid) and present (TH+; dashed) scenarios, and the quasi-extinction threshold (<5% of total abundance in Year 1) is given by thin flat line.

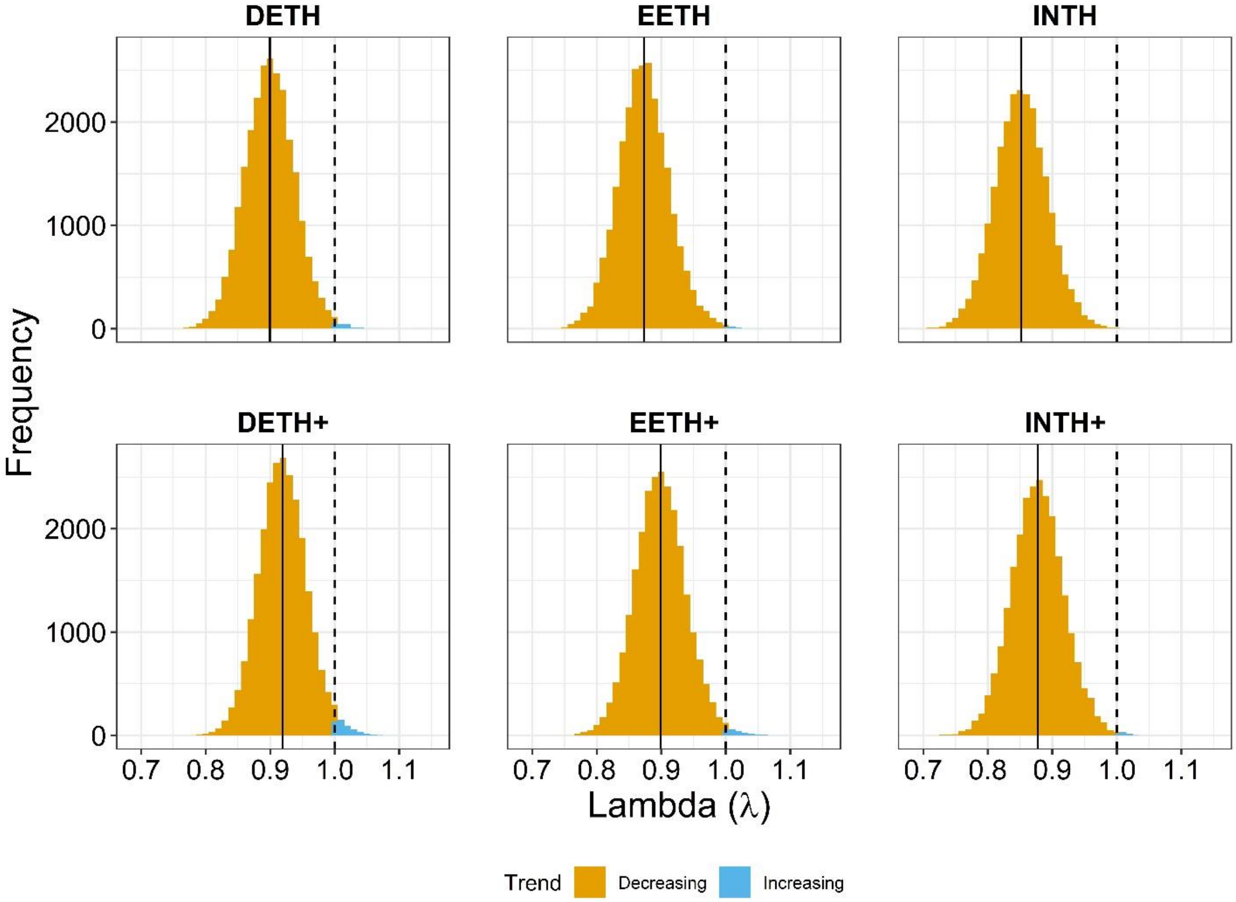

Figure C4. Histograms of asymptotic population growth rates (lambdas; λ) derived from two stage, prebreeding census transition matrices (**A**) used to project Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) population dynamics the Suwannee analysis unit. The letters in bold above each panel reflects one of six future condition scenarios, varying by threat level (columns) and conservation action presence or absence (rows). For each scenario, the expert threat specific percent reductions to survival (Table C3) were left unchanged (EE-; center), decreased (DE-; left), or increased (IN-; right) by 25%. In the conservation action absent (TH; top row) scenarios, the expert elicited spatial extents for each threat (excluding SNP) were used, and decreased by 25% for the conservation action present (TH+; bottom row) scenarios. Each scenario generated a maximum of 25K transition 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 C1), threat specific parameter reductions, and spatial extents (Table C3) at each iteration and timestep that created variation among the transition matrices and their associated λs. The solid vertical line represents the λ distribution mean, whereas the dashed vertical line is reference line at λ=1 to separate values of λ that indicate a decreasing population (orange) from those that indicate stable or increasing population (blue).

