Species Status Assessment for the Frosted Flatwoods Salamander (*Ambystoma cingulatum*) Version 1.0

Photo credit: Mark Mandica (The Amphibian Foundation)

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Acknowledgments

Prepared by Dr. Susan Walls (U.S. Geological Survey), Anna Farmer (Florida Fish and Wildlife Conservation Commission), with support from Dr. Sean Blomquist (USFWS), Harold Mitchell (USFWS), Arianne Messerman (University of Missouri), John Palis (Palis Environmental Consulting), Melanie Olds (USFWS), John Dunlap (USFS), Jamie Barichivich (USGS), Dr. Carola Haas (Virginia Tech University), George Brooks (Virginia Tech University) Katie O'Donnell (USGS), Pierson Hill (Florida Fish and Wildlife Conservation Commission), Jana Mott (The Nature Conservancy), Dr. Brooke Talley (Florida Fish and Wildlife Conservation Commission), Valuable peer review of the document provided by Dr. Kurt Buhlmann, University of Georgia, Savannah River Ecology Laboratory, and Dr. David Bishop, The Nature Conservancy.

EXECUTIVE SUMMARY

This species status assessment (SSA) reports the results of a comprehensive status review for the frosted flatwoods salamander (*Ambystoma cingulatum*), documents the species' historical conditions, and provides estimates of current and future conditions under a range of different scenarios. The flatwoods salamander (*Ambystoma cingulatum*) was federally listed in 1999 (64 FR 15691) as a threatened species under the Endangered Species Act of 1973 as amended (Act). In 2009, the flatwoods salamander was recognized as two distinct species, the reticulated flatwoods salamander (*A. bishopi*) and the frosted flatwoods salamander (*A. cingulatum*). Critical habitat was designated for both species (74 FR 6700) with 22,970 ac (9,297 ha) in 19 sub-units designated for the frosted flatwoods salamander.

Frosted flatwoods salamanders are moderately-sized (mean 76 mm snout-to-vent length, 135 mm total length) salamanders with relatively short, pointed snouts and stout tails (Martof and Gerhardt, 1965; Palis, 1996; John Palis, Palis Environmental Consulting, 1995, unpublished data). Individuals weigh from 4.5 to 14.8 g (adult males and adult gravid [containing mature eggs] females, respectively; Palis, 1996; Palis, Palis Environmental Consulting, 1995, unpublished data). Pierson Hill of the Florida Fish and Wildlife Conservation Commission captured a gravid female weighing 17.4g in 2018 (Pierson Hill Pers comm 2018). Their bodies are black to chocolate-black with fine, irregular, light gray lines or specks that form a reticulate or cross-banded pattern across the back.

Frosted flatwoods salamanders are pond-breeding amphibians with complex life cycles (i.e., there is an aquatic larval life history stage, as well as a terrestrial juvenile and adult stage). As adults, flatwoods salamanders return to seasonally-flooded wetlands to breed in the fall, where females lay eggs singly or in small clusters usually at the base of plants, in dry areas that will later fill with water provided by winter rainfall (Anderson and Williamson, 1976; Palis, 1995a, 1997). Well-developed embryos hatch into larvae after inundation and metamorphose between March and May after an 11 to 18 week larval period (Palis, 1995a). Juveniles normally disperse from ponds shortly after metamorphosis, but may stay in or near ponds during seasonal droughts

(Palis, 1997). Juveniles and adults are highly fossorial and spend much of their time in crayfish burrows or root channels until they reach sexual maturity (1 year for males; 2 years for females) and most return to their natal pond to breed during the fall months (Petranka, 1998).

Breeding wetlands are located within mesic (moderate moisture) to intermediate-mesic pine-dominated flatwoods/savanna communities where adults and juveniles live outside of the breeding season. Pine flatwoods/savannas are characterized by low, flat topography and relatively poorly drained, acidic, sandy soil that becomes seasonally saturated. This ecosystem is characterized by open pine woodlands maintained by frequent (1 to 3 years), growing season (summer) fires.

The SSA process can be categorized into three sequential stages. During the first stage, we used the conservation biology principles of resiliency, redundancy, and representation (together, the 3Rs) to evaluate individual frosted flatwoods salamander life history needs. The next stage involved an assessment of the historical and current condition of the species' demographics and habitat characteristics, including an explanation of how the species arrived at its current condition. The final stage of the SSA involved making predictions about its response to positive and negative environmental and anthropogenic influences. This process used the best available information to characterize viability as the ability of the species to sustain populations in the wild over time.

To evaluate the current and future viability of the frosted flatwoods salamander, we assessed a range of conditions to allow us to consider the species' resiliency, representation, and redundancy. For the purposes of this assessment, populations were delineated by occupied breeding wetlands (i.e., ponds) buffered by a 1500-foot radius of upland habitat following the critical habitat designation (74 FR 6700).

Resiliency, assessed at the population level, describes the ability of a population to withstand stochastic disturbance events. Like many amphibians that breed in ephemeral wetlands, flatwoods salamanders exhibit dramatic fluctuations in abundance across years. Specific environmental conditions are required for successful recruitment; drought years result in catastrophic reproductive failure. To discern long-term trends from natural fluctuations, a stochastic Integral Projection Model (IPM) was constructed from 10 years of drift fence data obtained at two breeding wetlands on Eglin AFB. A population viability analysis (PVA) was conducted, whereby simulated populations were projected into the future and extinction risks under various scenarios were calculated (George Brooks, Virginia Tech, 2019, unpublished data). Owing to the stochastic nature of recruitment, extinction risk was high for a single population. Thus, the species will need 101 resilient metapopulations distributed across its range to persist into the future and avoid extinction. As we consider the future viability of the species, more metapopulations with high resiliency distributed across the known range are associated with higher overall viability. For the reticulated flatwoods salamander, metapopulations were delineated by occupied breeding wetlands (i.e., ponds) buffered by a 1500 foot (approximately 500 m) radius of upland habitat in the 2009 critical habitat designation (74

FR 6700). In this document, we follow that definition of a, metapopulation although we discuss additional advancements in the understanding of flatwoods salamander populations. In addition to the PVA, species' resiliency was assessed based on breeding wetland occupancy and according to 6 resiliency categories describing habitat quality: (1) extent of woody vegetation in understory of upland habitat; (2) quality and composition of the wetland basin overstory; (3) presence and composition of the wetland midstory vegetation; (4) type of wetland understory vegetation and presence of organic duff/peat layer in basin; (5) adequacy of wetland hydroperiod for completion of metamorphosis; and (6) burn frequency/burn season for the compartment in which breeding sites are located. We discuss each of these factors.

Redundancy describes the ability of the species to withstand catastrophic disturbance events. A PVA conducted for this species revealed a high probability of local extirpation under a business as usual scenario (George Brooks, Virginia Tech, 2019, unpublished data). Multiple independent populations, exhibiting asynchronous dynamics, will be required to secure long-term viability of the species and avoid regional extinction. For the reticulated flatwoods salamander, we considered the distribution of the species remaining on the landscape. We also considered flood models (e.g. SLOSH, etc) for potential sea level rise to get an indication of threat for extant populations near the Gulf Coast. Roughly 25 metapopulations per each of the 4 Recovery Management Units (RMUs) is necessary to provide redundancy across the historic range; 101 resilient metapopulations in total will be required across the historic range to ensure the risk of extinction is low enough to allow the species to persist into the foreseeable future. Currently, all the extant metapopulations occur within RMU 1 (within the boundaries of the Apalachicola National Forest, and St Marks National Wildlife Refuge), except one metapopulation with low resiliency within RMU 3 located at Fort Stewart, Georgia.

Representation characterizes a species adaptive potential by assessing geographic, genetic, ecological, and niche variability. The frosted flatwoods salamander historically occurred throughout the Coastal Plain of the southeastern U.S., across South Carolina, Georgia, and the panhandle of Florida (Palis and Means, 2005). The species is currently represented in both genetic clades, albeit at one isolated and small population at Fort Stewart Army Base in Liberty County, Georgia in the Atlantic Coastal Plain on the eastern portion of the range. Multiple populations exist in and around the two areas of St. Marks National Wildlife Refuge and Apalachicola National Forest in Liberty and Wakulla Counties, Florida, respectively, representing the Gulf Coastal Plain on the western portion of the range. The RMUs were derived by dividing the range of the species into more manageable units, and assure better distribution of recovered populations across the range, by establishing 25 population targets in each of the RMUs. This would help prevent potentially clumping too many metapopulations into a confined geographic area within the range.

Together Resiliency, Redundancy, Representation, the 3R's, comprise the key characteristics that contribute to a species' ability to sustain multiple distinct populations in the wild over time (i.e., viability). Using the principles of the 3 R's, we characterized both the species' current viability and forecasted its future viability over a range of plausible future scenarios. We have assessed the frosted flatwoods salamander's levels of resiliency, redundancy, and representation currently and up to 80 years into the future by estimating the persistence of each current population and populations on currently occupied properties. Rankings are quantitative assessments of the relative condition of the frosted flatwoods salamander's remaining habitat within its known range based on the best available data as well as the knowledge and expertise of land managers and species experts (Appendix 1).

The most significant stressors to individuals and populations of the frosted flatwoods salamander include low population density, restricted range, low-quality breeding and upland habitat, vulnerability to stochastic events (e.g., extended drought, storm surge from hurricanes), inadequate habitat management (i.e., not enough growing season fire applied to the habitat to achieve meaningful restoration range wide, too little use of known restoration techniques, besides fire, to aid in the restoration of degraded former or potential breeding ponds), and inadequate funding to address recovery actions. Genetic bottlenecking and inbreeding could limit the ability for natural recovery in areas of extremely low population densities. Recovery actions (e.g. wetland creation, translocations) are necessary to reduce or eliminate these factors. Adjacent lands have some potential to support flatwoods salamanders, but surveys are mostly absent or lacking. Increasing survey effort within this region will eliminate uncertainty about the number and location of extant populations.

We considered a range of potential future scenarios that may be important influences on the status of the species, and our results describe this range of possible conditions in terms of how many, how much, and where habitat protections are needed to persist into the future (Table ES-1). The frosted flatwoods salamander will experience habitat loss and degradation in the future, and, in addition to the PVA results discussed above, we have forecasted what the species may have in terms of resiliency, redundancy, and representation at 1, 10, 20, 30 and 80 years in the future under the following scenarios:

1) Wetland succession continues due to inadequate or inappropriate habitat management on currently occupied properties throughout the range of the species;
2) Appropriate upland (terrestrial) habitat management occurs at currently occupied properties throughout the range of the species at 1, 10, 20, 30 and 80 years in the future
3) Restoration and management of wetland and upland habitats occur on currently occupied properties throughout the range of the species at 1, 10, 20, 30 and 80 years in the future.

Hurricane Michael (Oct. 10, 2018) caused significant impacts to the remaining occupied habitats within the range of *A. cingulatum*. The effects of the storm are still being evaluated. While the current situation of this species is indeed dire, there is hope for recovery. Currently, we have a great deal more available habitat than we do animals to populate these unoccupied, but historically extant, ponds and habitats. Focus on growing population sizes, expanding population numbers and increasing genetic health and viability is key to re-occupying these habitats. Proper management of these ephemeral wetlands and associated uplands to remain, or become suitable for supporting strong salamander populations is fundamental to achieving these goals. Regular, frequent, growing season fire is necessary for this to happen. Restoring the species' occurrence across its historic range is key to avoiding some of the effects of climate change on the most vulnerable (and currently occupied) remaining populations.

Table ES-1. Summary results of the frosted flatwoods salamander species status assessment.

3 R's	NEEDS	CURRENT CONDITIONS	FUTURE CONDITIONS (Viability)
Resiliency: large populations able to withstand stochastic events	Adequate water quality, wetland and upland groundcover and appropriate burn seasonality and frequency.	Currently 17 extant metapopulations exist within RMU 1 (in the ANF and SMNWR) and one metapopulation with low resiliency exists within RMU 3 (in Fort Stewart, GA) for a total of 18 metapopulations, Zero currently occur within RMU 2	Both sea level rise and increasing temperatures due to climate change are predicted to decrease the number of breeding ponds and resiliency of populations, particularly by 2100. The choice of management scenario has profound impacts on the number of breeding ponds in both the short and long-term. If species-specific wetland management (regularly burning of breeding ponds when they are dry) is not conducted, most active breeding ponds will become inactive by the Year 2050. We estimate 101 resilient metapopulations are needed to ensure the species persistence into the future.
Redundancy: number and distribution of populations to withstand catastrophic events	Multiple resilient populations throughout the Atlantic and Gulf clade areas within the historic range of the species.	Currently 17 extant metapopulations exist within RMU 1 (in the ANF and SMNWR) and one metapopulation exists within RMU 3 (in Fort Stewart, GA) for a total of 18 metapopulations, Zero currently occur within RMU 2	To avoid further population declines and ensure that populations are as resilient as possible in the face of anticipated climate changes, land managers will need to engage in and maximize the active restoration of potentially suitable breeding wetlands to offset anticipated breeding pond losses to sea level rise and other climate changes. In addition to wetland restoration efforts, salamander translocations to restored wetlands may be necessary if salamanders fail to colonize restored ponds. We estimate approximately 25 resilient metapopulations per RMU are required to ensure persistence of the species into the future.
Representation: genetic and ecological diversity to maintain adaptive potential	Decreased genetic inbreeding and less population isolation	Considered nearly extirpated in the Atlantic Clade (South Carolina and Fort Stewart), otherwise only the populations at ANF and SMNWR contribute to representation in the Gulf Coast Clade.	Use of genetic information to best determine how to implement reintroduction/translocation efforts to maximize genetic health of the populations is extremely important. Choosing currently unoccupied areas for repatriation is underway. Because of a lack of data on genetic information throughout the historical range of the species, we developed 3 representative units, which we call recovery management units (RMUs) to aid in ensuring the species persists in a diverse suite of ecological conditions.

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CHAPTER 1 – INTRODUCTION

The frosted flatwoods salamander (*Ambystoma cingulatum*) is a moderately-sized salamander endemic to longleaf pine (*Pinus palustris*)-dominated flatwoods/savanna communities in the Florida panhandle to southern South Carolina. These animals are currently found (east of the Apalachicola/Flint River system), on the Fort Stewart Military Installation in Southeast Georgia and, possibly, in the Francis Marion National Forest in South Carolina. Their status on private lands is largely unknown. Flatwoods salamanders were originally listed as a singular species on April 1, 1999 (64 FR 15691) under the Endangered Species Act of 1973, as amended (ESA). This listing was revised when *Ambystoma cingulatum* was split into two distinct species in 2009 (74 FR 6700). At the time of this revised listing, the reticulated flatwoods salamander (*A. bishopi*) was listed as endangered, and the frosted flatwoods salamander (*A. cingulatum*) retained threatened status. However, the most recent 5-year status review recommended changing the status of *A. cingulatum* to endangered (USFWS, 2019).

The Species Status Assessment (SSA) framework (USFWS, 2016) 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, for a listed species, to support all functions of the Endangered Species Program 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 Act decision making in many categories including listing, recovery, Section 7, Section 10, and reclassification decisions (the former four decision types are only relevant should the species warrant listing under the Act).

This document draws scientific information from resources such as primary peer-reviewed literature, reports submitted to the U.S. Fish and Wildlife Service (Service) and other public agencies, species occurrence information in GIS databases, and expert experience and observations. It is preceded by, and draws upon analyses presented in other Service documents, including the 1999 listing rule (64 FR 15691) and the 2009 revised listing which split the two species and designated critical habitat for both of them (74 FR 6700). Finally, we coordinate continuously with our partners engaged in ongoing research and conservation efforts. This assures consideration of the most current scientific and conservation status information. The frosted flatwoods salamander SSA is intended to provide the best available commercial and scientific information in the form of a review of available information strictly related to the current biological status of the species and factors that may affect its future biological status.

For the purpose of this assessment, we define viability as the ability of the species to sustain resilient populations in their ecosystem for at least 20 years. We chose 20 years because it is approximately 5-10 generations of the salamander and habitat changes are predicted to occur during this time. Using the SSA framework (Figure 1.1), we consider what the species needs to maintain viability by characterizing the status of the species in terms of its redundancy, representation, and resiliency (Wolf et al., 2015; USFWS, 2016).

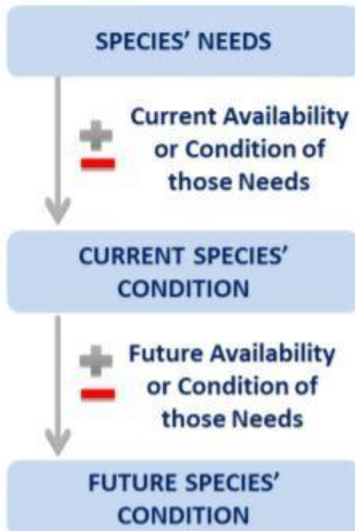

Figure 1.1. *Species Status Assessment Framework consisting of three basic stages. From USFWS (2016).*

- Resiliency describes the ability of a species to withstand stochastic disturbance. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Generally speaking, populations need abundant individuals within habitat patches of adequate area and quality to maintain survival and reproduction in spite of a random disturbance. We used breeding activity at known ponds and habitat quality as an index of resiliency of populations occupying breeding sites based on 6 categories: 1) uplands, 2) wetland overstory, 3) wetland midstory, 4) wetland understory, 5) wetland hydroperiod, and 6) burn frequency and season.

- Representation is assessed at the species' level. Representation describes the ability of a species to adapt to changing environmental conditions over time. For example, a species that has populations that exhibit geographic, genetic, or life history variation have greater ability to adapt to changing conditions. It is characterized by the breadth of genetic and environmental diversity within and among populations. Measures may include the number of varied niches occupied, the gene diversity, heterozygosity, or alleles per locus. Our analysis explores the relationship between the species life history and the influence of genetic and ecological diversity and the species ability to adapt to changing environmental conditions over time. The analysis identifies areas representing important geographic, genetic, or life history variation (i.e., the species' ecological settings).

- Redundancy describes the ability of a species to withstand catastrophic events (random events that have devastating consequences); it is about spreading risk among multiple populations to minimize the potential extinction of the species from catastrophic events. Redundancy is characterized by having multiple, resilient populations distributed within the species' ecological settings and across the species' range. It can be measured by population number, resiliency, spatial extent, and degree of connectivity. Our analysis explores the influence of the number, distribution, and connectivity of populations on the species' ability to withstand catastrophic events (e.g., rescue effect).

To evaluate the current and future viability of the frosted flatwoods salamander, we assessed a range of conditions to characterize the species' resiliency, representation, and redundancy (together, the 3Rs). This SSA Report provides a thorough account of known biology and natural history and assesses the risk of threats and limiting factors affecting the future viability of the species.

This SSA Report includes: (1) a description of frosted flatwoods salamander resource needs at the levels of the individual, population, and species, together with a characterization of the historic and current distribution of populations across the species' range (Chapter 2); (2) an assessment of the stressors and conditions that contributed to the current and future status of the species (Chapter 3) and (3) the degree to which various factors influenced viability (Chapter 4). Last, this report provides (4) a synopsis of the needs and stressors characterized in earlier chapters as a means of examining the future biological status of the species (Chapter 5). This document is a compilation of the best available scientific information (and associated uncertainties regarding that information) used to assess the viability of the frosted flatwoods salamander.

CHAPTER 2 – INDIVIDUAL, POPULATION AND SPECIES NEEDS: LIFE HISTORY, BIOLOGY, AND DISTRIBUTION

2.1 Description

Frosted flatwoods salamanders are moderately-sized (avg. 76 mm snout-to-vent length, avg. 135 mm total length), slender salamanders with relatively short, pointed snouts and stout tails (Martof and Gerhart, 1965; Palis, 1996; John Palis, Palis Environmental Consulting, 1995, unpublished data). Their heads are small and only about as wide as the neck and shoulder region (Petranka, 1998). They weigh from 4.5 to 11 grams (adult males and adult gravid [containing mature eggs] females), respectively (Palis, 1996; John Palis, Palis Environmental Consulting 1995, unpublished data). Pierson Hill reports a gravid female captured on the Apalachicola National Forest weighing 17.44g in 2018 (Pierson Hill, pers. comm. 2018). Their bodies are black to chocolate-black with fine, irregular, light gray lines or specks that form a reticulate or cross-banded pattern across the back. In some individuals, the gray pigment is widely scattered and "lichen-like." Melanistic, uniformly black individuals have been reported (Carr, 1940). The venter (underside) is dark gray to black with a scattering of gray spots or flecks.