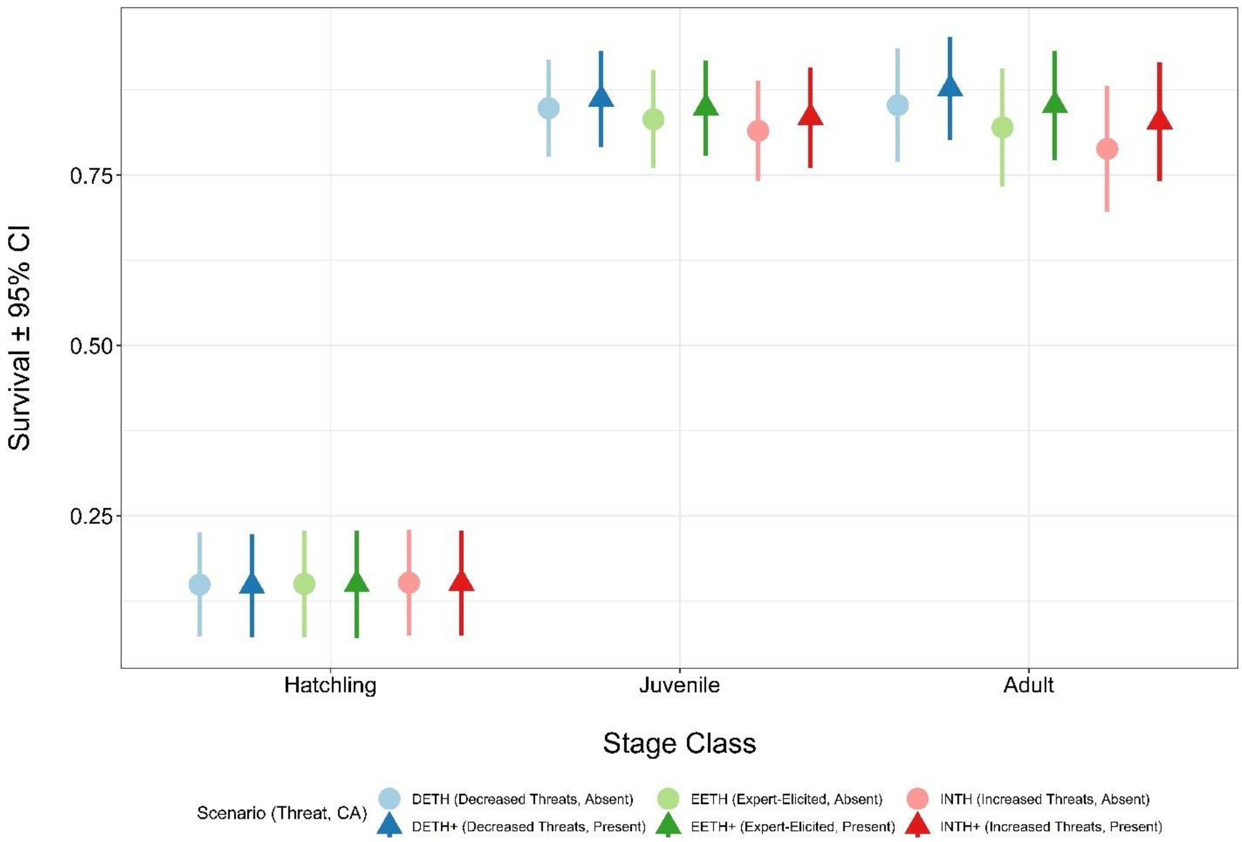

Figure C5. Mean stage class-specific Suwannee alligator snapping turtle (*Macrochelys suwanniensis*; SAST hereafter) survival parameters and their associated 95% confidence intervals (CI). The matrix model used to project SAST population dynamics was comprised of two stages (juveniles and adults), though the hatchling (neonate) survival parameter was contained within the adult fecundity element (F_A, Eq. 1, Table C1) and was exposed to threats in the model (Table C3). Within stage class (x-axis), 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).