2.2 Taxonomy and Nomenclature

The currently accepted classification for the frosted flatwoods salamander is (Integrated Taxonomic Information System, 2016):
Phylum: Chordata
Class: Amphibia
Order: Caudata
Family: Ambystomatidae
Genus: Ambystoma
Species: Ambystoma cingulatum

There are currently 33 species of *Ambystoma* recognized in North America (IUCN, 2018). Seventeen species are found exclusively in Mexico, eight are endemic to the U.S., eight are found in both the U.S. and Canada. Pauly et al. (2007) demonstrated that flatwoods salamanders are polytypic with a major disjunction at the Apalachicola River in Florida. Based on mitochondrial DNA, morphology, and allozymes, Pauly et al. (2007) recognized two species of flatwoods salamanders – the frosted flatwoods salamander, *Ambystoma cingulatum*, to the east of the Apalachicola drainage, and the reticulated flatwoods salamander, *A. bishopi*, to the west. The ringed salamander, *A. annulatum*, is the closest phylogenetic relative of the flatwoods salamanders, with all three species grouping together in their own clade (Kraus, 1988; Shaffer et al., 1991; Williams et al., 2013). In turn, this clade is the sister group to the tiger salamander clade (*A. californiense, A. mexicanum, A. ordinarium,* and *A. tigrinum*) (Williams et al., 2013).

2.3 Life History

The frosted flatwoods salamander is a pond-breeding amphibian with a complex life cycle; i.e., there is a terrestrial egg/embryo stage, an aquatic larval stage, as well as a terrestrial metamorphosed juvenile and adult stage (Figure 2.1). As adults (Figure 2.1A), flatwoods salamanders migrate to ephemeral (seasonally-flooded) wetlands to breed in the fall (October to December), where females lay eggs singly or in small groups (average of 7) in moisture-retaining microhabitats, particularly among and beneath rosette-forming herbs and grasses (Figure 2.1B; Anderson and Williamson, 1976; Palis 1995a, 1997). After a period of approximately 22-36 days (Anderson and Williamson, 1976). Well-developed embryos may hatch into larvae following inundation by rising water levels within the wetland basins (Figure 2.1C) in the winter. The aquatic gilled larvae feed primarily on invertebrate zooplankton, particularly isopods, copepods, and amphipods (Whiles et al., 2004). Metamorphosis occurs between March and May, typically after an 11 to 18 week larval period (Palis, 1995a). Recent data from mesocosms indicate a larval period of 11 to 22 weeks (Pierson Hill, pers. comm. 2018). Juveniles (Figure 2.1D) normally disperse from ponds shortly after metamorphosing, but may stay near ponds during seasonal droughts (Palis, 1997). Juveniles, along with adults, are highly fossorial and spend much of their time in crayfish burrows or root channels until they reach sexual maturity (1 year for males; 2 years for females) and return to their natal pond to breed during the fall months (Petranka, 1998). Adults are known to live at least 4 years in captivity (Palis and Means, 2005), and Palis et al. (2006) attributed a decline in the number of adults captured in a 4-yr drift fence study to attrition without recruitment of juveniles during an extended drought. However, recent evidence suggests that frosted flatwoods salamanders can live up to 5 years (Pierson Hill, pers. comm. 2018) and the closely related reticulated flatwoods salamanders can live for as long as 9 years in the wild (Kelly Jones, Virginia Tech, pers. comm., 2019).

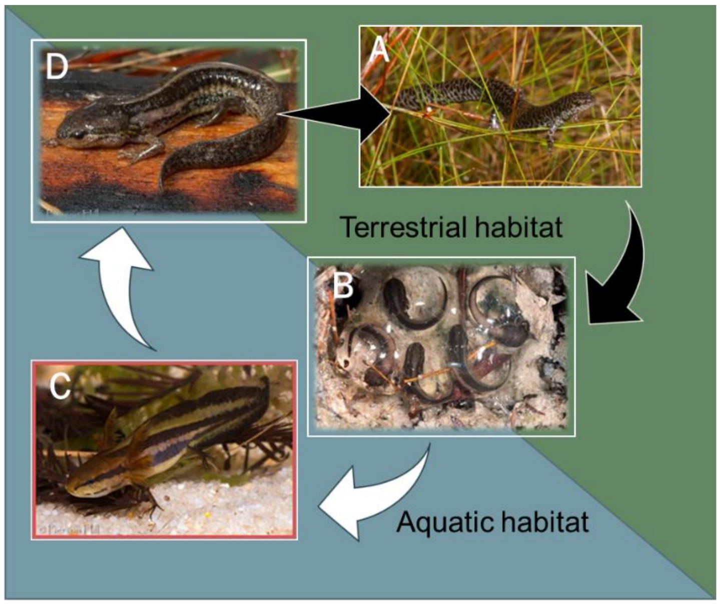

Figure 2.1. *Life cycle of the frosted flatwoods salamander. (A) Adult salamander in wire grass, Aristida sp. (B) Developing embryos oviposited on bare mineral soil. (C) Aquatic larva. (D) Newly metamorphosed terrestrial juvenile. All photographs courtesy of Pierson Hill.*

2.4 Habitat

Frosted flatwoods salamanders live within mesic (moderate moisture) to intermediate-mesic pine-dominated flatwoods/savanna communities. The dominant overstory species is generally longleaf pine (*Pinus palustris*) but this may differ in some localities. For example, in Florida, longleaf pine (*Pinus palustris*) is dominant in the Apalachicola National Forest, whereas in the St. Marks National Wildlife Refuge (SMNWR), slash pine (*Pinus elliotti*) dominates the coastal uplands.

Pine flatwoods/savannas are characterized by low flat topography and relatively poorly drained, acidic, sandy soil that becomes seasonally saturated. In the past, this ecosystem was characterized by open pine woodlands maintained by frequent fires. Naturally ignited by lightning during late spring and early summer, these flatwoods historically burned at intervals ranging from 1 to 4 years (Clewell, 1989), with an average fire return interval of approximately every two years (Noss, 2018) The topography can vary from nearly flat to gently rolling hills.

The groundcover of longleaf pine flatwoods/savanna ecosystem is typically dominated by warm season fire-adapted grasses, notably wiregrass (*Aristida stricta* [= *A. beyrichiana*]) (Kesler et al., 2003). Other herbaceous plants often found in the groundcover include toothache grass (*Ctenium aromaticum*), bluestems (*Andropogon* spp.), beakrushes (*Rhynchospora* spp.), pitcherplants (*Sarracenia* spp.), meadowbeauties (*Rhexia* spp.), and a variety of legumes. Low-growing shrubs, such as saw palmetto (*Serenoa repens*), gallberry (*Ilex glabra*), blueberries (*Vaccinium* spp.), and huckleberries (*Gaylussacia* spp.) co-exist with a highly diverse suite of grasses and forbs in the groundcover.

Flatwoods salamanders typically breed and deposit eggs in dry wetlands prior to being inundated with water (Anderson and Williamson, 1976; Hill, 2013; Powell et al., 2013; Gorman et al., 2014). These wetlands are generally characterized as acidic (pH 3.4 to 5.6), tannin-stained ephemeral wetlands that typically range in size from <1 to 10 acres, but may reach or exceed 30 acres (Palis, 1997; Safer, 2001). Ponds are often round or oval, but larger breeding sites may be quite irregular in shape. The basins are bowl or plate-shaped in profile and often perched above the normal water table on clay lenses (Wolfe et al., 1988). Pond depth fluctuates greatly, but is usually less than 0.5 meters (Palis, 1997) in microhabitats where larval salamanders are found. Ponds typically fill in late fall or early winter, and dry in late spring, or early summer. Summer thunderstorms may refill some ponds but most of these dry again before early fall, which performs the critical function of eliminating aquatic predator and competitor species, prior to the breeding season.

During breeding, gravid females select relatively open areas within wetlands that have lush carpets of fire-maintained wetland plant communities. Optimal sites have structurally complex and diverse assemblages of herbaceous and graminaceous vegetation, with little or no accumulated litter, peat, or muck. Characteristic plant species include the pipeworts (*Eriocaulon* spp), panic grasses (*Dichanthelium* spp), and various asters (*Bigelowia*, *Balduinia*, *Coreopsis*, *Erigeron*). Here, females are intercepted by searching for males, who deposit spermataphores on the ground that are then picked up by females (Hill, 2013). Following acceptance of spermataphores, females may spend several nights searching out concave micro-depressions within the vegetation for egg deposition. Small depressions likely reduce temperature fluctuations and minimize desiccation of developing embryos in the otherwise dry wetland (e.g. Gorman et al., 2009; Jones et al., 2012).

Following hatching, larval frosted flatwoods salamanders. Generally occupy shallow (<0.5 m) areas of wetlands that are dominated by emergent wetland grasses (e.g. *Rhynchopsora spp., Carex spp., Dichanthelium spp*.). Larvae seek refuge within the dense matrix formed by submerged grasses during the day and emerge at night to forage on zooplankton (Palis, 1995, Pierson Hill, pers. obs.).

Suitable wetlands tend to be distributed on the landscape in a clumped fashion. Occupied wetlands have an average of 6 and at least 2 suitable wetlands within a 0.5 km radius and an average of 15 (at least 7) suitable wetlands within a 1.5 km radius (Brooks et al. 2019a and

George Brooks, Virginia Tech, 2019, unpublished data). Further, occupied wetlands are connected in stepping stone arrangement to an average of 22 wetlands using a 1.5 km threshold dispersal distance (George Brooks et al. 2019a and George Brooks, Virginia Tech, 2019, unpublished data).

Under current conditions, the overstory within breeding ponds is typically dominated by pond cypress (*Taxodium ascendens* [=*T. distichum* var. *imbricarium*; Lickey and Walker, 2002]), blackgum (*Nyssa sylvatica* var. *biflora*), longleaf and slash pine, but can also include red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), sweetbay (*Magnolia virginiana*), and loblolly bay (*Gordonia lasianthus*). Canopy cover of occupied sites is typically moderate and ranges from near zero to almost 100% (Palis, 1997). The midstory, which is sometimes very dense, is most often composed of young of the aforementioned species, myrtle-leaved holly (*Ilex myrtifolia*), St. John's-worts (especially *Hypericum chapmanii* and *H. fasciculatum*), titi (*Cyrilla racemiflora*), sweet pepperbush (*Clethra alnifolia*), fetterbush (*Lyonia lucida*), vine-wicky (*Pieris phillyreifolius*), and bamboo-vine (*Smilax laurifolia*). Increasing midstory is negatively associated with herbaceous cover (and therefore larval occurrence), and optimal sites have sparse or absent midstory. When dry, breeding ponds burn naturally due to periodic wildfires (especially during late spring and summer), thus fire scars are frequent on live trees within the basin, and smaller trees and shrubs are often killed or top-killed. Depending on canopy cover and midstory, the herbaceous groundcover of breeding sites can vary widely, although larvae are most often associated with higher amounts of herbaceous cover which, on average, is greater than 40% coverage of the wetland (Gorman et al., 2009; Gorman et al., 2013; Enge et al., 2013). Most, but not all, breeding sites exhibit distinct vegetative zonation, with bands of different herbaceous plant assemblages in shallow versus deeper portions of the pond. The groundcover is dominated by graminaceous species, including beakrushes, sedges (*Carex* spp.), panic grasses (*Panicum* spp.), bluestems (*Andropogon* spp.), jointtails (*Coelorachis* spp.), longleaf three-awned grass (*Aristida palustris*), plumegrasses (Erianthus spp.), nutrush (*Scleria baldwinii*), hatpins (*Eriocaulon* spp.), Characteristic forbs may include milkworts (*Polygala* spp.), meadow beauties (*Rhexia* spp.), marsh pinks (*Sabatia* spp.), bladderworts (*Utricularia* spp.) and seedboxes (*Ludwigia* spp.).

There is a broad faunal association with these specific wetland habitats. Burrows of crayfish (genus *Procambarus*, principally) are a common feature, and provide important subterranean refugia for flatwoods salamanders and other animals (Neil, 1951). These ponds often harbor small fishes; the most typical species include pygmy sunfishes (*Elassoma* spp.), Eastern mosquitofish (*Gambusia holbrookii*), and banded sunfishes (*Enneacanthus spp.*) (Palis, 1997). Typical amphibian associates of flatwoods salamander larvae include southern leopard frog (*Rana sphenocephala*), ornate chorus frog (*Pseudacris ornata;* but not at SMNWR), and dwarf salamander (*Eurycea quadridigitata*) larvae, as well as larval and adult newts (*Notophthalmus viridescens*) (Palis, 1997).

Currently, remaining salamander populations struggle to persist in less than ideal habitat which may differ from what is presented above. For example, the interruption/disruption of natural fire cycles at many sites has led to greater canopy closure in the overstory of both the flatwoods uplands and ephemeral ponds (Bishop and Haas, 2005; Gorman et al., 2009; Gorman et al., 2013) and the shrub layers of both habitats have similarly increased (Gorman et al., 2013). This has resulted in a lower cover of herbaceous groundcover that is less diverse. Further, the herbaceous layer within the wetland may be obscured or non-existent, replaced with a dense layer of shrubs, such as titi, fetterbush, gallberry, saw palmetto, wax myrtle (*Myrica cerifera*) and/or dog hobble (*Leucothoe* spp.) due to fire suppression or exclusion from wetland basins (Gorman et al., 2013). This occurs primarily because even where prescribed burns do occur, they are most often conducted in winter and early spring when ponds would typically be flooded and less likely to burn (Bishop and Haas, 2005). To increase the opportunity for flatwoods salamander habitat to burn more effectively, land managers should diversify burning strategies (Bishop and Haas, 2005). Specifically, prescribed fire management should aim to mimic the seasonality (spring-summer) and weather conditions under which natural wildfires would burn through ephemeral wetlands. For example, if a growing season fire cannot be performed, another option may include burning uplands during the dormant season and return in the growing season to burn wetlands when they are dry (Gorman et al., 2009). Additionally, mechanical treatments can be coupled with fire to restore sites that have become too overgrown for fire alone to restore the site (Gorman et al., 2013). Other types of suboptimal habitat, such as roadside ditches and borrow pits, may have the physical and biotic characteristics of natural breeding sites and may occasionally be used by flatwoods salamanders, especially when located near natural breeding ponds (Anderson and Williamson, 1976; John Palis, 1995b; Stevenson, 1999; Tom Gorman and Carola Haas, Virginia Tech. Univ., unpublished data 2014).

2.5 Diet

Because of its complex life cycle, the diet of the frosted flatwoods salamander consists of aquatic prey consumed by larvae as well as terrestrial prey consumed by adults and juveniles. In 2004 (prior to the taxonomic separation of the two species), Whiles et al. (2004) documented that freshwater crustaceans comprised 96% of all invertebrates consumed by larval flatwoods salamanders. Prey consisted mostly of isopods (*Caecidotea*), amphipods (*Crangonyx*), cyclopoid copepods, and cladocerans (primarily *Simocephalus* and other daphnids). The numbers and proportions of cladocerans in stomachs differed among larval size classes, with higher numbers and proportions in small larvae than in medium or large-sized larvae (Whiles et al., 2004). Conversely, significantly higher numbers and proportions of isopods were consumed by larger larvae, compared to medium and small larvae (Whiles et al., 2004). As with prey abundance, Whiles et al. (2004) found that crustaceans, especially isopods and amphipods, dominated the prey mass in stomachs of larval flatwoods salamanders: on average, stomach mass was comprised of 65% isopods, 28% amphipods, and all other prey taxa comprised the remaining 7% of stomach mass. Whiles et al. (2004) found only one vertebrate prey item in the larvae they examined, a larval dwarf salamander, *Eurycea quadridigitata*. The abundance of isopods and amphipods as prey items of larval flatwoods salamanders suggests that larvae forage primarily in benthic detritus where these invertebrates are found (Whiles et al., 2004).

Terrestrial juvenile and adult flatwoods salamanders are primarily fossorial and spend much of their time in crayfish burrows and root channels, where they are known to consume earthworms (Goin, 1950). Although it has not been documented, it is likely that juveniles and adults also feed opportunistically on other terrestrial invertebrates (larval and adult insects, spiders, centipedes, isopods, and snails), as has been documented for other species of *Ambystoma* (Petranka, 1998).

2.6 Genetic Distribution

Goin (1950) was the first to recognize two distinct subspecies of flatwoods salamanders based on variation in morphology and color pattern: he classified populations in the eastern portion of the range as *A. cingulatum cingulatum* and those in the western Florida panhandle as *A. cingulatum bishopi*. This distinction was later challenged by Martof and Gerhardt (1965), and the premise that *A. cingulatum* was a single, undifferentiated species persisted in the literature until 2007. At that time, Pauly et al. (2007) conducted a range-wide phylogeographic analysis based on morphology, allozymes, and mitochondrial DNA and demonstrated that the "flatwoods salamander" actually consisted of two distinct species, with the faunal break occurring at the Apalachicola-Flint River. Based on these findings, the Service recognized two distinct species in 2009, frosted flatwoods salamander (*Ambystoma cingulatum*) and reticulated flatwoods salamander (*Ambystoma bishopi*) (74 FR 6700).

In addition to identifying *A. bishopi* as a genetically distinct species from *A. cingulatum*, Pauly et al. (2007) also documented two distinctive clades within *A. cingulatum*, with one occurring in the eastern Florida panhandle on the Gulf Coastal Plain and the other occurring on the Atlantic Coastal Plain. The separation of these two clades coincides with the Suwannee River (Pauly et al., 2007). Subsequent genetic analyses (using both mitochondrial and nuclear genes) provided strong support for the existence of two lineages within *A. cingulatum*: the Atlantic coastal plain populations were distinct from *A. cingulatum* populations immediately east of the Apalachicola-Flint Rivers in FL (e.g., the region of the Apalachicola National Forest) and that the South Carolina specimens were genetically similar to other *A. cingulatum* from the Atlantic Coastal Plain of Georgia and Florida (Pauly et al., 2012). Current lack of demographic connectivity due to habitat loss and fragmentation, artificial barriers (e.g., agriculture, roads, and hydrological alteration), and distance may now further prevent gene flow among remaining isolated populations, which are separated by distances that likely exceed the dispersal capabilities of the species, based on known dispersal distances of other ambystomatid salamanders (Semlitsch et al., 2017). The two clades are similar in their habitat management and recovery needs; they are very similar in appearance as well, and visual identification is not reliable enough to tell them apart consistently without knowledge of where they were captured.

2.7 Ecological Needs

The following ecological needs exist at the individual, population, and species levels (Table 2.1):

1. Individual Resource Needs (Figure 2.1)

    a. Eggs/embryos: Flatwoods salamanders breed and deposit eggs in wetlands that are not yet inundated with water (Anderson and Williamson, 1976; Hill, 2013; Powell et al., 2013; Gorman et al., 2014). Within wetlands, adults select open areas with complex and diverse stands of low-growing herbaceous vegetation for egg deposition. In this microhabitat, eggs are typically located in small depressions beneath the leaves of rosette-forming herbs or tucked deeply within the crowns of bunch grasses. These situations create a microclimate that likely minimizes temperature fluctuations and prevents the desiccation of developing embryos in the otherwise dry wetland. The selection by females of egg deposition habitat with complex herbaceous vegetation coincides with observations of all the other life stages of this species selecting habitat with complex and diverse stands of herbaceous vegetation within the breeding wetland (e.g., Gorman et al., 2009, Jones et al., 2012). After 22-36 days, well-developed embryos hatch into larvae with the onset of winter rains that flood oviposition sites (Anderson and Williamson, 1976; Palis, 1995a, 1997).
    b. Larvae: Larval flatwoods salamanders occur in acidic (pH 3.4 to 5.6), tannin-stained ephemeral wetlands (swamps or marshes) that typically range in size from <1 to10 acres (ac) (0.4 to 4.0 hectares [ha]), but may reach or exceed 30 ac (12 ha) (Palis, 1997; Safer, 2001). Occurrence of larvae is associated with low conductivity (William J. Barichivich, pers. comm. 2019; Walls et al., 2019). Water depth fluctuates greatly, but is usually 0.5 meters (m) (Palis, 1997) in areas where larval salamanders are found. Larvae are most often associated with higher amounts of herbaceous cover (Gorman et al., 2009, Gorman et al., 2013) which, on average, covers >40% of the wetland (Gorman et al., 2009, Gorman et al., 2013). A minimum wetland hydroperiod (length of time wetland retains water) of at least 11-18 weeks is needed to complete metamorphosis (Palis, 1995a). More recent data indicate a larval period of 76 to 153 days (11-22 weeks), with an average of 116 days (Pierson Hill, pers. comm. 2019).
    c. Juveniles: Juveniles normally disperse from ponds shortly after metamorphosing, but may stay near ponds during seasonal droughts (Palis, 1997). Juveniles, along with adults, are highly fossorial and spend much of their time in crayfish burrows or root channels until they reach sexual maturity (1 year for males; 2 years for females). Suitable, fire-maintained terrestrial habitat is necessary for dispersal, migration to/from adjacent wetlands, and sheltering during the non-breeding season.
    d. Adults: Individual *A. bishopi* have an average life span of 4-4.5 years, and can potentially live for at least 9 to 12 years (based on field observations and population models; Palis and Means, 2005; Kelly Jones, Virginia Tech, pers. comm 2018.; George Brooks, Virginia Tech, pers. comm. 2019), during which time they selectively breed in open canopy wetlands that are embedded within fire-maintained, longleaf pine-wiregrass habitat. Fire is necessary to maintain open canopies and areas of complex and diverse stands of herbaceous vegetation for egg deposition within breeding wetlands (Chandler et al., 2017, Brooks et al., 2019b). During the non-

breeding season, adults reside in the surrounding uplands. The presence and density of burrows may be important for growth and survival (Powell et al., 2015). Corridors of suitable habitat may be needed for dispersal and migration.

2. Population Needs
   a. Resource Needs and/or Circumstances: Factors that influence survival, reproduction, and juvenile recruitment affect abundance and, thus, the overall persistence of individual populations. Stochastic events, such as extremes in precipitation (droughts and floods), disease, hurricanes, storm surge, and introduction of predators and nonindigenous species can threaten individual populations.
   b. Population-level Resiliency: Small, isolated populations often have low genetic variation, leaving them particularly susceptible to the consequences of stochastic events. Effective population sizes (N_e) need to be adequate to prevent local populations from declining and disappearing. . In a metapopulation framework, genetic rescue (an increase in population growth and resilience due to immigration; Whiteley et al., 2015) from neighboring populations can increase genetic diversity, abundance, and effective population size. Thus, to maintain population resiliency, demographic and genetic connectivity among adjacent populations need to be maintained as well as ecosystem processes that promote longleaf pine-wiregrass habitat and adequate wetland hydrology.

3. Species Needs
   a. Resource Needs and/or Circumstances: Species experience declines and potential extinctions as the proportion of extirpated populations increases. Such extirpations occur with diminished resilience and increased isolation of individual populations. Fragmentation of the longleaf pine ecosystem, resulting from habitat loss and degradation, has disrupted both demographic and genetic connectivity within and among metapopulations across the landscape of the species' range. Large tracts of intact longleaf pine flatwoods habitat are fragmented by roads and pine plantations, leaving most flatwoods salamander populations widely separated from each other by unsuitable habitat.
   b. Ecologically appropriate fire regimes are necessary to maintain suitable habitat structure for the species. In the absence of natural wildfires, prescribed fire should closely mimic historical fire patterns in both frequency (1-3 years) and seasonality (April-July) in order to maintain the herbaceous structure of both uplands and wetlands.
   c. Species-level Redundancy: Processes that increase the number and connectivity of populations and metapopulations (barring an impassable barrier such as a perennial stream [64 FR 15691]) across the landscape are essential to achieve redundancy. Habitat restoration, reconnection of isolated populations, and recolonization of previously occupied sites may allow for decreased local extinction and increased local colonization, gene flow/demographic connectivity, and dispersal success (Semlitsch et al., 2017). Actions such as construction of new ponds, acquisition of

new habitat, corridor development, and assisted colonization may also increase redundancy of populations (Semlitsch et al., 2017).

d. Species-level Representation: Populations must be distributed in a variety of habitats throughout the range so that there are always some populations experiencing conditions that support some level of reproductive success. For *A. cingulatum*, re-establishment of extirpated populations throughout the species' historical range, particularly eastern Florida, Georgia, and South Carolina would increase the diversity of habitats and environmental conditions in which this species is found.

Table 2.1. *Life history stage and resource needs of the frosted flatwoods salamander.*

Life history Stage	**Resources and/or circumstances needed for individuals to complete each life history stage**	**Resource Function (BFSD*)**	**Information Source**
Eggs/Embryos	Bare mineral soil in herbaceous ecotone (maintained by frequent fire) between upland and wetland habitats for oviposition and development; seasonally appropriate rain events to flood oviposition sites and fill ponds	B	Anderson and Williamson, 1976; Palis 1995a, 1997
Larvae	Ephemeral wetlands with herbaceous cover and hydroperiods of at least 11-22 weeks to complete metamorphosis	F,S	Palis, 1995a; Gorman et al., 2009; Gorman et al., 2013 : Pierson Hill pers. comm. 2018
Juveniles	Fire-maintained upland habitat in longleaf pine-wiregrass ecosystems that is suitable for dispersal, migration from adjacent wetlands, and sheltering during the non-breeding season. Juveniles, along with adults, are highly fossorial and spend much of their time in crayfish burrows or root channels until they reach sexual maturity (1 year for males; 2 years for females)	F,S,D	Petranka, 1998
Adults	Upland habitat: same as for juveniles. Wetland habitat: suitable oviposition sites as described for eggs/embryos	B,F,S,D	Anderson and Williamson, 1976; Palis 1995a, 1997; Petranka, 1998

*B=breeding; F=feeding; S=sheltering; D=dispersal

2.8 Historical Range and Distribution

Historically, flatwoods salamanders (both species) occurred throughout the Coastal Plain of the southeastern U.S., across South Carolina, Georgia, Alabama, and the panhandle of Florida (Palis and Means, 2005). *Ambystoma cingulatum* occurred in Alachua, Baker, Bradford, Columbia, Duval, Franklin, Jefferson, Liberty, Marion, Nassau, Suwannee, and Wakulla counties in Florida; Atkinson, Ben Hill, Berrien, Brantley, Brooks, Bryan, Bulloch, Camden, Candler, Charlton, Clinch, Colquitt, Cook, Echols, Effingham, Emanual, Evans, Glynn, Grady, Irwin, Jeff Davis, Jenkins, Lanier, Liberty, Long, Lowndes, McIntosh, Mitchell, Screven, Tattnal, Thomas, Tift, Ware, Wayne, and Worth counties in Georgia; and Allendale, Bamberg, Beaufort, Berkeley, Charleston, Colleton, Dorchester, Hampton, Jasper, and Orangeburg counties in South Carolina. (William J. Barichivich, USGS, pers. comm. 2018; Figure 2.3). Over time and despite increased efforts to survey historical locations and find new populations, the combined range of both species (before the taxonomic split in 2009) of *A. cingulatum* has dwindled from 476 historical locations (i.e., mostly individual breeding sites) prior to 1999 to only 63 locations over the last five years (86.8% loss; Semlitsch et al., 2017).

2.9 Current Range and Distribution

When the 2009 final rule was published (74 FR 6700), there were 25 existing populations of *A. cingulatum*. A single population was defined as consisting of salamanders that use breeding ponds within 3.2 km (2 miles [mi]) of each other, barring an impassable barrier such as a perennial stream (64 FR 15691). Ecologically, for *Ambystoma*, the inter-pond distance used in this legal definition best describes a metapopulation, a set of local populations or breeding sites within an area, where typically dispersal from one local population or breeding site to other areas containing suitable habitat is possible, but not routine. The critical habitat designation took this into account by using known occurrences buffered by 457 m (1500 ft) as the base unit for analysis (74 FR 6700). Dispersal distance and genetic structure is not known for flatwoods salamanders, but the dispersal distances of eight other species of *Ambystoma* are considerably shorter than 3.2 km, ranging from 40 to 380 m (Scott et al., 2013). Similarly, Peterman et al. (2016, 2018) found that, for the ringed salamander (*A. annulatum*; the closest phylogenetic relative of flatwoods salamanders), breeding ponds within 2.09 km and 2.51 km of each other were connected, both demographically and genetically, respectively. Thus, the number of actual "populations" of *A. cingulatum*, as defined by the 2009 final rule, is likely conservative and inclusive of several different populations. Regardless, for consistency, we retain use of the term "population" herein as defined in the final rule.

Of the 25 populations recognized in 2009 (74 FR 6700), there were nine known and currently occupied breeding populations at the end of the 2014/15 breeding season (based on unpublished data from William J. Barichivich, USGS; Jana Mott, TNC; Kevin Enge, FFWCC; John Jensen, GADNR; and John Palis, Palis Environmental Consulting; Roy King, Ft. Stewart and Will Dillman, SCDNR, pers. comm. 2018). More recent, comprehensive data for these 25 populations are not available, and we review the current status in more detail in Chapter 3. Seven of these nine populations occur at Apalachicola National Forest and St Marks National Wildlife Refuge in Liberty and Wakulla Counties, FL, respectively (Table 2.2; Figure 2.2). A small population (one known breeding pond) remains on Fort Stewart, Liberty County, GA (Figure 2.2). Despite considerable sampling effort the status of the population on the Francis Marion National Forest in Berkeley County, SC remains uncertain as no observations of flatwoods salamanders have been made since 2010 (Figure 2.2). We have limited information on salamander occupancy on private lands, so our analysis of populations primarily focuses on public lands where sampling has been more rigorous since 2009.

RMU's boundaries were chosen to reflect relatively equal areas within the range, to more efficiently evaluate range wide recovery and management needs from a historical range perspective. This allows us to set targets throughout the range allowing for distribution across the range, and prevent too much clustering in certain areas and better reflect the historic habitat distribution. These four RMUs are divided on the basis of natural geographic boundaries, including ecoregions and watersheds, and aid in developing a recovery strategy that encompassed representative portions of the species historical range. The RMU boundaries were developed by selecting all Level 4 ecoregions that included historic records derived from a museum database query (Jamie Barichivich, USGS, pers. comm. 2019; Figure 2.2). Additionally, RMU boundaries were chosen to reflect rivers and other faunal breaks.

All but one currently extant metapopulation occurs in RMU 1, and one remains in RMU 3 at Fort Stewart Military Installation, GA. Our goal is to establish 25 metapopulations in each of the 4 RMUs and cover as much of the former range and distribution as appropriate.

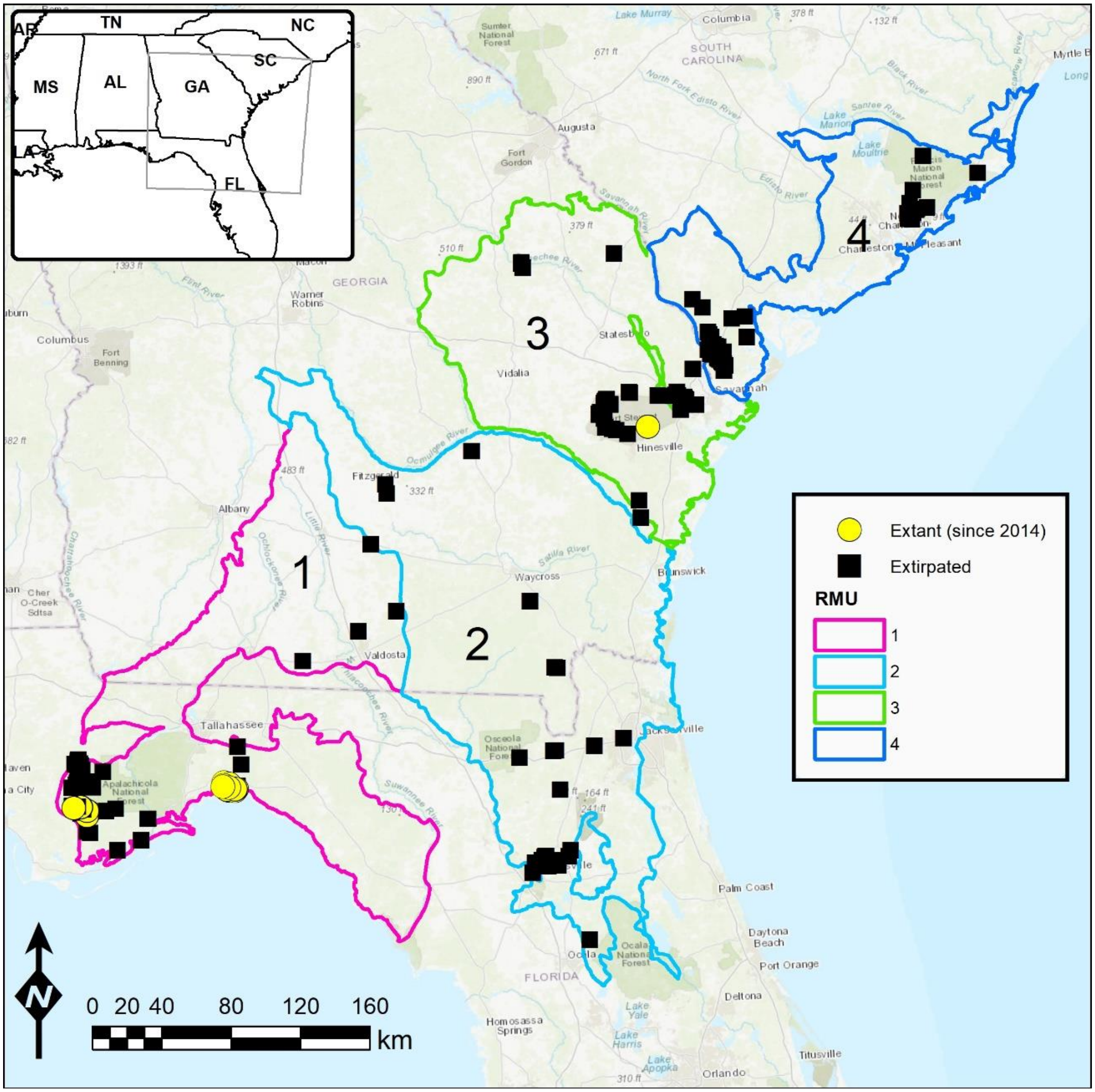

Figure 2.2 *Historic and current locations of frosted salamanders within the RMU context*

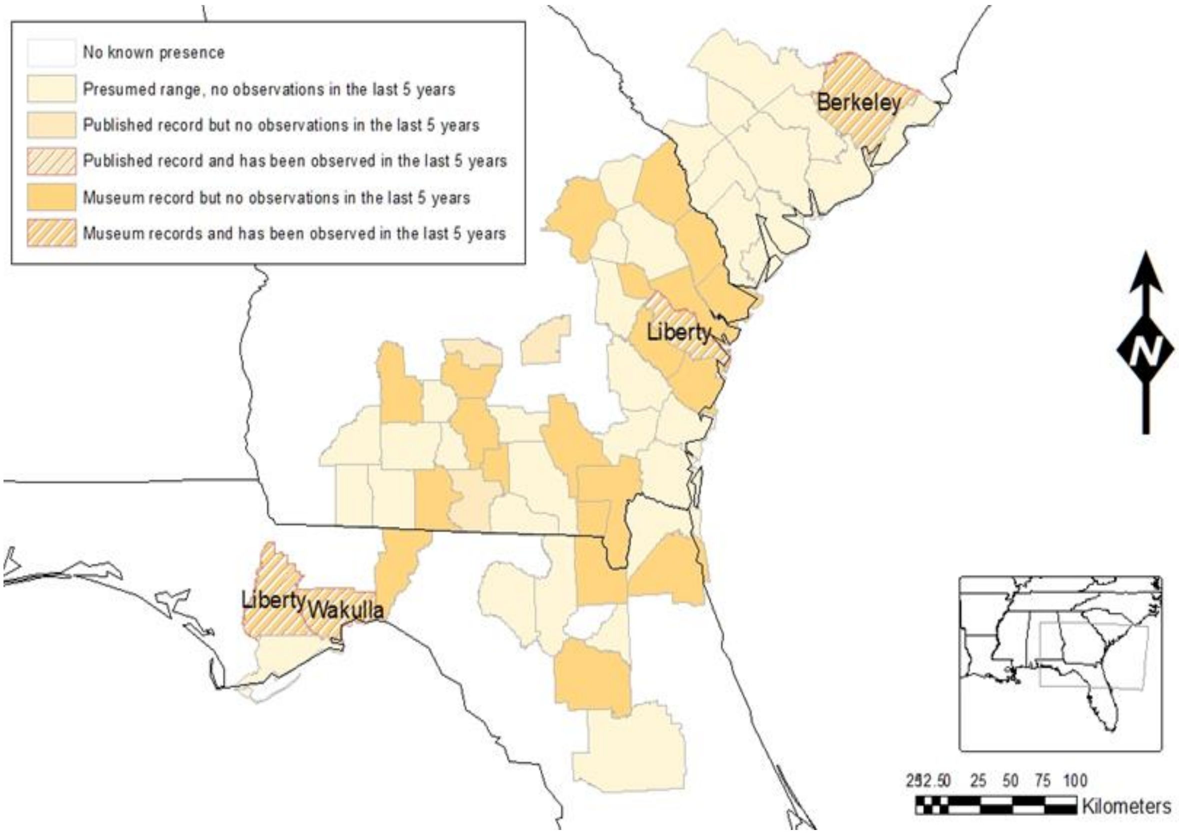

Figure 2.3 *Geographic range of the frosted flatwoods salamander (W.J. Barichivich, USGS, 2018, pers comm.). "Last 5 years"=2010-2015.*

2.10 Land Ownership

Based on our GAP/GIS analysis and public land database layers (from ESRI and the Florida Natural Areas Inventory database), there is a total of 9,297 ha, partitioned among 19 critical habitat units, designated for the frosted flatwoods salamander, with 73.7% of this habitat located on public lands (Figure 2.3; Walls et al., unpubl. data). An analysis of potentially suitable habitat outside of designated CHUs is currently underway (J. Bracken, unpubl. data).

Table 2.2. *The number of populations of A. cingulatum from the 2009 USFWS final rule (74 FR 6700) and the documented occupancy of those populations from 2010 to 2015.Numbers in parentheses represent the total number of populations in each area; ? = uncertainty regarding whether sites have been surveyed in the last five years; NF = National Forest; NWR = National Wildlife Refuge; WMA = Wildlife Management Area.*

	Populations according to 2009 final rule	**Observations from 2010 to 2015[a]**
Florida (15)		
Apalachicola NF	10	5/10
St. Marks NWR	2	2/2
Osceola NF	1	0/1
Aucilla WMA/private	1	0/1
Private property, Baker Co.	1	?
Georgia (6)		
Fort Stewart	5	1[b]/5
Townsend Bombing Range	1	0/1
South Carolina (4)		
Private Properties, Jasper Co.	2	?
Francis Marion NF	1	1[b, c]/1
Santee Coastal Preserve, Charleston Co.	1	0/1
Total	25	9/22

[a] number of populations where *A. cingulatum* was observed /number of populations sampled
[b] Population appears to be reduced to a single breeding pond
[c] The last observation of flatwoods salamanders from this site occurred in 2010

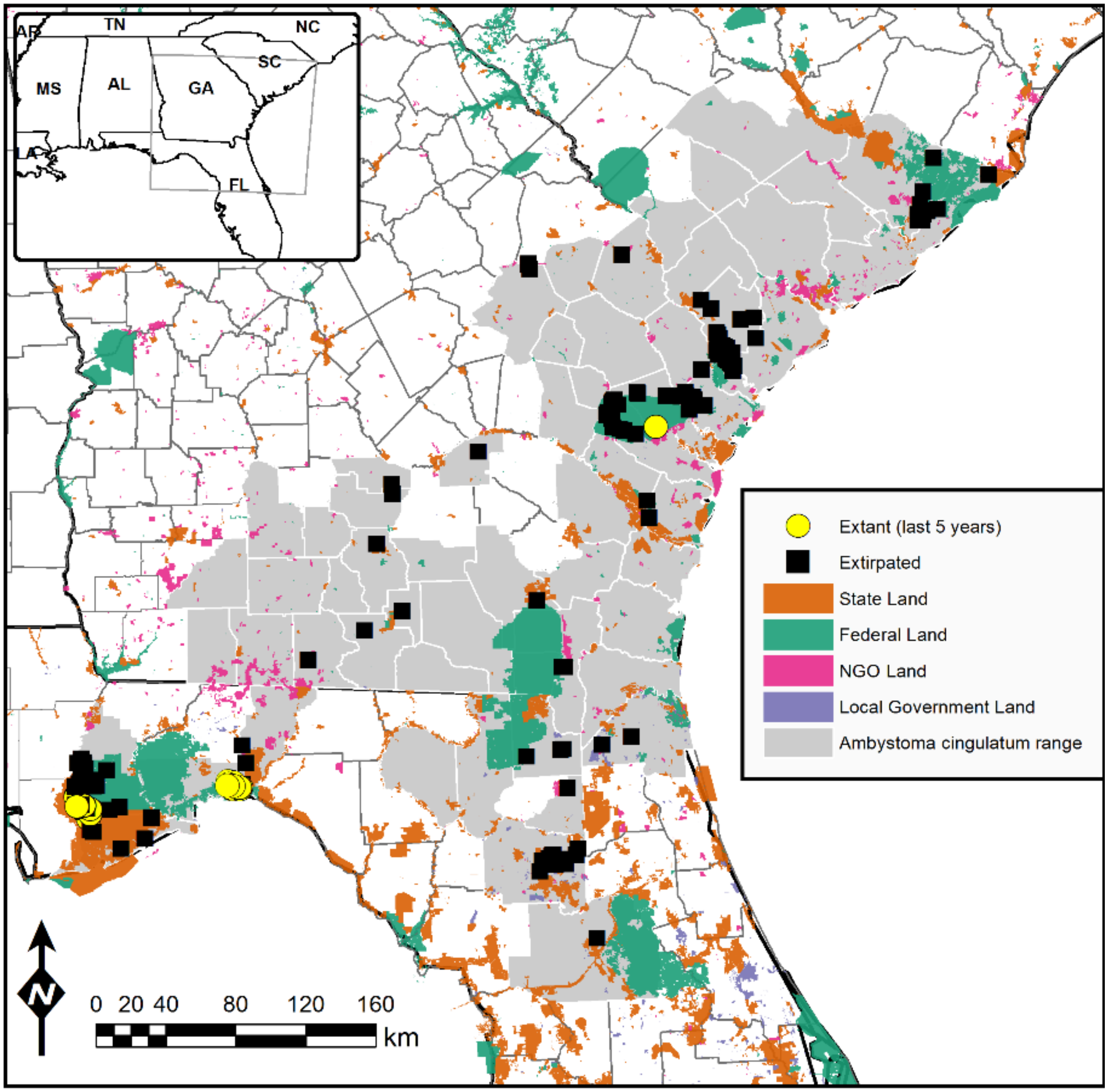

Figure 2.4 *Land ownership and historical localities of the frosted flatwoods salamander (K. O'Donnell, USGS, unpubl. data).*

CHAPTER 3 – CURRENT SPECIES CONDITION

3.1 Resiliency

3.1.1 Resiliency Metrics

Resiliency is a reflection of a population's health and ability to withstand stochastic events (e.g., drought, storms, and disease outbreaks). Key stressors (e.g., habitat loss and degradation, climate change, contaminants, and invasive species) may lower resiliency by inducing physiological stress in members of a population. In turn, stress suppresses immune systems, which can, for example, increase susceptibility to disease outbreaks, impair growth and survival, compromise body condition, and increase vulnerability to competitors and predators – all of which make populations more vulnerable to declines (Figure 3.1). Resiliency is generally measured using demographic factors that reflect population health, such as fecundity, survival, population size and growth. For many imperiled species, however, including frosted flatwoods salamanders, data are not available on either population health or demography; thus, alternative measures of resiliency must be used.