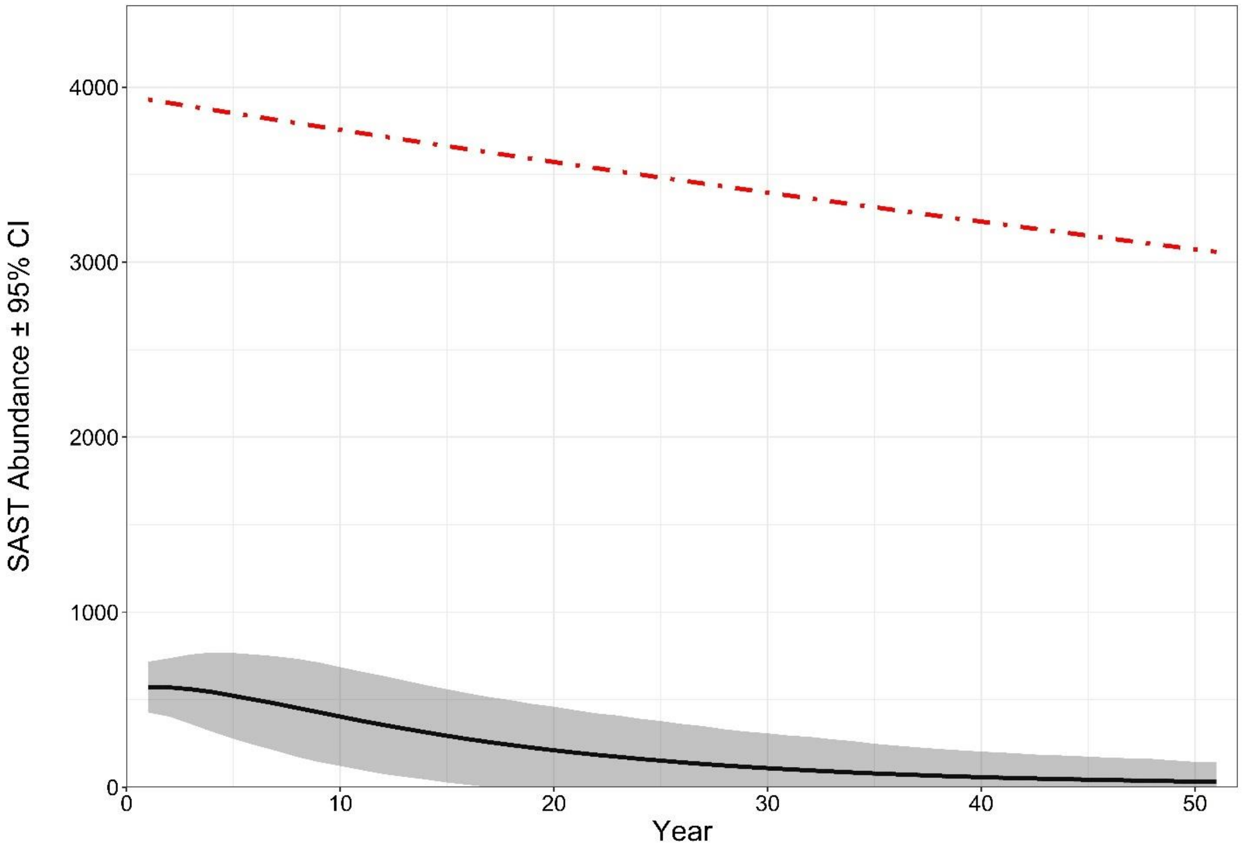

Figure C6. Simulated Suwannee alligator snapping turtle (*Macrochelys suwanniensis*) total abundance (females only, juvenile and adult stage classes) over a 50-year period under the decreased threat level with conservation action (DETH+) scenario. The solid black line depicts the mean total abundance trajectory across 500 stochastic simulations and the shaded area reflects the 95% confidence intervals (CI). The red dot-dashed is the Suwannee analysis unit's population ceiling, which decreases by 0.50% (±0.71 SD) per year (Eq. 5), based on the mean number of habitat loss mechanisms elicited from experts. The initial population ceiling was set at the expert-elicited current maximum AST abundance +25%, adjusted to include non-hatchling females only.

APPENDIX D - Mean Predicted Future Abundances

Table D1. Model-predicted mean abundances and standard deviations at 5 year intervals for *M. suwanniensis* 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	SUWA	0	573.5	71.0	DETH+	SUWA	0	571.0	71.6
DETH	SUWA	5	623.0	137.4	DETH+	SUWA	5	633.0	153.7
DETH	SUWA	10	611.0	177.3	DETH+	SUWA	10	629.5	194.5
DETH	SUWA	15	412.4	174.8	DETH+	SUWA	15	433.1	186.1
DETH	SUWA	20	269.5	150.8	DETH+	SUWA	20	287.8	161.7
DETH	SUWA	25	176.1	124.6	DETH+	SUWA	25	190.5	131.0