Habitat quality is known to influence dispersal, survival, and genetic variation in amphibians (Rothermel, 2004; Rothermel and Semlitsch, 2006; Richter et al., 2013) and, thus, may be considered a correlate of population health. We assessed resiliency in the frosted flatwoods salamander based on measures of habitat quality at each extant site. Similar landscape characteristics have been used to assess resiliency (in response to a drought) in other species (Oliver et al., 2013).

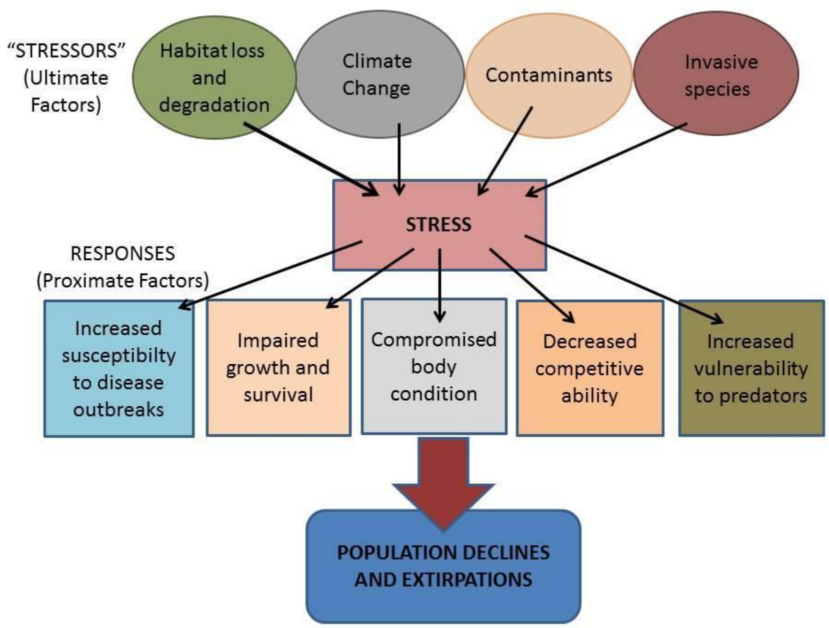

Figure 3.1. *Conceptual model of the mechanisms by which key stressors decrease population resiliency, thus leading to population declines. Stressors induce physiological stress which, in turn, suppresses immune responses of members of a population. Responses to such immunosuppression – either individually or in combination – can then result in population declines and extirpations. Modified from Hayes et al., 2010.*

3.1.2 Habitat-based Assessment of Resiliency: Expert Elicitation of Land Managers and Species Experts

We elicited habitat assessments from land managers and species experts that manage habitat for, and/or conduct research with, *A. cingulatum* on five public properties (Apalachicola National Forest, St. Marks National Wildlife Refuge, Flint Rock properties, Fort Stewart [Georgia], and Francis Marion National Forest [South Carolina]). For each property, we asked participants to assess the current number of extant breeding sites on their property according to 6 resiliency categories: (1) extent of woody vegetation in understory of upland habitat; (2) quality and composition of the wetland basin overstory; (3) presence and composition of the wetland midstory vegetation; (4) type of wetland understory vegetation and presence of organic duff/peat layer in basin; (5) adequacy of wetland hydroperiod for completion of metamorphosis; and (6) burn frequency/burn season for the compartment in which breeding sites are located.

At present, we summarized this information as participants' overall assessment of resiliency of flatwoods salamander habitat on the four properties for which we received results (Table 3.1). We received a total of twelve responses for four properties (two for ANF, four for FR, one for FS, and five for SMNWR). We did not receive responses from managers at Francis Marion National Forest (South Carolina), this location has not detected a flatwoods salamander since 2010, and that property is not included further. Data were summarized as percentages because individual participants responded only to the ponds familiar to them and varied by respondent. Overall, only 13 breeding ponds were assessed as highly resilient (either a 4 or 5) and 36 ponds were moderately resilient. The remaining 26 ponds had low resiliency (either a 1 or 2).

Table 3.1. *Land manager assessments of the overall resiliency of flatwoods salamander habitat on their property. Responses are the percent of extant breeding ponds that fit into each point on the 5-point resiliency scale: (1) extremely low resiliency; (2) low resiliency; (3) moderate resiliency; (4) high resiliency; or (5) extremely high resiliency. ANF =Apalachicola National Forest; SMNWR = St. Marks National Wildlife Refuge; FR = Flint Rock properties; FS = Fort Stewart (Georgia)*

Property	**Extremely Low**		**Low**		**Moderate**		**High**		**Extremely High**		**Total Ponds**
	Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	
ANF	34	33.8	16	8.6	47	39.7	3	2.6	0	0.0	38
SMNWR	3	5.3	17	17.3	56	32.7	26	24.4	10	16.7	30
FR	19	32.5	13	21.7	35	25.3	25	25.0	0	0.0	6
FS	0	N/A	0	N/A	100	N/A	0	N/A	0	N/A	1
Total											75

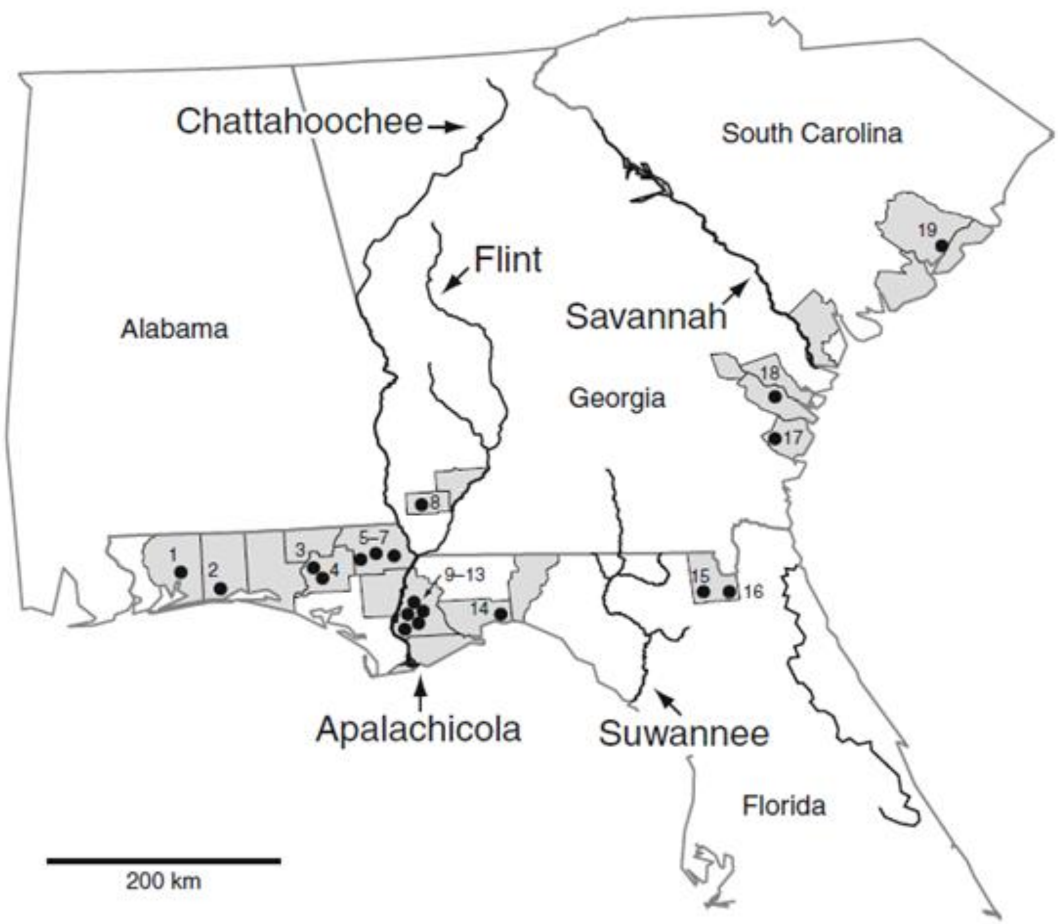

Figure 3.2. *Localities (indicated by numbers) at which flatwoods salamanders were collected for molecular and morphological analysis by Pauley et al. (2007, 2012).*

3.2 Representation

Representation describes the ability of a species to adapt to changing environmental conditions over time as characterized by the breadth of genetic and environmental diversity within and among populations. Representation may be measured in terms of the breadth of geographic, genetic, or life history variation among populations, which enhances a species' ability to adapt to changing environmental conditions. Based on molecular (mitochondrial DNA) and morphological analyses, Pauly et al. (2007) found support for three distinctive clades of flatwoods salamanders: *A. bishopi* to the west of the Apalachicola and Flint Rivers in Florida and Georgia, and two lineages east of the Apalachicola and Flint Rivers. The two eastern groups were the eastern Gulf Coastal Plain group of populations between the Suwannee and Apalachicola Rivers (Pops. 9–14 in Figure 3.2), which they labelled the Eastern Panhandle

Clade, and another from the Atlantic Coastal Plain, which these authors called the Atlantic Coastal Plain Clade (Pops. 15–18 in Figure 3.2). Sequence divergences between the Eastern Panhandle Clade and the Atlantic Coastal Plain Clade ranged from 1.2% to 1.6% (for comparison, divergences between *A. cingulatum* and *A. bishopi* ranged from 5.6% to 6.2%, whereas divergence levels within each clade never exceeded 0.4% (Pauly et al., 2007). Importantly, there was no detectable geographic structure within any given clade, with some haplotypes shared across the clade's geographic range (Pauly et al., 2007). With acquisition of salamanders from South Carolina (which were not available for their earlier analyses) and additional molecular analyses based on nuclear datasets, Pauly et al. (2012) further clarified the divergence between the Eastern Panhandle Clade and the Atlantic Coastal Plain Clade, indicating that specimens from South Carolina (north of the Savannah River) were closely related to those from eastern Georgia and Florida (south of the Savannah River), suggesting that the Savannah River is not a historical barrier to gene flow between populations on either side of the river.

Because Pauly et al. (2007, 2012) determined that there are two genetically distinctive lineages within *A. cingulatum*, yet no detectable geographic structure within either clade, we consider that there are two representation units (i.e., an area that encompasses a group of populations that share similar genetic and life history traits and which occupy geographically and ecologically comparable locations) for this species – the Eastern Panhandle Unit and the Atlantic Coastal Plain Unit.

However, based on our current knowledge, only one confirmed extant population remains within the Atlantic Coastal Plain Unit (at Fort Stewart Military Installation). The last observation of a frosted flatwoods salamander in South Carolina was in 2010. At least one individual reticulated flatwoods salamander is known to have lived for 9 years in the wild (Kelly Jones, pers. comm. 2018), suggesting the potential for *A. cingulatum* to still be present at the last known site in South Carolina. However, this age likely does not represent the average longevity of flatwoods salamanders (George Brooks, unpubl. data 2019). Thus, the time that has elapsed since 2010 may exceed the normal lifespan of this species, suggesting that *A. cingulatum* may have been extirpated from South Carolina. The lack of multiple, resilient populations within this representation unit increases this species' risk of extinction; i.e., without multiple groups of genetically diverse populations that occupy a variety of habitats across the species' range, the ability of *A. cingulatum* to adapt to changing environmental conditions over time is compromised. Re-establishment of extirpated populations throughout this species' historical range, particularly in Georgia, South Carolina and northeastern Florida, would increase the diversity of habitats and environmental conditions in which the frosted flatwoods salamander is found. Our RMU boundaries (Figure 2.2) reflect this approach.

3.3 Redundancy

Redundancy refers to a species' ability to withstand catastrophic events and is enhanced by the presence of numerous, resilient populations (and their connectivity) within representative units. To measure redundancy, we used the number of active breeding ponds within each representation unit, and the mean (and standard deviation) distance of ponds to the next nearest known breeding pond within that representation unit.

Semlitsch et al. (2017) compiled locality information across the former historical range (of both species of flatwoods salamanders combined) and showed that, over time, the combined range of these two species dwindled from 476 historical locations prior to listing in 1999 to only 63 locations from 2010 to 2015 (86.8% population loss; Fig. 3.3A-C; Bevelhimer et al., 2008; Pauly et al., 2012). Semlitsch et al. (2017) also showed that mean inter-pond distance increased from 8.9 km prior to 1999 (before USFWS listing of what was then a single species, *A. cingulatum*), to 12.7 km from 2000 to 2009 (post-listing period), and to 28.3 km from 2010 to 2015 (post-taxonomic split into *A. cingulatum* and *A. bishopi* [Pauly et al., 2007] and designation of critical habitat [74 FR 6700]).

Because individual salamanders probably do not disperse more than 1–2 km within a generation and multi-generation gene flow likely is limited to 5–10 km or less for most ambystomatid species (Semlitsch, 2008; Peterman et al., 2015), loss of flatwoods salamander populations over time, even prior to 1999, has evidently created severe isolation that is a critical component of an increased extinction risk. The potential for metapopulation dynamics (i.e., the natural exchange of individuals among discrete populations [via migration or dispersal] in the same general geographical area: Akçakaya et al., 2007) is now extremely limited. Studies have shown that the loss of fragmented populations is common, and recolonization is critical for their regional survival (Fahrig and Merriam, 1994; Burkey, 1995). Amphibian populations may be unable to recolonize areas after local extinctions due to their physiological constraints, relatively low mobility, and site fidelity (Blaustein et al., 1994).

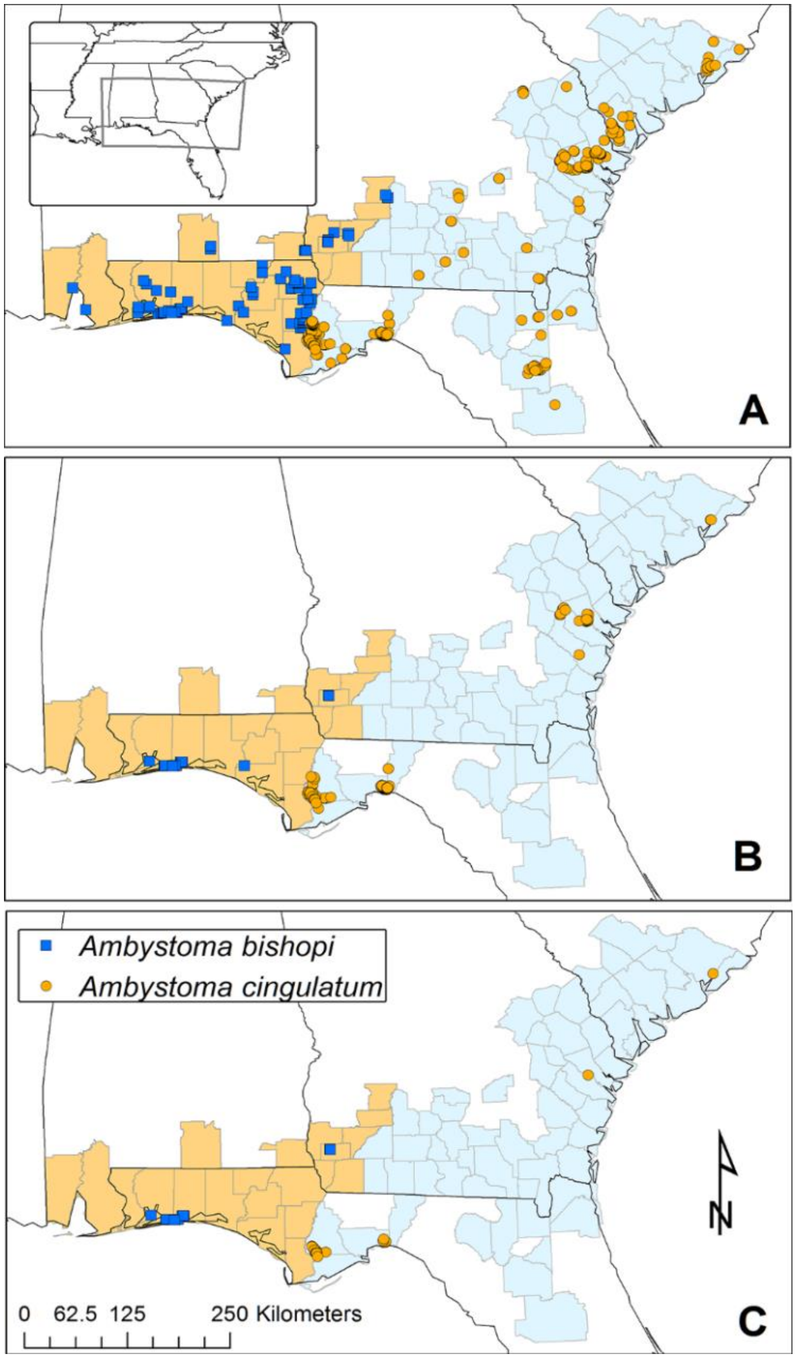

Figure 3.3. *Known localities of flatwoods salamanders (frosted flatwoods salamander, Ambystoma cingulatum, and reticulated flatwoods salamander, Ambystoma bishopi) over three time periods. A) All known records, B) 2000-2009 (post-listing), and C) 2010 to 2015 (post-taxonomic split). Orange circles = A. cingulatum and blue squares = A. bishopi. Shaded counties indicate the range of each species. From Semlitsch et al. (2017).*

3.4 Approach to Assessment of Redundancy and Representation

Although we have very limited information about historic population sizes, in recent decades most wetlands seem to be occupied by a fairly small number of breeding adults. Further, flatwoods salamanders exhibit sporadic recruitment, as a result of environmental stochasticity that impacts breeding conditions in a given year. Consequentially, a PVA conducted for the species revealed a high probability of extinction for individual populations, even over moderate timespans. For these reasons, conservation of this species likely will rely heavily on redundancy (that is a large number of small population clusters).

In determining our estimate of the number of resilient populations that are needed to have a low extinction risk into the future, we used the following methodology. Although information is limited, we used the distribution of wetlands and uplands occupied on Eglin Air Force Base (George Brooks and Nick Caruso, Virginia Tech, 2019, unpublished data), including consideration of wetlands that have become extirpated in the last 30 years, to describe conditions required for a landscape to support a functional population and metapopulation. By combining this information with probability of extinction within 40 years and 50 years as estimated through a PVA (George Brooks, Virginia Tech, 2019 unpublished data), we estimated requirements for population resiliency and scaled up to rangewide levels. This value was based on PVA estimates for ponds 4 and 5 at Eglin AFB, which is the only reliable information available. Our assumptions and approach were as follows.

- Assume a single breeding wetland in isolation is not resilient. Long-term resiliency is achieved through metapopulation dynamics. A network containing multiple occupied ponds within the known dispersal distance of the species would be considered a metapopulation. Based on data at Eglin Air Force Base (George Brooks and Nick Caruso, Virginia Tech, 2019, unpublished data), populations that have persisted over time occur in clusters of at least 3 regularly occupied wetlands within a 0.5 km radius. The entire network contains suitable upland habitat between occupied wetlands. We define "regularly occupied" as occupied at least once every 3 years, and all recently occupied wetlands on Eglin meet or exceed this criterion for the last 10 years.
- Assume half of the metapopulations would exhibit independent dynamics from others, so one would need twice as many as if all were independent to reduce risk of extinction. This assumption accounts for the fact that years of reproductive failure or success could vary based on spatially variable rainfall patterns, wetland basin shapes, etc. This assumption is poorly supported, and we have observed much higher levels of synchrony. Creating the conditions that promote more asynchrony should be a goal within each RMU. (See section 3.6 for more on methods to achieve asynchrony.)
- PVA results indicate that each metapopulation has a 45% probability of extinction within 40 years and a 50% probability of extinction within 50 years. Using this value is a cautious approach as a metapopulation containing a cluster of at least 3 regularly occupied wetlands may have lower probability of extinction than a single wetland within a cluster. However, since the wetlands at Eglin AFB from which the estimates were

derived occur in a cluster, and cycle fairly synchronously, this is an appropriate assumption.

- We elicited extinction risk tolerances of 17 members of the recovery team following a methodology similar to that described by Regan et al. (2013). The elicitation process resulted in four equally supported estimates of extinction risk tolerance of 0.0000000000000000000000000000000000385 in 40 years, 0.0002479 in 40 years, 0.0000000000000000000000026697 in 50 years, and 0.0026818 in 50 years.
- For each scenario, we could calculate the number of metapopulations required to stay below the desired extinction risk. For example, if the tolerable risk level is 0.000247 over 40 years, then we would need 11 independent metapopulations (i.e. 22 actual metapopulations) to reduce global extinction risk below this threshold. We performed the same calculation for the top 4 supported scenarios, and took the weighted average of the number of metapopulations required to achieve each tolerable risk level. This resulted in estimates of the number of metapopulations needed of 206, 158, 22, and 18, for an average value of 101 resilient metapopulations. If the metapopulations are distributed equally across the historic range, that would result in approximately 25 metapopulations (rounding off remainders of the 4 RMU numbers) in each of the 4 RMUs.

With the goal of restoring 34 metapopulations in each of the RMU's there was a need to know there was adequate habitat to support our targets. Figure 3.4 and Figure 3.5 compare the potential habitat from the (U.S. Geological Survey - Gap Analysis Project, 2017, with the Florida Natural Areas Inventory, 2017 Species Distribution Model for Frosted Flatwoods Salamander Frosted Flatwoods Salamander (*Ambystoma cingulatum*) to the RMU boundaries and demonstrates there is enough habitat to support our targets (Table 3.2 and 3.3). The dark red shaded areas represent the total potential habitat that could possibly support metapopulations of reticulated flatwoods salamanders. We used a 500-m radius circle (194 acres) as an estimate of the habitat require to support a metapopulation (see description of this in Sec 2.8) and overlaid these on the model.

Comparing the two habitat models, there is a significant difference in the amounts of suitable habitats. The USGS uses 2001 land cover data, while the FNAI model uses 2017 data. Both models demonstrate that enough habitat exists to support our recommended total of 101 metapopulations. Our minimum target of 101 metapopulations, when distributed over the 4 RMU's is clearly less than what the potential habitat suggests could be supported. However, not all of the dark shaded habitat may be available for recovery purposes as some portions are privately owned, the habitat has been significantly altered/drained, or other factors. The two figures represent ample suitable habitat to support enough resilient metapopulations to be potentially delisted.

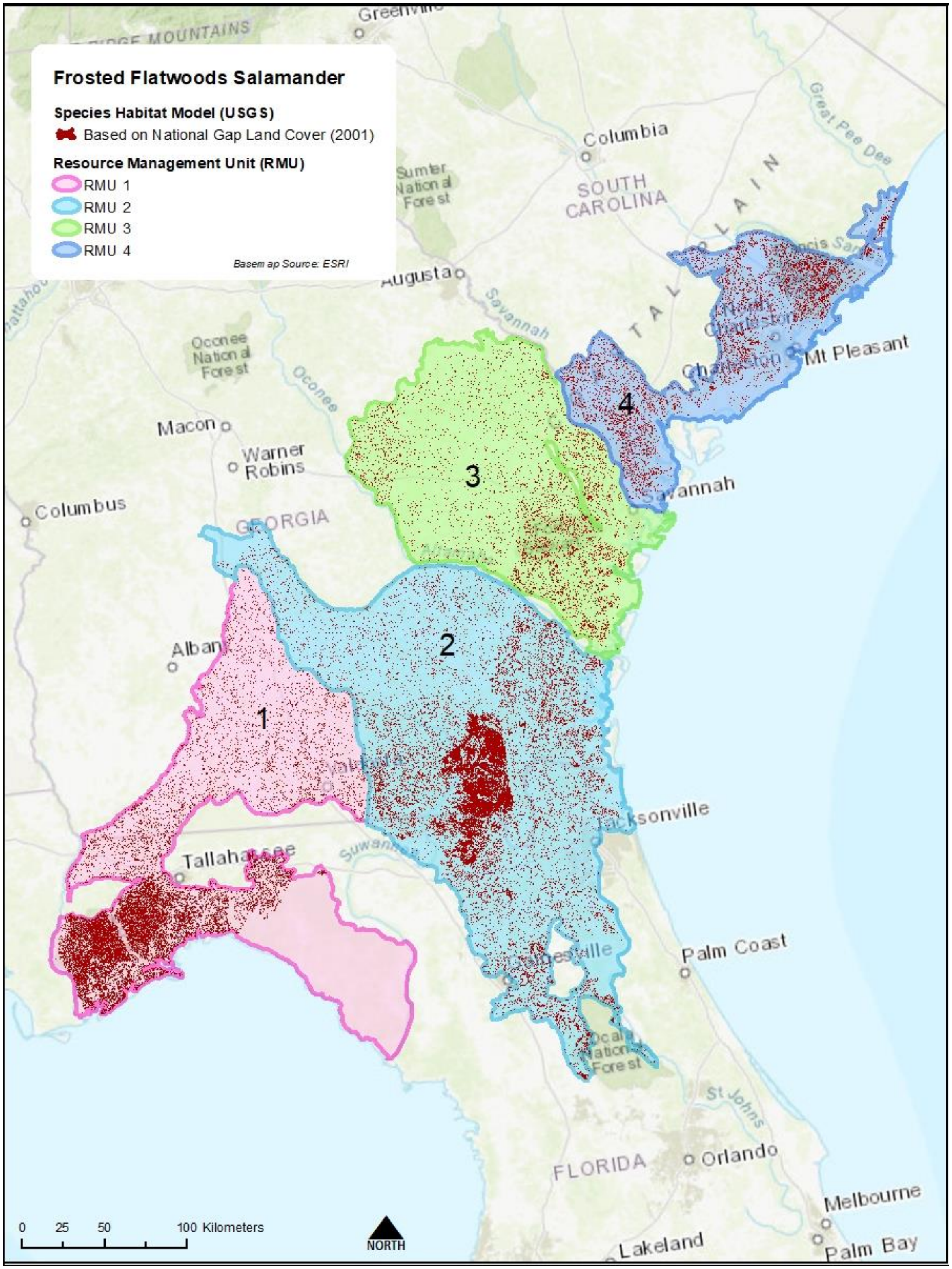

Figure 3.4 *Potential suitable habitat within each RMU boundary using USGS habitat model.*

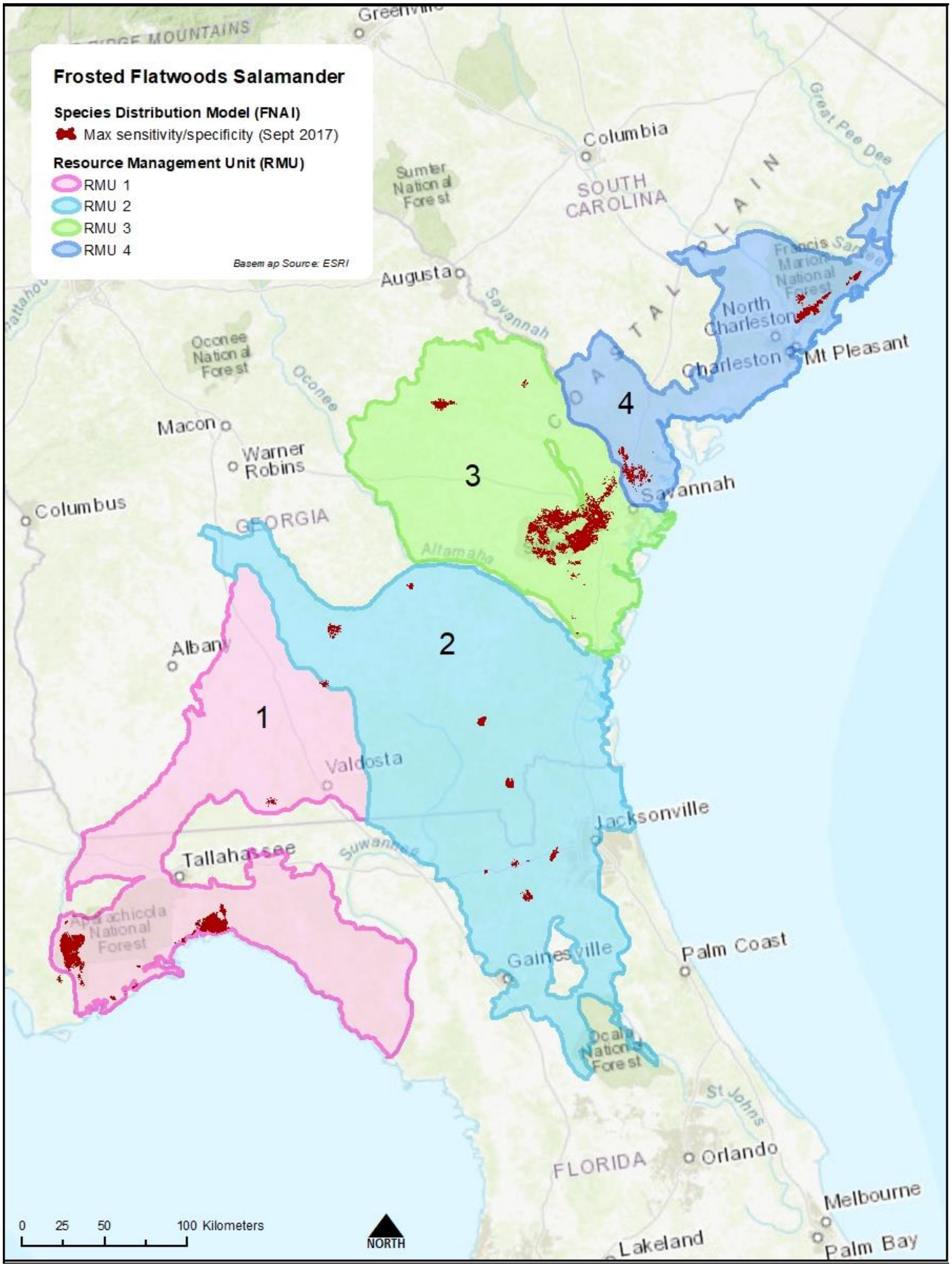

Figure 3.5 *Potential suitable habitat within each RMU boundary using FNAI habitat model.*

Table 3.2 *Theoretical number of metapopulations supported if all suitable habitat were used for salamander recovery using USGS model.*

Unit	Area (Acres)	Area (m^2)	Theoretical Number of Metapopulations Supported
RMU 1 *(~75% rmu/data overlap)*	1,008,448	4,082,667,300	5,198
RMU 2 *(~90% rmu/data overlap)*	1,335,688	5,405,346,000	6,882
RMU 3 *(~95% rmu/data overlap)*	363,837	1,472,396,400	1,875
RMU 4	448,979	1,816,956,900	2,313

Table 3.3 *Theoretical number of metapopulations supported if all suitable habitat were used for salamander recovery based on the FNAI model.*

Unit	Area (acres)	Area (m^2)	Theoretical # of Metapopulations Supported
RMU 1	114,405	462,982,500	589
RMU 2 (~90% rmu/data overlap)	40,790	165,071,700	210
RMU 3	2,158,518	873,522,900	1,112
RMU 4	61,557	249,114,600	317

3.5 Existing Regulatory Mechanisms

There are no existing regulatory mechanisms for the protection of the upland habitats where frosted flatwoods salamanders spend most of their lives. Section 404 of the Clean Water Act is the primary Federal law that has the potential to provide some protection for the wetland breeding sites of the frosted flatwoods salamander. However, due to case law (Solid Waste Agency of Northern Cook County (SWANCC) v. U.S. Army Corps of Engineers 2001; Rapanos v. U.S. 2006) and current practice, isolated wetlands are no longer considered to be under Federal jurisdiction (not regulatory wetlands). Wetlands are only considered to be under the jurisdiction of the Corps if a ‘‘significant nexus’’ exists to a navigable waterway or its tributaries.

Currently, some Corps Districts do not coordinate with the Service on flatwoods salamanders and, because isolated wetlands are not considered under their jurisdiction, they are often not included on maps in permit applications (Leibowitz and Brooks 2008). We are aware of two isolated wetlands that supported flatwoods salamander populations that have been lost since 2006 under this scenario. Longleaf pine habitat management plans have been written for public lands occupied by the frosted flatwoods salamander. They include management plans for State-owned lands and integrated natural resource management plans (INRMPs) for Department of Defense lands. Most of the plans contain specific goals and objectives regarding habitat management that would benefit frosted flatwoods salamanders including prescribed burning. However, because multiple-use is the guiding principle on most public land, protection of the flatwoods salamander may be just one of many management goals including timber production and military and recreational use. Implementation of the plans has often been problematic due to financial and logistic constraints. In addition, the plans do not provide assured protection from habitat destruction or degradation from land use changes (e.g., the proposed road on Eglin AFB and Hurlburt Field where *A. bishopi* is located), although ESA section 7 rules still apply (USFWS & NMFS 1998). At the State and local levels, regulatory mechanisms are limited.

The Clean Water Act (CWA) covers ephemeral wetlands, when they impact downstream waters and, in many cases, wetlands used by flatwoods salamanders are connected to downstream waters. However, it is unclear how these newly released regulations will aid in recovery of flatwoods salamanders. On 21 April 2014, the US Environmental Protection Agency and the US Army Corps of Engineers, in response to the SWANCC and Rapanos decisions, proposed clarifications to the CWA that would affect which types of waters would be considered jurisdictional under the Act (see US Army Corps of Engineers and US Environmental Protection Agency “Definition of Waters of the United States under the Clean Water Act,” CFR Docket ID No. 79 FR 22188). The clarifications (as of this writing) include reasserting CWA jurisdiction to wetlands adjacent to (i.e., bordering, contiguous, and neighboring) jurisdictional lakes, rivers, and streams. Furthermore, wetlands that are other waters, or those that are nonadjacent to waters of the United States, will have jurisdiction assessed on a case-by-case basis. The proposed regulations also allow the evaluation of other waters either alone or in combination with other similarly situated waters in the region to determine whether they significantly affect the chemical, physical, or biological integrity of traditional navigable waters, interstate waters, or the territorial seas. Other waters are similarly situated when they perform similar functions and are located sufficiently close together or sufficiently close to a water of the United States. The fact that CWA jurisdiction may be extended to geographically isolated wetlands (GIWs) on the basis of a watershed assessment of connectivity and the effect of GIWs on downstream waters suggests that watersheds in regions with large amounts of functioning GIWs (such as the prairie pothole region of the Upper Midwest and Canada, California vernal pools, Carolina bays and cypress ponds of the southeastern United States and other GIWs) may gain CWA protections under these new rules should they be finalized.

There are few, if any, range-wide mechanisms in place to adequately protect ephemeral wetlands, like those necessary for successful flatwoods salamander breeding. This includes the breeding ponds themselves as well as the surrounding upland habitat. The exceptions to this are the federally designated critical habitat units for *A. cingulatum* but this only applies to federal actions, not to state or private sector actions. Other landowner-specific tools are available but these need to be developed and implemented at each property (e.g., safe harbor agreement; see www.fws.gov/endangered/landowners/).

3.6 Current Conservation Measures

The frosted flatwoods salamander has experienced an 86.8% loss of historic populations between 1999 and 2015, placing them at imminent risk of extinction. To prevent their complete extinction and the disappearance of wild populations from their range, it will be necessary to carry out captive breeding, reintroductions and/or translocations to suitable habitat. In August 2014, a structured decision making (SDM) workshop was held with key stakeholders to make decisions (for both species combined) regarding how best to employ captive breeding, as well as reintroductions and/or translocations of individuals to suitable habitat, to minimize the risk of extinction of these species. As described in O'Donnell et al. (2017), the workshop participants developed four fundamental objectives: maximize (1) persistence (time to extinction); (2) viability (number of populations); (3) land steward cooperation; and (4) minimize costs. The group identified a number of alternative actions designed to achieve these objectives. The actions were then grouped into several strategies ("portfolios"): (A) a "do nothing" option; (B) in situ translocations only; (C) establishing three captive populations (no animals released); (D) establishing two captive populations plus reintroducing captive-bred larvae into 3 to 5 ecologically-suitable, unoccupied historic sites; and (E) establishing 3 captive populations, conducting reintroductions at 6 to 8 sites, including 3 to 5 sites restored/constructed for purposes of reintroduction. The use of a stochastic population viability model allowed the group to estimate long- and short-term extinction probabilities under each alternative scenario (McGowan et al., 2014). Assuming that breeding in captivity is successful for these species, the group projected that a "realistic maximum number of animals" (50 breeding pairs per facility, generating 2500 offspring per year) could be produced in just five years.

We evaluated the five alternative strategies against our four objectives using a consequence table and found that alternative D was preferred because it provided the same persistence probability and only slightly lower population viability than alternative E for approximately 60% of the cost of the next best alternative (O'Donnell et al., 2017). Despite reaching this decision, a replicate captive breeding facility for *A. cingulatum* has not yet been established (the only existing captive population for this species is being maintained by Mark Mandica with The Amphibian Foundation in Atlanta, GA), although USFWS Fish Hatcheries and many zoological facilities are willing to serve as facilities when successful captive rearing protocols are available. Moreover, no progress has been made yet in getting frosted flatwoods salamanders to breed in captivity; thus, to date no animals have been produced that could be used for reintroduction purposes.

In February 2015, a group of key partners met for a second SDM workshop to address how to restore wetland and upland habitat to minimize the extinction risk of *A. cingulatum* at SMNWR, one of the two remaining strongholds for this species (O'Donnell et al 2019.), To address this need, the group decided to launch a head-start effort of larval *A. cingulatum* at SMNWR, using cattle-watering tanks as aquatic mesocosms (Semlitsch and Boone, 2009). In 2016, researchers with the Florida Fish and Wildlife Conservation Commission (FWC), in partnership with the Apalachicola National Forest, started a similar program as well. This *in situ* approach has successfully been employed for the conservation and recovery of at least one other federally endangered amphibian, *Rana sevosa* (USFWS, 2014). The objectives of this ongoing effort are to rear larvae through to metamorphosis, releasing most at their natal ponds while placing some in captive facilities for future captive breeding. By rearing amphibian larvae in outdoor mesocosms, the extremely low survival that has been observed in wild populations of other species of *Ambystoma* (e.g., from egg to metamorphosed juvenile, <1.0% survival for *A. annulatum* and *A. maculatum* [Semlitsch et al., 2014; Anderson et al., 2015] and as low as 1.0-3.3% for *A. maculatum* [Shoop, 1974]) can be increased to as high as 90% (in the absence of predators) in mesocosms (R. D. Semlitsch, pers. comm. February 2015; T. L. Anderson, pers. comm. 25 March 2015; Anderson and Semlitsch, 2014; Anderson and Whiteman, 2015). Moreover, under these circumstances, individuals can metamorphose at a larger body size, which has been demonstrated empirically to correlate to adult fitness in at least two other species of *Ambystoma* (*A. talpoideum* and *A. opacum*; Semlitsch et al., 1988; Scott, 1994). For both of these species, larger juveniles at metamorphosis were also larger adults at first reproduction which, in turn, produced larger clutches of eggs at a younger age (Semlitsch et al., 1988; Scott, 1994).

Thus far, this effort has resulted in the production of 2,012 metamorphosed frosted flatwoods salamanders (728 at SMNWR and 1,735 at ANF), representing larval survival from 50.4% to 97.8% (Table 3.2). Many (at ANF) or all (at SMNWR) of these individuals have been marked, and all have been released at their natal ponds. Marked individuals will be monitored in capture-mark-recapture studies to assess survival and other vital rates. By releasing metamorphosed salamanders at their site of origin, the near-term objective of this effort is to build up resiliency of natural populations and to eventually release head-started individuals at other sites to increase representation and redundancy throughout their historic range. It is not known, however, how many metamorphs are needed for release – and over how many years – to see an increase in abundance and, thus, resilience at release sites.

Habitat restoration and/or creation is also necessary to stabilize populations because declines are primarily due to habitat loss, fragmentation and degradation, which are further exacerbated by prolonged winter droughts. Altered fire regimes and fire suppression, characterized by dormant-season (winter) burns and longer fire return intervals, are the leading contributors to habitat degradation. A number of approaches have been used to restore wetland and upland habitats for various pond-breeding amphibians (e.g. Litt et al., 2001; Gorman et al., 2013). Disagreement and uncertainty exists, however, because of critical gaps in understanding the relative effectiveness and cost of specific interventions.

Because extinction risk is higher when population dynamics are synchronous within or across metapopulations or RMUs, creating or restoring habitat conditions that would facilitate differences in survival or reproduction should reduce risks of extinction. Current restoration efforts aim to ensure there are habitats that can facilitate some survival and reproduction even in years with extreme flooding or extreme drought. For example, restoring suitable larval habitat to more central portions of large wetland basins may facilitate growth and survival of larvae even in drought years. Restoring some small basins that are unlikely to be connected to permanent water bodies even when flooding (so would avoid colonization by large fish predators) could facilitate growth and survival of larvae in wet years.