DETH	SUWA	30	114.4	101.1	DETH+	SUWA	30	124.5	102.5
DETH	SUWA	35	75.6	81.3	DETH+	SUWA	35	83.1	84.6
DETH	SUWA	40	50.5	65.8	DETH+	SUWA	40	56.3	72.4
DETH	SUWA	45	34.5	54.4	DETH+	SUWA	45	38.6	62.9
DETH	SUWA	50	23.5	42.7	DETH+	SUWA	50	26.7	51.8
EETH	SUWA	0	572.9	72.0	EETH+	SUWA	0	566.2	75.3
EETH	SUWA	5	588.3	145.7	EETH+	SUWA	5	581.3	140.4
EETH	SUWA	10	546.2	170.5	EETH+	SUWA	10	544.1	165.3
EETH	SUWA	15	333.8	141.9	EETH+	SUWA	15	341.6	148.8
EETH	SUWA	20	196.5	104.3	EETH+	SUWA	20	205.9	114.9
EETH	SUWA	25	114.5	74.8	EETH+	SUWA	25	123.3	90.4
EETH	SUWA	30	67.2	54.4	EETH+	SUWA	30	73.9	68.5
EETH	SUWA	35	39.5	38.0	EETH+	SUWA	35	45.2	51.6
EETH	SUWA	40	23.3	26.2	EETH+	SUWA	40	27.9	40.0
EETH	SUWA	45	14.0	18.7	EETH+	SUWA	45	17.8	31.3
EETH	SUWA	50	8.6	13.4	EETH+	SUWA	50	11.5	25.8
INTH	SUWA	0	572.0	71.5	INTH+	SUWA	0	571.4	72.3
INTH	SUWA	5	526.2	134.3	INTH+	SUWA	5	538.7	137.1
INTH	SUWA	10	453.0	139.9	INTH+	SUWA	10	474.3	155.5
INTH	SUWA	15	246.4	110.1	INTH+	SUWA	15	265.8	123.3
INTH	SUWA	20	127.7	71.6	INTH+	SUWA	20	143.6	84.9
INTH	SUWA	25	65.3	44.9	INTH+	SUWA	25	76.4	57.2
INTH	SUWA	30	33.7	27.5	INTH+	SUWA	30	41.1	38.7
INTH	SUWA	35	17.4	17.0	INTH+	SUWA	35	22.3	25.3
INTH	SUWA	40	9.1	10.3	INTH+	SUWA	40	12.2	17.3
INTH	SUWA	45	4.7	6.4	INTH+	SUWA	45	6.7	11.8
INTH	SUWA	50	2.4	4.1	INTH+	SUWA	50	3.7	8.1