Table 3.4 *Preliminary results of head-start and salvage efforts for larval frosted flatwoods salamanders at St. Marks National Wildlife Refuge and the Apalachicola National Forest (FWC, USFS, USGS, USFWS unpubl. data). Mass (g), total length (mm; TL) and snout-vent length (mm; SVL) were measured at metamorphosis and are reported as means (1 standard deviation). *Researchers searched 34 ponds; 404.25 person-hours; 21 different searchers; 3 ponds had dead eggs only.*

Location	Season	Tanks	Ponds	Larvae / eggs	Metamorphs	Mass	TL	SVL	% Survival
SMNWR	2015-16	20	10	93	91	2.09 (0.73)	77.2 (10.07)	39.2 (4.32)	97.8
	2016-17	40	20	474	400	1.21 (0.26)	58.73 (4.97)	33.05 (3.02)	84.4
	2017-18	52	4	470	237	…	…	…	50.4
ANF	2016-17	48	8*	486	404	1.17 (0.27)	63.30 (5.53)	35.20 (2.75)	83.6
	2017-18	75	13	1,000	880	1.11 (0.23)	60.95 (4.52)	33.93 (2.09)	88.0
	2018-19	46	8	466	453	1.76(0.39)	71.70 (6.20)	38.2 (2.8)	97.2

3.7 Summary of Overall Current Condition: Population Resilience, Species Representation and Redundancy

Comparison of historical locations with records since 2000 demonstrates that the distributions of both species of flatwoods salamanders have been significantly reduced (Semlitsch et al., 2017). This decline is occurring at multiple spatial scales; (i.e., there has been a reduction in the number of populations along with a loss of individual breeding ponds within populations), which has diminished the probability of long-term persistence of this species.

Like many amphibians that breed in ephemeral wetlands, flatwoods salamanders exhibit dramatic fluctuations in abundance across years. Specific environmental conditions are required for successful recruitment; drought years result in catastrophic reproductive failure. To discern long-term trends from natural fluctuations, a stochastic Integral Projection Model (IPM) was constructed from 10 years of drift fence data obtained at two breeding wetlands on Eglin AFB. A population viability analysis (PVA) was conducted, whereby simulated populations were projected into the future and extinction risks under various scenarios were calculated (George Brooks, Virginia Tech, 2019, unpublished data). Owing to the stochastic nature of recruitment, extinction risk was high for single populations.

Population resiliency of the frosted flatwoods salamander can be summarized as low to moderate. However, the two representation units differ greatly in both resiliency and redundancy. Remaining populations of the frosted flatwoods salamander are distributed between two representation units – the Eastern Panhandle Unit and the Atlantic Coastal Plain Unit. The Eastern Panhandle Unit encompasses two stronghold occupancy areas – the Apalachicola National Forest (ANF) and St. Marks National Wildlife Refuge (SMNWR). Both ANF and SMNWR have redundant populations – ANF has approximately 38 extant breeding sites and SMNWR (including Flint Rock properties) has 36 which, on average, are of moderate resiliency and these two areas contain the only populations with high resiliency. In contrast, the Atlantic Coastal Plain Unit contains only one known breeding site of moderate resiliency. Should this population become extirpated, and if no other natural populations are found in South Carolina or Georgia, then the loss of this population would also mean the loss of an entire representation unit, leaving this species with only one.

In terms of redundancy, flatwoods salamanders currently exist only as isolated metapopulations in a few locations within their historical range (74 FR 6700; Semlitsch et al., 2017). Designated critical habitat (see section 4.2) is fragmented by cultivated cropland, developed lands, managed plantations, harvested forests, and other types of land use that is consistent with anthropogenic disturbance, and adjacent breeding sites are generally outside the range of likely dispersal. Connectivity among adjacent neighboring aquatic breeding sites is insufficient to maintain metapopulation dynamics, yet it is essential to enable rescue, through dispersal, of others that are declining, buffer against stochastic local extinction, maintain adequate genetic diversity, and to sustain metapopulations afflicted by disease (Heard et al., 2015). Distances between neighboring wetlands, on average, exceed known dispersal distances for ambystomatid salamanders, which can directly affect the probability of migration (Gibbs, 1993; Semlitsch and Bodie, 1998; Semlitsch, 2002; Lay et al., 2015).

Thus, with two exceptions (e.g., Apalachicola National Forest and St. Marks NWR), *A. cingulatum* populations have become increasingly isolated and are currently so spatially separated that it is unlikely, if not impossible, for animals to share any genetic material. Because of this genetic distinctiveness, Pauly et al. (2012) advised against the use of eastern panhandle populations as a source for future reintroduction on the Atlantic Coastal Plain, assuming that source populations from within the Atlantic Coastal Plain were available. Moreover, the remaining populations in South Carolina and at Fort Stewart, Georgia are extremely important from a conservation perspective as they represent the only known extant populations of *A. cingulatum* in the entire Atlantic Coastal Plain (Pauly et al., 2012). Yet only 8 adults and approximately 12 larvae have been captured on the Francis Marion National Forest in South Carolina in the past 20 years (Harrison, 2004, Harrison 2005, Palis 2009, internal USFS records) and none since 2010. The lack of known populations of *A. cingulatum* in Georgia and South Carolina will require consideration of where recovery populations may come from, if the Service determines that those populations are not recoverable by natural means.

CHAPTER 4 – FACTORS INFLUENCING VIABILITY

The scientific community agrees that amphibians are being impacted by six primary threats: 1) habitat loss and alteration, 2) chemical contamination, 3) global climate change, 4) disease, 5) invasive species, and 6) commercial exploitation (Semlitsch, 2003; Collins and Crump, 2009). The primary threats currently affecting flatwoods salamanders are changes in habitat (loss, fragmentation, and degradation, invasive species, pesticide use, hydrologic changes) and climate (particularly drought and variation in the timing of rainfall) (74 FR 6700; Figure 4.1). Habitat continues to be lost, degraded or altered by conversion for agriculture, silviculture, or commercial/residential development; strip mining; drainage or enlargement (with subsequent introduction of predatory fishes) of breeding wetlands; and alteration of terrestrial and wetland habitat resulting from fire suppression or alteration of natural fire regimes (74 FR 6700). Another principle threat is recurring drought during the aquatic larval period (Means et al., 1996; Palis et al., 2006; 74 FR 6700; Westervelt et al., 2013).

For amphibians, synergisms among these six factors are now widely recognized to be the drivers of population declines (Sih et al., 2004; Hayes et al., 2010). For example, the presence of the herbicide atrazine can increase the susceptibility of other species of *Ambystoma* to infections from ranavirus (Forson and Storfer, 2006). For flatwoods salamanders, habitat degradation, in the form of fire suppression, allows woody vegetation (e.g., broad-leaved species of trees, along with shrubs such as saw palmetto [*Serenoa repens*] and gallberry [*Ilex glabra*]), to invade uplands and their embedded ephemeral flatwoods wetlands. Consequently, the herbaceous ecotone that is preferred oviposition habitat disappears, the canopy begins to close, and the broad-leaved vegetation increases evapotranspiration at leaf-out, thus shortening wetland hydroperiods and compromising the ability of larval salamanders to reach metamorphosis. These conditions may then be exacerbated by drought (resulting from increases in temperature and decreases in precipitation), which further shortens hydroperiods, impacting metamorphosis and, thus, reproductive success (Figure 4.1).

4.1. Habitat Loss, Fragmentation and Degradation

The main threat to the flatwoods salamander is loss of both its longleaf pine/slash pine flatwoods terrestrial habitat and its isolated, seasonally inundated breeding habitat. The combined pine flatwoods (longleaf pine-wiregrass flatwoods and slash pine flatwoods) historical acreage was approximately 32 million ac (12.8 million ha) (Wolfe et al., 1988; Outcalt, 1997). The combined flatwoods acreage has been reduced to 5.6 million ac (2.27 million ha) or approximately 18% of its original extent (Outcalt, 1997). These remaining pine flatwoods (non-plantation forests) areas are typically fragmented and degraded, with second-growth forests.

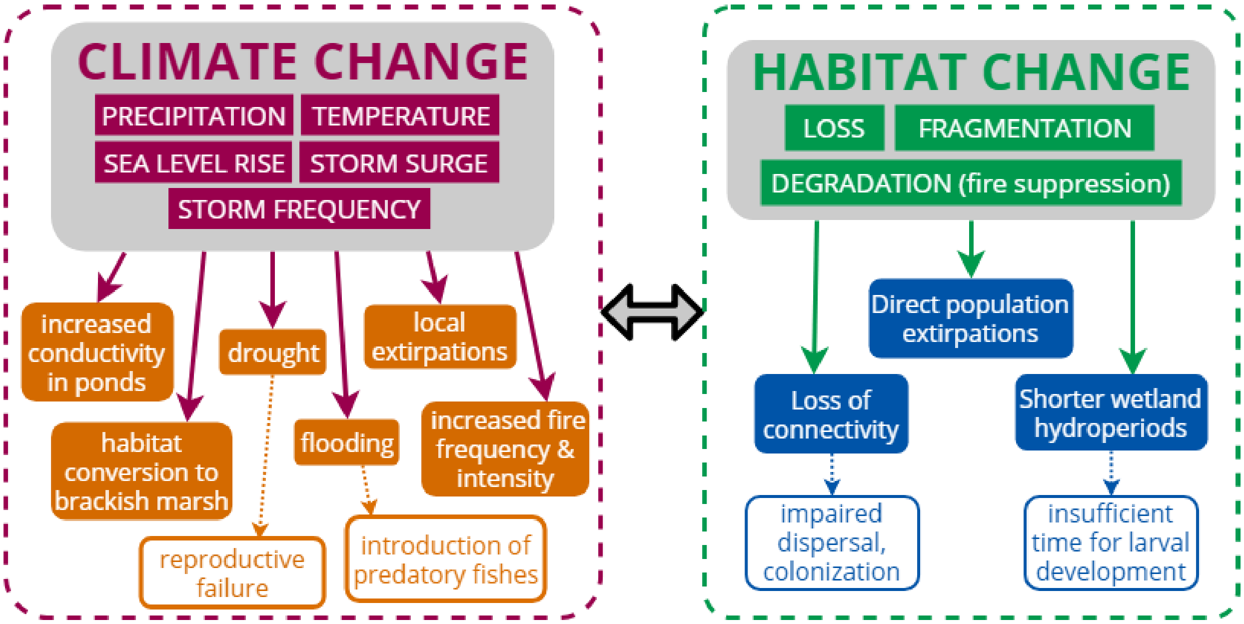

Figure 4.1. *Two of the principal stressors, changes in climate and habitat that impact populations of the frosted flatwoods salamander, along with their ecological and demographic consequences. Dashed lines represent examples of indirect effects from key consequences. The double arrow between the two sets of consequences indicates that changes in climate and habitat have consequences that interact synergistically to compound the negative effects of each stressor individually (credit: Katherine M. O'Donnell).*

Many ecologists consider altered or disrupted fire patterns (e.g. fire suppression) to be the primary reason for the degradation of remaining longleaf pine forests. Before human intervention, mesic flatwoods habitats would burn as often as every 1-3 years. Natural wildfires, typically ignited by lightning, occurred primarily in the early summer when conditions were predictably hot and dry (Noss, 2018). Over the last century, natural wildfires are regularly suppressed. On pinelands where fires still regularly occur, they are largely human-ignited prescribed fires. Prescribed fires are overwhelmingly implemented during the cooler wetter conditions of winter and early spring when they are easier to control (Bishop and Haas, 2005; Noss, 2018). The disruption of the natural fire cycle in pine forests has resulted in an increase in hardwood midstory and understory and a decrease in herbaceous ground cover (Wolfe et al., 1988; Gorman et al., 2013). The consequences to deprivation of fire from seasonal wetlands are more severe -- lack or may have a greater negative impact. Exclusion of fire within ponds during periods of dry-down allows wetlands to experience successional changes, rapidly becoming invaded by woody shrubs and deciduous hardwood trees. The resulting increase in shade and litter input produces a broad range of chemical and physical (vegetative) changes that ultimately render them unsuitable for the salamander (Bishop and Haas, 2005; Gorman et al., 2013). For example, fire releases alkaline cations from burned vegetation into water, which increases pH (Noss, 2018), possibly buffering the acidifying effect of accumulated tannins.

Broad application of prescribed fire in the dormant season (late fall through early spring)), can have negative effects on both salamanders and their habitat (Bishop and Haas, 2005): dormant season fire can remove cover used by salamanders during ingress and egress from breeding ponds, remove vegetative cover in egg deposition sites, destroy developing eggs (Powell et al., 2013), and may even cause direct or indirect mortality when it coincides with salamander movements. However, these burns are important for achieving other management objectives, such as reducing woody fuels and decreasing wildfire danger, but targeted efforts should be placed on burning the sites when breeding wetlands are dry, avoiding burning when salamanders may be migrating to and from the pond, and follow-up burns should be used to ensure wetlands benefit from fire.

To achieve the broadest range of ecological benefits, prescribed fire should occur during the lightning season (May-Aug) under conditions that will allow fires to burn into breeding wetlands. If winter drought conditions preclude successful salamander reproduction, land managers may burn during the dormant season when wetlands are dry and will carry fire. Further, to increase herbaceous vegetation and open the canopy, it may be necessary to burn the some upland sites during the dormant season to create a ‘‘safety’’ buffer to prepare for a growing season fire that targets the basin of the wetland (Gorman et al., 2009). Mechanical treatments with handheld equipment (e.g., brush saws and chainsaws) may also be used to successfully reduce canopy cover and facilitate herbaceous vegetation growth (Gorman et al., 2013).

Fragmentation of the longleaf pine ecosystem, resulting from habitat conversion, threatens the survival of the remaining flatwoods salamander populations. Large tracts of intact longleaf pine flatwoods habitat are fragmented by roads and commercial pine plantations. Most flatwoods salamander populations are widely separated from each other by unsuitable habitat. Amphibian populations may be unable to recolonize areas after local extinctions due to their physiological constraints, relatively low mobility, and site fidelity (Blaustein et al., 1994).

Road construction in the last two decades destroyed a historic breeding pond (of *A. bishopi*) in Escambia County, Florida. Roads also contribute to habitat fragmentation by isolating blocks of remaining contiguous habitat. They may disrupt migration routes and dispersal of individuals to and from breeding sites. In addition, vehicles may also cause the death of flatwoods salamanders during migrations across roads (Means, 1996, Pierson Hill pers. obs. 2019). Road construction is also a recurring threat to the remaining flatwoods salamander habitats. Roads generally can cause disruptions to groundwater and sheetflow, and have serious direct and indirect impacts on breeding ponds.

Conversion of natural pine flatwoods to intensively managed (i.e., impacted by heavy mechanical site preparation, high stocking rates, and low fire frequencies) slash or loblolly pine plantations often degrades flatwoods salamander habitat by creating well-shaded, closed-canopied forests with an understory dominated by shrubs or pine needles (Means et al., 1996). Ponds surrounded by pine plantations and deprived of fire may become unsuitable breeding sites due to canopy closure and the resultant reproduction in herbaceous vegetation, which is needed for egg deposition and larval development sites (Palis, 1993; Gorman, et.al., 2014) According to Enge et al. (2014), commercial forestry using silvicultural Best Management Practices (Florida Forest Service, 2008) will likely extirpate flatwoods salamander populations over time. More favorable practices for ephemeral pond-breeding amphibians are provided by Calhoun and deMaynadier (2004) and Bailey et al. (2006). Disturbance-sensitive groundcover species, such as wiregrass, dropseed, and perennial forbs are either greatly reduced in extent or are replaced by weedy pioneering species (Schultz and White, 1974; Moore et al., 1982; Outcalt and Lewis, 1988; Hardin and White, 1989). Wiregrass is an herbaceous species often lost in habitat conversion and considered an indicator of site degradation from fire suppression and/or soil disturbance (Clewell, 1989). It also appears to be absent from areas where flatwoods salamanders no longer occur (Palis, 1997). Past pine plantations were created on natural pine sites, whereas future pine plantations will increasingly be created on former agricultural land (Wear and Greis, 2002); thus, this type of habitat conversion is not considered an on-going threat to the flatwoods salamander. However, this could limit recovery potential from changes to the upland habitat.

Land use conversions to urban development and agriculture eliminated large acreages of pine flatwoods in the past (Schultz, 1983; Stout and Marion, 1993; Outcalt and Sheffield, 1996; Outcalt, 1997). State forest inventories completed between 1989 and 1995 indicated that flatwoods losses through land use conversion were still occurring (Outcalt, 1997). Urbanization, especially in the panhandle of Florida and around major cities, is reducing the available pine forest habitat. Wear and Greis (2002) identified conversion of forests to urban land uses as the most significant threat to southern forests. These authors predicted that the South could lose about 12 million forest acres (about 8% of its current forest land) to urbanization between 1992 and 2020.

Forestry management which includes intensive site preparation may adversely affect flatwoods salamanders both directly and indirectly (Means et al., 1996). Bedding (a technique in which a small ridge of surface soil is elevated as a planting bed) alters the surface soil layers, disrupts the site hydrology and often eliminates the native herbaceous groundcover. This can have a cascading effect of reducing the invertebrate community that serves as a food source for flatwoods salamander juveniles and adults. Intensive site preparation also negatively impacts subterranean voids such as crayfish burrows and root channels that are the probable fossorial habitats of terrestrial salamanders and may result in entombing, injuring, or crushing individuals.

Flatwoods salamander breeding sites have also been degraded or altered. The number and diversity of these small wetlands have been reduced by alterations in hydrology, agricultural and urban development, incompatible silvicultural practices, shrub encroachment, dumping in or filling of ponds, conversion of wetlands to fish ponds, domestic animal grazing, and soil disturbance (Vickers et al., 1985; Ashton, 1992). Hydrological alterations, such as those resulting from ditches created to drain flatwoods sites or fire breaks and plow lines, for example, represent one of the most serious threats to flatwoods salamander breeding sites. Lowered water levels and shortened hydroperiods at these sites may prevent successful flatwoods salamander recruitment.

Off-road vehicle (ORV) use within flatwoods salamander breeding ponds and their margins severely degrades wetland habitat. Continued use of sites by ORVs can completely degrade the integrity of breeding sites by killing herbaceous vegetation and rutting the substrate, which can alter hydrology. Mechanical disturbance of the soil promotes red imported fire ants, a known predator of small amphibians. There is also the potential for direct injury and/or mortality of flatwoods salamanders by ORVs at breeding sites. Habitat loss from agricultural conversion or commercial development, pond alteration and additional introduction of predatory fish, fire suppression leading to altered forest habitat and crayfish harvesting comprise the most serious threats to *A. cingulatum* populations (Palis and Hammerson, 2008).

4.2 Changes in Land Use in Designated Critical Habitat Units

In 2009, the Service designated 19 critical habitat units (CHUs) for the frosted flatwoods salamander (74 FR 6700), omitting military lands: Department of Defense installations have an operational integrated natural resources management plan (INRMP) prepared under section 101 of the Sikes Act (16 U.S.C. 670a) that provides acceptable conservation benefits (74 FR 6700). All CHUs (across both species) were known to be occupied at the time when critical habitat was designated in 2009.

To quantify the current suitability of CHUs for supporting populations of the frosted flatwoods salamanders, the U. S. Geological Survey's (USGS) national GAP project, derived from the classification of Landsat TM satellite imagery, was used to evaluate the quantity and quality of designated flatwoods salamander critical habitat. ArcGIS (v. 10.2) was then used to clip GAP raster data to each CHU boundary using each CHU polygon vector. The amount (area, in ha) of each of seven to 27 habitat types within each of three different land use categories was then quantified and grouped as Agriculture/Disturbed (7 habitats), Plantation (7 habitats), and Natural (27 habitats). The amount of wetland habitat (< 4.0 ha in size) present in each CHU was also quantified using two different datasets that vary in the features they target: the National Wetlands Inventory (NWI) and the USGS National Hydrography Dataset (NHD: http://nhd.usgs.gov/). The value of 4 ha was used in this assessment because breeding habitat for flatwoods salamanders has been defined as being this size or smaller (74 FR 6700). The potential for CHUs to support metapopulation dynamics was also determined by calculating the minimum distance (m) between known occupied wetlands and adjacent wetlands within each CHU.

The critical habitat units for *A. cingulatum* range in size from 62.5 ha (FFS-5A) to 2,174.3 ha (FFS-1G), with a median value of nearly 240 ha. On average, 39% of the designated critical habitats are currently comprised of vegetation types (agriculture/disturbed and plantation habitat) that are not suitable for *A. cingulatum* (Table 4.1; William J. Barichivich, USGS, pers. comm. 2019). These habitat categories include cultivated cropland, developed lands, managed plantations, harvested forests, and other types of land use that is consistent with anthropogenic disturbance. Each of two CHUs (FFS-1I and FFS-3C) have only one wetland of the size (< 4 ha) used for breeding by this species. On average, wetlands are 457.5 m (using NWI) to 808.7 m (using NHD) apart, which exceeds the maximum dispersal distance (380 m; Scott et al., 2013) that has been reported for eight other species of *Ambystoma* (Table 4.1). The most recent year occupancy was confirmed among 13 CHUs was 2011 but, for five of these, the year of last observation was in the 1980s and 1990s (Table 4.1).