APPENDIX E – Suwannee alligator Snapping Turtle Suitable Habitat

Spatial analysis of the *M. suwanniensis* range was performed to determine the extent of suitable habitats available and the amount of lands in conservation.
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.

Figure1. Suitable *M. suwanniensis* habitat within the range of the species.

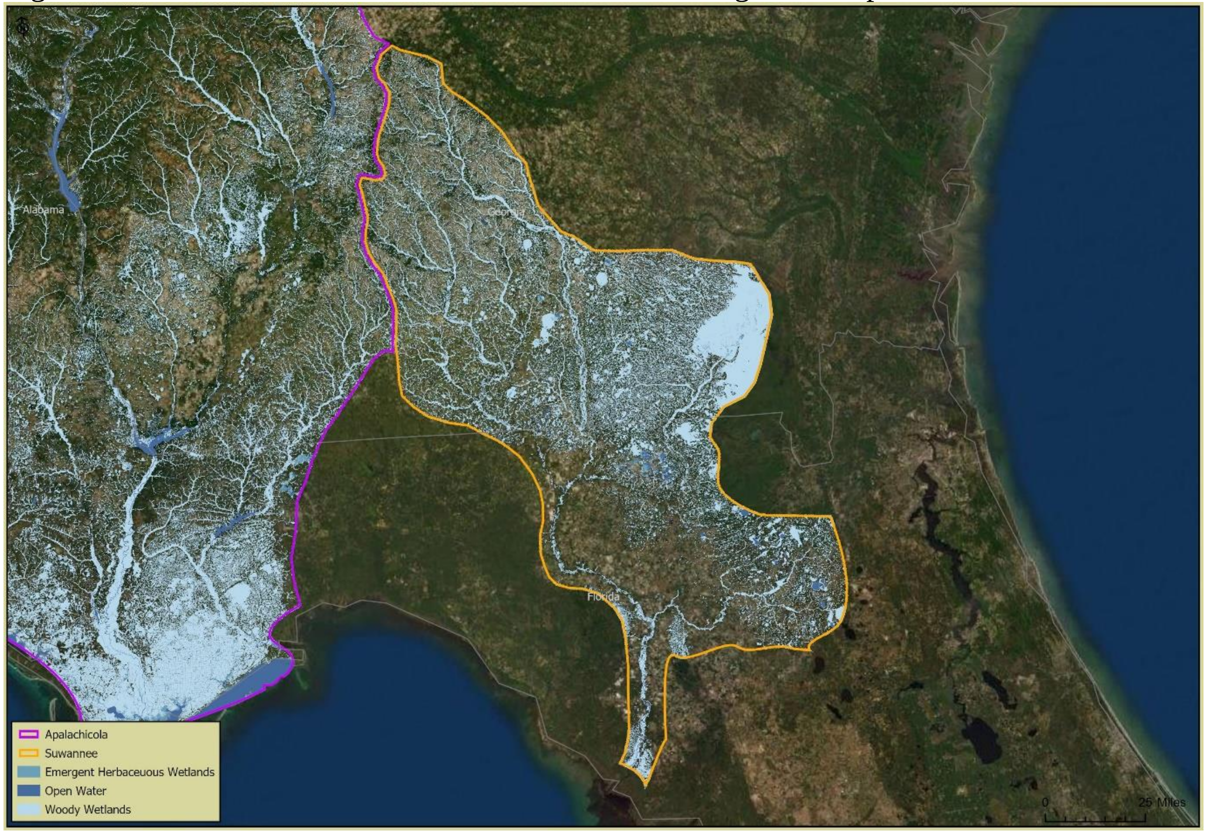

Table 1. Acres of suitable *M. suwanniensis* suitable habitat within the range of the species.