For species that use separate juvenile and adult habitats (such as flatwoods salamanders), terrestrial adult and juvenile population sizes can be limited by the size of their habitat, especially during pulse episodes of juvenile migration (e.g., emergence of metamorphs) into adult populations (Halpern et al., 2005). In addition, dispersal, survival and genetic variation may be influenced by habitat quality (e.g., Rothermel, 2004; Rothermel and Semlitsch, 2006; Richter et al., 2013). A time lag often exists between when habitat alteration occurs and when the effects of that modification on populations become apparent (Richter et al., 2013; Semlitsch et al., 2017). Nevertheless, anthropogenic habitat disturbance in each unit, the distance between neighboring ponds and, possibly, the size of CHUs may affect abundance, dispersal, survival and genetic variation in this species, all of which are measures of population resilience.

Table 4.1. *Summary of key features of designated critical habitat units (CHUs) for the frosted flatwoods salamander. NHD=National Hydrology Dataset; NWI=National Wetlands Inventory. Susan Walls et al., unpubl. data. 2019*

CHU	CHU area (ha)	% CHU comprised of agriculture/ disturbed and plantation habitat types	Mean distance to nearest neighbor pond (NHD)	Mean distance (m) to nearest neighbor pond (NWI)	Last year confirmed occupancy	Year last surveyed
FFS-1A	924.4	71.0	672.3	530.8	2011	2016
FFS-1B	296.5	59.9	617.5	398.1	2011	2016
FFS-1C	393.2	39.5	694.4	557.7	<2002	2016
FFS-1D	230.0	23.1	870.3	511.0	<2002	2016
FFS-1E	1489.0	47.0	587.2	444.6	2016	2016
FFS-1F	65.5	31.9	345.8	329.2	2007	2016
FFS-1G	2174.3	39.2	569.2	462.9	2016	2016
FFS-1H	359.0	19.3	471.85	456.0	2016	2016
FFS-1I	65.5	18.0	1807.1	887.3	2002	2016
FFS-1J	239.9	27.8	852.7	1009.1	2007	2016
FFS-3A	1245.5	55.8	500.7	295.8	2016	2016
FFS-3B	730.0	48.7	450.2	321.7	2016	2016

FFS-3C	65.7	72.8	704.3	727.6	1997	…
FFS-4A	222.5	59.9	415.0	458.1	1996	2016
FFS-4B	65.6	45.9	…	287.4	…	…
FFS-5A	62.5	25.9	3255.5	225.3	1992	…
FFS-5B	74.0	38.2	366.2	228.1	1998	…
FFS-6	526.40	13.1	932.5	291.6	2010	2016
FFS-7	65.6	7.5	443.3	269.7	1987	2016[1]
Median (25^{th}, 75^{th} percentiles)	239.67 (65.7, 728.5)	–	–	–	–	–
Mean $\pm$ 1 se		39.18 $\pm$ 4.40	808.7 $\pm$163.96	457.5 $\pm$ 50.99	–	–

[1]Surveyed annually by SCDNR (Wade Kalinowsky)

4.3 Climate Change and Associated Factors

In 2009, the Service acknowledged the negative effects of drought on flatwoods salamanders, but had no data supporting global climate change as a specific threat (74 FR 6700). Climate change, especially in combination with other stressors, is a daunting challenge for the persistence of amphibians and drought is not the only climate-related threat to pond-breeding amphibians (Walls et al., 2013). Flooding, such as that which occurs during extreme precipitation events, along with storm surge and its associated salt water intrusion during hurricanes and other tropical cyclones (Lin et al., 2014), can potentially impact amphibians (like flatwoods salamanders) that use freshwater coastal wetlands (Walls et al., 2013). Moreover, phenological shifts in the timing of key climatic events (e.g., pond-filling and drying) can have significant consequences to individual survival and species persistence (Walls et al., 2013, and references therein). Last, sea level rise threatens the loss of coastal freshwater wetlands, their surrounding upland habitats, and landscape connectivity (Tebaldi et al., 2012; Benscoter et al., 2013; Woodruff et al., 2013; Wahl et al., 2014; Leonard et al., 2017).

The most recent Assessment Report of the Intergovernmental Panel on Climate Change (IPCC) reinforces earlier conclusions that climate change is projected to alter the frequency and magnitude of flood and drought events in a warmer climate (Jiménez Cisneros et al., 2014). In addition to increased temperatures, more variable patterns of precipitation are predicted to occur in the future, with longer droughts and larger (but fewer) rainfall events (Heisler-White et al., 2008; Lucas et al., 2008). Model projections for the 2090s indicate that the proportion of the global land surface in extreme drought is predicted to increase by a factor of 10 to 30 (Burke et al., 2006; Kundzewicz et al., 2007). The number of extreme drought events per 100 years and mean drought duration are anticipated to increase by factors of two and six, respectively, by the 2090s (Burke et al., 2006; Kundzewicz et al., 2007). Simultaneously, the frequency of heavy rainfall or the proportion of total precipitation from heavy rainfall events will likely increase over many areas of the world in the 21st century (Seneviratne et al., 2012). Increases in the occurrence of drought and heavy precipitation events are known to be impacting a variety of amphibians, including those that breed in ephemeral wetlands (Walls et al., 2013). In addition to rainfall amounts, the timing of precipitation events is an important stimulus for reproduction in many pond-breeding amphibians (Walls et al., 2013). Thus, climate change may have an impact on frosted flatwoods salamanders by altering the timing of fall and winter rains, as well as creating drier winters than historically would have occurred (Chandler, 2015).

4.3.1 Changes in Temperature and Precipitation

Long-term variation in temperature and precipitation will likely affect flatwoods salamanders through a variety of direct and indirect pathways (Figure 4.2; Blaustein et al., 2010). Changes in precipitation affect wetland inundation directly as well as indirectly by affecting evapotranspiration and groundwater levels which, in turn, impact wetland hydrology (Figure 4.2). Inadequate rainfall and extreme drought can shorten pond hydroperiods, leading to reproductive failure or the elimination of reproduction altogether: in the mole salamander (*Ambystoma talpoideum*), as much as 90% of a population may skip breeding in a drought year (Kinkead and Otis, 2007). Flooding of breeding sites from late-season hurricanes may also impact reproductive success (Walls et al., 2013). Such heavy rainfall can prematurely fill the basins of ephemeral ponds, forcing females to oviposit along the outer margins of the pond basin, which may not be inundated later in the season (Walls et al., 2013). Indeed, flatwoods salamander reproduction in the ANF was heavily impacted by drought in 2017 and 2018, and from heavy rainfall in 2019 (P. Hill, pers.comm). Thus, factors that influence the timing and amount of precipitation during rainfall events, along with persistence and duration of wetland hydroperiods, are likely <u>*the most important*</u> constraints on the reproductive success and persistence of flatwoods salamanders (Figure 4.2; Blaustein et al., 2010). Variation in precipitation and temperature directly impact other components of the biological community, such as the availability of prey and the presence of predators (Figure 4.2).

Temperature changes can exacerbate the negative effects of other factors such as disease agents and contaminants (Raffel et al., 2006) and directly influence fire intensity, evapotranspiration, and soil moisture which, in turn, can impact adult and metamorphosed juvenile flatwoods salamanders in the terrestrial environment (Figure 4.2). Increased drought and temperatures could also make prescribed fire more difficult to employ. In an important study with metamorphosed streamside salamanders (*Ambystoma barbouri*), Rohr and Palmer (2013) experimentally manipulated temperature (within the critical thermal limits for this species), moisture (wet or dry conditions) and chronic exposure (as embryos and larvae) to varying, sublethal concentrations of the herbicide atrazine. These authors found that even moderate correlates of climate change (i.e., temperature variation within nonlethal limits) had significant negative effects on survival, growth, behavior, and foraging, especially when acting in the presence of other stressors (Rohr and Palmer, 2013). These results can help make predictions about possible responses of the frosted flatwoods salamander to similar conditions, even though this experiment was not conducted with this species.

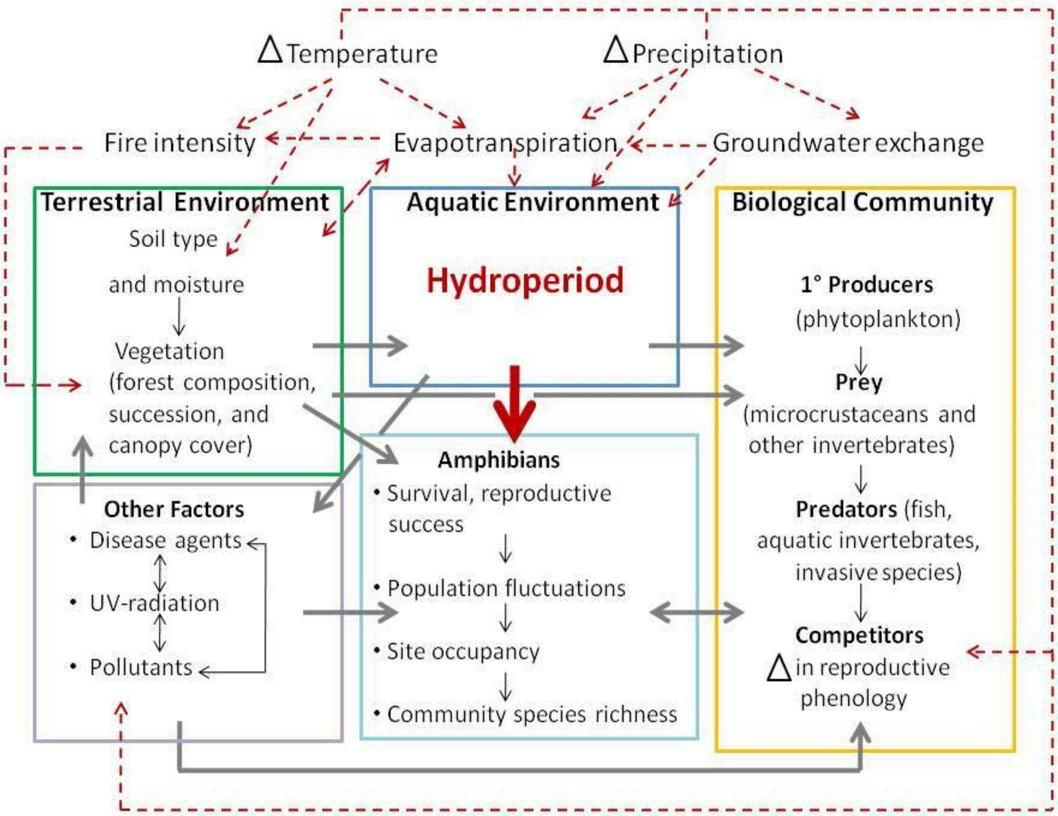

Figure 4.2. *Conceptual model of pathways through which changes in temperature and precipitation may impact the resiliency of flatwoods salamander populations. Changes in temperature and precipitation directly affect terrestrial and aquatic habitats, the biological community (of which flatwoods salamanders are a component) and other factors such as disease agents, UV-B radiation and pollution. Factors that influence the availability of water, such as the hydroperiod of aquatic habitats, are likely the most important constraints on the reproductive success and persistence of flatwoods salamanders (indicated by heavier red arrow). Dashed red arrows indicate interactions among meteorological variables, and their effects on biological communities and the environments in which they occur. Heavy solid gray arrows indicate relationships among compartments; lighter solid arrows indicate relationships within compartments. Modified from Blaustein et al., 2010.*

4.3.2 Hurricane-related Storm Surge and Sea Level Rise

The frequency of tropical storms and major hurricanes in the North Atlantic has increased over the past 100 years (Seneviratne et al., 2012). Current climate models project a 28% reduction in the overall frequency of Atlantic storms, yet an 80% increase in the frequency of more intense (Saffir-Simpson Category 4 and 5) Atlantic hurricanes over the next 80 years under the A1B emissions scenario (Seneviratne et al., 2012). These models also predict increases in tropical cyclone-related rates of rainfall (Seneviratne et al., 2012).

To examine the potential impact of hurricane-related storm surge and sea level rise on coastal *A. cingulatum* breeding sites, Walls et al. (2019) focused on the most vulnerable designated critical habitat units – FFS-3A, 3B, and 3C at St. Marks National Wildlife Refuge (SMNWR) This federal property includes approximately 43 miles (69.2 km) along the Gulf Coast of northwest Florida, and many flatwoods salamander breeding sites are in close proximity to the coast line. We used sea level rise and marsh mitigation data from NOAA https://coast.noaa.gov/digitalcoast/data/ in a SLOSH (Sea, Lake, and Overland Surges from Hurricanes) model https://www.nhc.noaa.gov/surge/slosh.php to assess whether these critical habitat units are vulnerable to storm surge, sea level rise and encroaching marsh.

As Figure 4.3 shows, in a Category 1 storm, the SLOSH model predicts that the maximum storm surge would inundate all but a couple of breeding sites in FFS-3A and FFS-3B (Figure 4.3A). In a Category 3 storm, the maximum storm surge would completely inundate all breeding sites in FFS-3A and 3B and would approach the southern boundary of FFS-3C (Figure 4.3B). In more intense storms (Categories 4 and 5), the maximum surge level would completely inundate FFS-3C as well (not shown).

Figure 4.3C indicates the predicted outcome of 3 feet (1 m) of future sea level rise. Sea levels would closely approach critical habitat units FFS-3A and 3B and would inundate a few breeding sites under this scenario. However, perhaps the most dramatic occurrence would be the advancement of marsh habitat in front of encroaching sea levels: the habitat that encompasses most of the current freshwater flatwoods wetlands in these units would change to brackish marsh, conditions that would be incompatible with persistence of flatwoods salamanders in this region.

Hurricane Michael hit the Gulf Coast on October 10th, 2018 as a Category 5 storm. Much of the known breeding and upland habitat at St. Marks National Wildlife Refuge was inundated with salt water from the storm surge for weeks after the storm including 17 known breeding wetlands (Walls et al., 2019). In addition, they found that post-hurricane conductance observations at overwashed wetlands were, on average, more than 90 times higher than in the previous spring prior to the hurricane. Surveys are currently underway to accurately gauge the extent of the storm's effects on the populations and habitat.

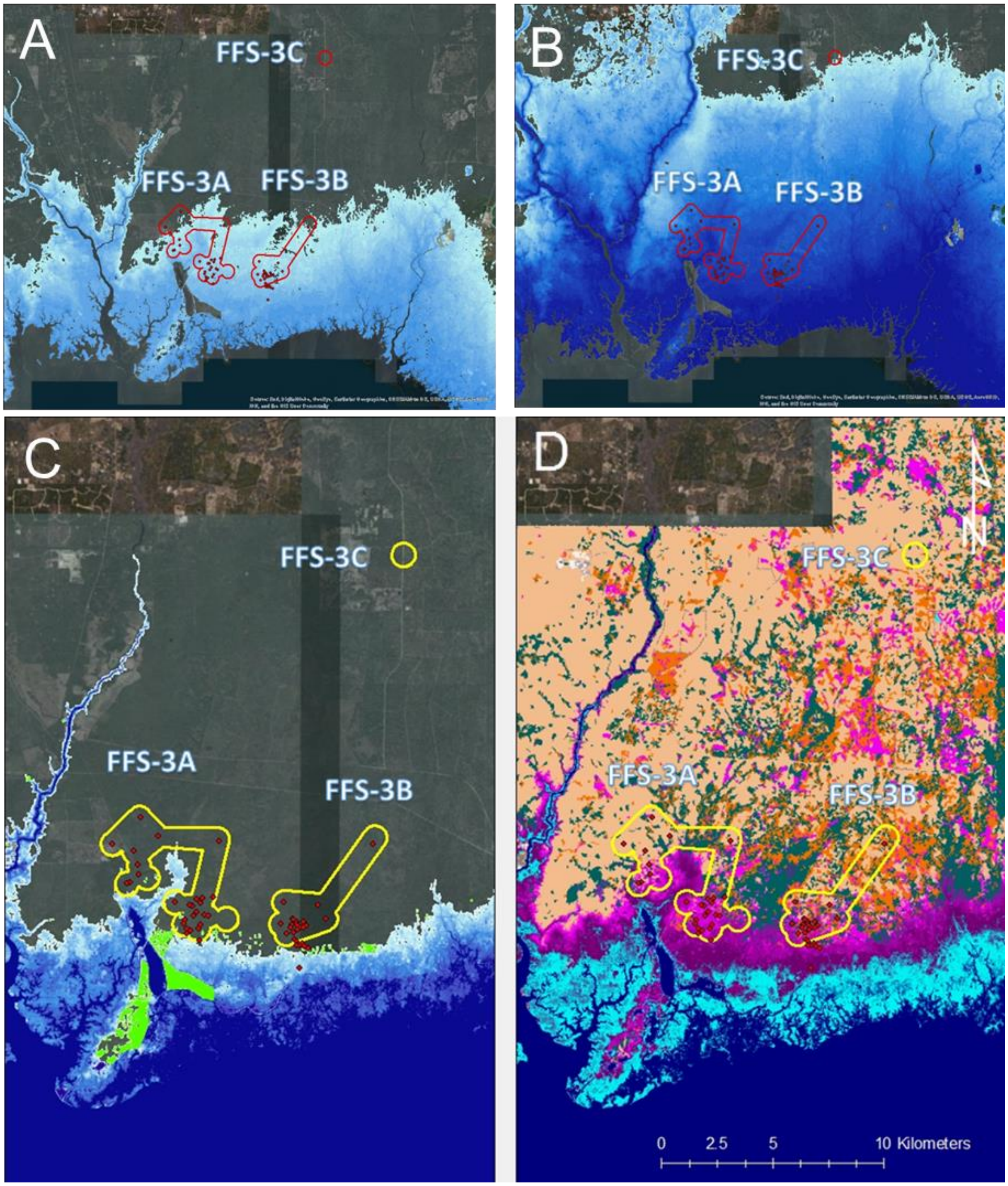

Figure 4.3. *SLOSH models of maximum surge level during a (A) Category 1 and (B) Category 3 storm. (C) Areas of inundation (in light blue) under a scenario of 3 feet of sea level rise for Critical Habitat Units FFS-3A and 3B at SMNWR. (D) Areas converted to marsh habitat (in pink) as a consequence of 3 feet of sea level rise. Inset: location of SMNWR along the Gulf Coast of the U.S. Modified from Walls et al. (2019)*

4.4 Other Stressors

4.4.1 Disease

Ranaviruses in the family *Iridoviridae* and chytrid fungus may pose potential threats, although the susceptibility of the frosted flatwoods salamander to these diseases is unknown. Ranaviruses have been responsible for die-offs of tiger salamanders throughout western North America and spotted salamanders (*A. maculatum*) in Maine (Daszak, et al., 1999). The chytrid fungus (*Batrachochytrium dendrobatidis,* or Bd), which causes chytridiomycosis in many amphibians, has been discovered and associated with mass mortality in tiger salamanders in southern Arizona and California, and the Santa Cruz long-toed salamander (*A. macrodactylum croceum*) (Vredenburg and Summers, 2001; Davidson, et al., 2003; Padgett-Flohr and Longcore, 2005). Recently, a newly discovered species of chytrid fungus, *Batrachochytrium salamandrivorans* (Bsal), was isolated from a mortality event that caused the near extinction of a population of fire salamanders (*Salamandra salamandra*) in Europe (Martel et al., 2013; Spitzen-van der Sluijs et al., 2013). Currently, it is not known whether any amphibian mortality events in the U.S. are attributable to this pathogen, or whether this new species even occurs in this country. Efforts to begin sampling for Bsal in the U.S. are currently underway. This discussion of disease in other species of closely related salamanders indicates the potential existence of similar threats to frosted flatwoods salamander populations for which we will monitor.

4.4.2 Predation

Exposure to increased predation by fishes is a potential threat to the frosted flatwoods salamanders when isolated, seasonally ponded wetland breeding sites are changed to, or connected to, more permanent wetlands inhabited by fishes that are not typically found in temporary wetlands. Wetlands/ponds may be modified specifically to serve as fish ponds or sites may be altered because of drainage ditches, firebreaks, or vehicle tracks which can all provide avenues for fish to enter the wetlands from other water bodies. Studies of other ambystomatid species have demonstrated a decline in larval survival in the presence of predatory fish (Semlitsch, 1987; 1988).

4.4.3 Contaminants and Natural Stressors

Even at nonlethal levels, natural and anthropogenic stressors that are commonly found in many aquatic systems (e.g., herbicides, variation in pH, salinity and temperature, and presence of both native and nonindigenous predators) can pose significant threats to larval amphibians (Burracoand Gomez-Mestre, 2016). In other species of *Ambystoma*, herbicides, such as atrazine, can increase susceptibility to ranavirus infections (Forson and Storfer, 2006). Exposure of embryos and larvae of another *Ambystoma* to nonlethal concentrations of this contaminant decreased water-conservation behaviors, foraging efficiency, mass, and time until death in individuals after they had metamorphosed (Rohr and Palmer, 2013).