Area	Emergent Herbaceuous Wetlands	Open Water	Woody Wetlands	Total Acres	Analysis Unit Acres	Percentage of Unit is Suitable Habitat
Suwannee River Basin	62,981	64,890	1,620,961	1,748,832	5,934,668	29.47%

APPENDIX G – Suitable *M. suwanniensis* Habitat on Lands in Conservation within the Range of the Species

Table 1. Acres of suitable *M. suwanniensis* habitat within conservation areas in the Suwannee River basin.

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 1. Lands in conservation within the range of the *M. suwanniensis* in the Suwannee Unit.

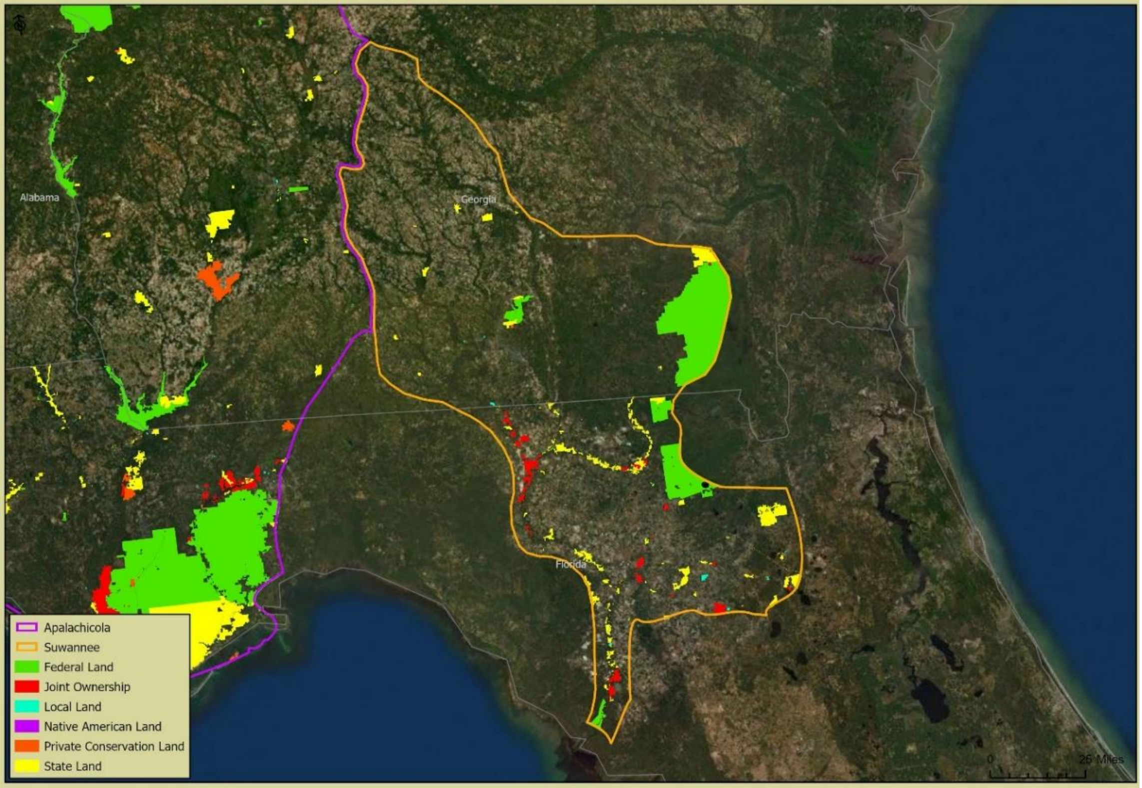

Figure 2. Suitable *M. suwanniensis* habitat on conservation lands within the Suwannee Unit.