4.4.4 Invasive Species

Nonindigenous feral swine can significantly impact frosted flatwoods salamander breeding sites through rooting; intensive approaches (e.g., control measures and fencing) may be needed to avoid degradation to occupied sites and sites going through restoration.

Red imported fire ants (*Solenopsis invicta*) are potential predators of frosted flatwoods salamanders. Fire ants are well-known predators of fauna ranging from small invertebrates (Porter and Savignano 1990) to reptiles, amphibians, and mammals (Allen et al., 2004). Experimental elimination of fire ants in pineland habitats was shown to benefit herpetofaunal abundance and species richness (Allen et al., 2017). Fire ants were the primary cause of mortality for juvenile marbled salamanders (*A. opacum*) and mole salamanders (*A. talpoideum*) in outdoor experimental enclosures (Todd et al., 2017). Fire ants have been observed around most breeding sites in the ANF (Pierson Hill, pers. comm. 2019). They have also been seen in areas disturbed by the installation of drift fences at known breeding sites (T. Gorman, pers. comm., 2015) and attacking salamanders in drift fence traps in the ANF (Pierson Hill, pers. comm., 2018). Controlling fire ants in areas with a high degree of disturbance can be accomplished by using hot water rather than pesticides (Tschinkel and King, 2007), so on a small scale fire ants can be controlled around breeding sites. Further study on the effects of fire ants on flatwoods salamanders is recommended because the severity and magnitude, as well as the long term effect of fire ants on frosted flatwoods salamander populations is currently unknown. We consider predation to be a threat to the frosted flatwoods salamander at this time.

Invasive plant species such as cogongrass (*Imperata cylindrica*) threaten to further degrade existing habitat. Cogongrass, a perennial grass native to Southeast Asia, is one of the leading threats to the ecological integrity of native herbaceous flora, including that in the longleaf pine ecosystem (Jose et al., 2002). Frosted flatwoods salamander habitat management plans will need to address threats posed by invasive plants and develop strategies to control them. It has been documented that cogongrass can displace most of the existing vegetation except large trees. Especially threatening to the frosted flatwoods salamander if the ability of cogongrass to outcompete wiregrass (*Aristida* sp.), a key vegetative component of reticulated flatwoods salamander habitat.

CHAPTER 5 – FUTURE CONDITIONS

In addition to the PVA results described earlier, we used expert elicitation and climate change predictions to assess the future condition for frosted flatwoods salamanders by modeling the number of active breeding ponds under different management and climate scenarios at multiple timescales (1, 10, 20, 30, and 80 years) in the future. Scenario time frames were selected based on the time frames of the management and climate predictions with the greatest relevance for the species in conjunction with consideration of the reliability of expert elicitations. As the demographic and genetic data for defining populations in this species is limited, we considered each individual breeding pond to be representative of a single population. This approach is supported by studies in closely-related *Ambystoma* species that suggest strong fidelity to natal ponds, relatively short dispersal distances and limited dispersal ability, and genetic differentiation between neighboring ponds (Gamble et al., 2007; Peterman et al., 2015; Scott et al., 2013; Wendt, 2017). The impact of management and climate scenarios on the numbers of individuals within populations was not considered due to the lack of population demographic data for this species.

5.1 Development of Management Scenarios

Three types of management scenarios were developed based on the current number of active breeding ponds observed during recent surveys (2014–2018) and breeding pond succession and restoration rates elicited from knowledgeable land managers and species experts. A wetland loss scenario estimated the loss of active breeding ponds over time due to a loss of nesting habitat from natural habitat succession in which wetland herbaceous vegetation is reduced due to shrub encroachment and organic matter accumulation over time. This scenario assumed that no species-specific management of breeding ponds would occur and no measurable or successful restoration of potentially suitable (but currently degraded) breeding ponds would offset the loss of currently active breeding ponds. This represents a worst-case pond management scenario and the current scenario on many properties within the range of this species that lack adequate species-specific management or wetland restoration programs. We also modeled a wetland maintenance scenario where currently active breeding ponds are maintained in suitable condition by species-specific wetland management activities, but without successful efforts to restore additional potential breeding ponds. This scenario would reflect a situation where all species-specific management is focused on currently active breeding ponds. Finally, we modeled a wetland restoration scenario in which no active breeding ponds are being lost and currently unsuitable breeding ponds are restored to increase the population size. This represents a best-case scenario in which species management is a high priority, where all active breeding ponds are maintained by appropriate species-specific management such that no succession and loss of active breeding ponds occur, and all restored breeding ponds are colonized by the species. This scenario is not currently achievable due to the species management challenges discussed in previous chapters. However, if current barriers to species management are resolved and species management is considered a top priority for land managers, this scenario might be possible. In reality, the management of breeding ponds on most currently occupied properties lies somewhere between the wetland loss and wetland maintenance scenarios where the loss of breeding ponds over time due to wetland succession is offset, at least to some degree, by the addition of new breeding ponds from active restoration programs. However, survey results show recent declines of active breeding ponds on all properties, suggesting that all occupied properties are losing active breeding wetlands over time. These declines reflect species population declines due to deficits in wetland habitat management and other factors.

To predict the number of active breeding ponds under each management scenario, we invited all land managers, species managers, and species experts involved with the management of the species or their habitat on occupied properties to attend an informational webinar and answer a survey on how each management scenario would affect the number of active breeding ponds over time based on a 4-step elicitation method (O'Hagan et al., 2006). We invited a total of 43 participants to two webinars held on May 29, 2018 and June 12, 2018 and received surveys from a total of 13 participants (30% response rate). Responding participants included regional amphibian experts, representatives from all currently occupied states and properties, as well as lead state and federal agencies responsible for species management on those properties (Appendix 1). Participants were asked to estimate the number of inactive breeding ponds that could be restored to suitable habitat conditions and colonized by the species for the wetland restoration scenario, as well as the number of active breeding ponds that would become unsuitable for successful reproduction due to natural habitat succession without species-specific management for the wetland loss scenario within 1, 10, 20, 30, and 80-year time frames. Participants were also asked to provide their level of confidence on a scale of 0-100%. Respondents could choose to respond for the entire range of the species or for a specific property. Each occupied property had 2-5 respondents that were familiar with the species and habitat on that property. For the sake of simplicity, it was assumed that restored ponds would be naturally colonized by the species, although this would be unlikely if restored ponds were greater than the presumed dispersal distance of 500-m (app. 1500 ft.) from occupied ponds. We also assumed that all restored ponds would be maintained over time, although we recognize that this does not always occur due to the challenges of managing wetlands for this species. Survey respondents were also asked to set an upper limit on the number of ponds that could be restored based on the number of suitable ponds on their property, or if responding for the species range-wide, an upper limit of restorable ponds on all the currently occupied properties. The wetland maintenance scenario assumed that the current number of active breeding ponds would remain constant over time and was not based on expert elicitation.

5.1.1 Wetland Succession Scenario

The mean number of active breeding ponds differed greatly between the management scenarios over time (Figure 5.1). However, there was greater variation in the participant responses for the wetland restoration scenario, indicating greater uncertainty among respondents for this scenario. Under the wetland succession scenario, the number of active breeding ponds decreased rapidly resulting in a mean of 1 (±1.8 SD) active breeding ponds after 80 years from participants who responded for the species range-wide and 1 (±1.2 SD) active ponds after 80 years when all estimates were added from participants who responded by property (Figure 5.1, Tables 5.1–5.2). Three of the 10 respondents thought that all breeding ponds in the species' range would be inactive after 20 years under the wetland succession management scenario (Table 5.1).

Under the wetland succession scenario, 1 breeding pond was assessed as highly resilient (either a 4 or 5) and 32 ponds were moderately resilient 20 years in the future. The 42 remaining ponds had low resiliency (either a 1 or 2) (Table 5.3). Based on comments provided by participants during the elicitation, population resiliency would also be expected to decrease under this scenario since it would lead to the degradation of breeding habitat within all ponds resulting in less nesting habitat and lower larval survival rates due to decreased larval cover from predators within breeding ponds.

Under this scenario, population redundancy would be expected to decrease as the number of active breeding ponds (each representing a population) decreased on each property. Additionally, the only remaining active breeding pond representing the eastern clade of this species became inactive after 10 years under this management scenario (Table 5.2). Thus, the eastern clade of the species would have no representation under this scenario as no active breeding ponds would remain after 10 years. These results indicate the species experts and land managers acknowledge that active management of breeding ponds is critical to maintaining breeding habitat for the frosted flatwoods salamander.

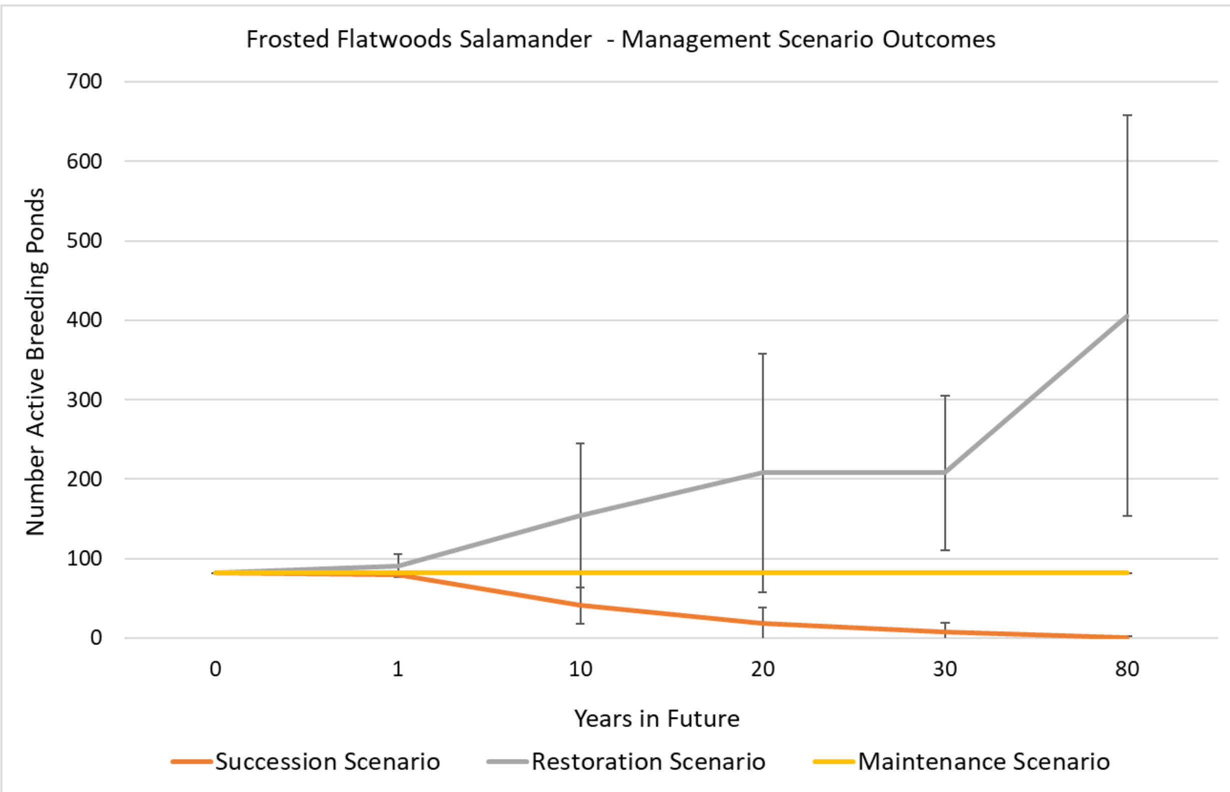

Figure 5.1. *Predicted change in mean ± SD active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds under three wetland management scenarios over time. Wetland succession and restoration scenarios based on expert elicitation. Wetland maintenance scenario assumes current number of active ponds are maintained over time.*

Table 5.1. *Predicted mean (± SD) total number of active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on currently occupied properties 1, 10, 20, 30, and 80 years in the future based on elicited estimates under the wetland succession scenario for*

respondents responding for the species range-wide. Not all respondents provided answers for later timescales. A lack of response is indicated by a dash.

	Number Active Breeding Ponds				
Property	**1 Year**	**10 Years**	**20 Years**	**30 Years (Year 2050)**	**80 Years (Year 2100)**
Respondent 1	80	43	13	2	0
Respondent 2	80	55	42	-	-
Respondent 3	78	40	7	0	0
Respondent 4	80	60	40	20	5
Respondent 5	82	40	0	0	0
Respondent 6	78	20	5	-	-
Respondent 7	77	12	0	0	0
Respondent 8	77	6	0	0	0
Respondent 9	82	75	55	30	0
Respondent 10	80	60	20	10	0
Mean (±SD)	79 (±1.8)	41 (±22.6)	18 (±20.3)	8 (±11.5)	1 (±1.8)

Table 5.2. *Predicted mean (± SD) numbers of active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on currently occupied properties 1, 10, 20, 30, and 80 years in the future based on elicited estimates under the wetland succession scenario for respondents responding by property. E= eastern clade W = western clade*

		Number Active Breeding Ponds				
Property	**Clade**	**1 Year**	**10 Years**	**20 Years**	**30 Years (Year 2050)**	**80 Years (Year 2100)**
Fort Stewart, GA	E	1 (±0.7)	0 (±0)	0 (±0)	0 (±0)	0 (±0)
Apalachicola National Forest, FL	W	38 (±1.9)	24 (±15.2)	12 (±9.3)	3 (±2.9)	1 (±1.2)
Flint Rock properties, FL	W	5 (±1.2)	1 (±1.5)	1 (±1.7)	0 (±0)	0 (±0)
St. Marks National Wildlife Refuge, FL	W	33 (±1.2)	12 (±12.5)	5 (±6.6)	0 (±0.5)	0 (±0)
Total		77 (±2.6)	37 (±19.7)	18(±11.5)	3 (±2.9)	1 (±1.2)

Table 5.3*. Land manager assessments of the* ***overall resiliency*** *of flatwoods salamander habitat on their property under the wetland succession scenario 20 years in the future. Responses are the percent of extant breeding ponds that fit into each point on the 5-point resiliency scale: (1) extremely low resiliency; (2) low resiliency; (3) moderate resiliency; (4) high resiliency; or (5) extremely high resiliency. ANF =Apalachicola National Forest; SMNWR = St. Marks National Wildlife Refuge; FR = Flint Rock properties; FS = Fort Stewart (Georgia).*

Property	**Extremely Low**		**Low**		**Moderate**		**High**		**Extremely High**		**Total Ponds**
	Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	
ANF	40	40.0	23	10.9	37	29.1	0	0.0	0	0.0	38
SMNWR	17	0.0	50	0.0	50	0.0	0	0.0	0	0.0	30
FR	0	0.0	0	0.0	67	0.0	0	0.0	0	0.0	6
FS	0	N/A	0	N/A	0	N/A	100	N/A	0	N/A	1
Total											***75***

5.1.2 Wetland Maintenance Scenario

Currently, there are 82 active breeding ponds for this species (Table 5.4). Under this scenario, the number of breeding ponds would stay the same on each property over time unless reduced or increased by something other than wetland management. This scenario represents a situation in which there is active and adequate species-specific management at currently active breeding ponds (i.e. resiliency is high [either a 4 or 5]) with no effective attempt to restore additional ponds for the species. Under this scenario, both the eastern and western clades are represented, but there is no redundancy of eastern clade populations because they are limited to the single, current active breeding pond on Fort Stewart.

Table 5.4. *Number of active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds with high resiliency on currently occupied properties used for wetland maintenance scenario. E= eastern clade W = western clade*

Property	**Clade**	**Active Breeding Ponds**
Fort Stewart, GA	E	1
Apalachicola National Forest, FL	W	39
Flint Rock properties, FL	W	6
St. Marks National Wildlife Refuge, FL	W	36
Total		82

5.1.3 Wetland Restoration Scenario

Under the wetland restoration scenario, both the mean number of active breeding ponds and the variation in participant responses increased. Under the wetland restoration scenario, the number of active breeding ponds increased rapidly resulting in a mean of 406 (±252.2 SD) active breeding ponds after 80 years for participants who responded for the species range-wide (Figure 5.1, Table 5.5) and 373 (±135.2 SD) active ponds after 80 years when all estimates were added for participants who responded by property (Table 5.6).

Under this scenario, species resiliency and population redundancy would be expected to increase as the habitat in each pond improved and the number of active breeding ponds (each representing an individual population) increased on each property. Under the wetland restoration scenario, 56 breeding ponds were assessed as highly resilient (either a 4 or 5) and 19 ponds were moderately resilient at year 20. No ponds had low resiliency (either a 1 or 2) (Table 5.7). Furthermore, the eastern clade of the species would be represented by 20 active breeding ponds on one property under this scenario providing an increased, but relatively small, degree of redundancy for these genetically distinct populations.

These results indicate that species experts and land managers believe breeding pond restoration and active wetland management are successful approaches to management of frosted flatwoods salamander habitat. As with the wetland loss scenario, these results emphasize the importance of ongoing management to the persistence of the species.

Table 5.5. *Predicted mean (± SD) total number of active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on currently occupied properties 1, 10, 20, 30, and 80 years in the future based on elicited estimates under the wetland restoration scenario for respondents responding for the species range-wide. Not all respondents provided answers for later timescales. A lack of response is indicated by a dash.*

	Number Active Breeding Ponds				
Property	**1 Year**	**10 Years**	**20 Years**	**30 Years (Year 2050)**	**80 Years (Year 2100)**
Respondent 1	92	122	182	-	-
Respondent 2	92	182	282	382	800
Respondent 3	127	382	582	-	-
Respondent 4	87	157	157	-	-
Respondent 5	90	150	175	200	300
Respondent 6	82	96	105	125	175
Respondent 7	82	112	160	250	600
Respondent 8	82	92	110	150	400
Respondent 9	85	100	120	140	160
Mean (±SD)	91 (±14.1)	155 (±90.6)	208 (±150)	208 (±96.9)	406 (±252.2)

Table 5.6. *Predicted mean (± SD) numbers of active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on currently occupied properties 1, 10, 20, 30, and 80 years in the future based on elicited estimates under the wetland restoration scenario for respondents responding by property. E= eastern clade W = western clade*

		Number Active Breeding Ponds				
Property	**Clade**	**1 Year**	**10 Years**	**20 Years**	**30 Years**	**80 Years**

					(Year 2050)	**(Year 2100)**
Fort Stewart, GA	E	2 (±1.4)	11 (±7.8)	18 (±10.6)	15 (±0)	20 (±0)
Apalachicola National Forest, FL	W	47(±6.2)	120(±62.3)	200(±122)	193(±152)	238(±123.7)
Flint Rock properties, FL	W	6 (±0)	9 (±2.8)	15 (±2.1)	13 (±0)	13 (±0)
St. Marks National Wildlife Refuge, FL	W	37(±0.5)	47 (±4.7)	57 (±13.7)	82 (±26.2)	102 (±54.4)
Total		92(±6.4)	187(±63)	290(±123.2)	303(±154.3)	373(±135.2)

5.2 Development of Climate Change Scenarios

The most recent Assessment Report of the Intergovernmental Panel on Climate Change (IPCC, 2014) predicts broad-scale global climate changes over the 21st Century, which will likely have negative impacts on the suitability of existing breeding sites and surrounding uplands for frosted flatwoods salamanders. Global surface temperatures are expected to rise 0.3 – 0.7℃ by 2035 and 0.3 – 4.8℃ by 2100 depending on greenhouse gas emissions levels resulting in increased frequency and duration of periods of extreme high temperatures (IPCC, 2014). Predicted global precipitation changes are highly variable and are uncertain for Florida, however an increase in the frequency and intensity of drought and flood events are anticipated (IPCC, 2014, Kirtman et al., 2017). The most recent IPCC report predicts sea level rise increases of 0.26 – 0.82 m by Year 2100, depending on the emissions scenario (IPCC, 2014). However, several recent models have suggested higher levels of sea level rise may be plausible (Kopp et al., 2014; Le Bars et al., 2017; Sweet et al., 2017). Sweet et al. (2017) projected global mean sea level increases of 0.3 – 2.5 m by 2100 with higher levels surrounding U.S. coasts under the highest emissions scenarios. Historically, sea level rise in Florida has been generally consistent with the global mean although some local variations have been observed along the Gulf Coast (Geselbracht et al., 2015). With the increasing frequency and intensity of storms, increased inundation from storm surges will exacerbate the loss and degradation of freshwater wetlands in coastal areas leading to vegetation and salinity changes (IPCC, 2014).

To assess the future impact of these projected climate changes, we determined the likely impact on current breeding ponds based on the best available climate change sources for Florida. When Florida data were unavailable, we used regional predictions or global predictions from the most recent IPCC report. Climate change scenarios are currently based on a GIS analysis of potential inundation and vegetation changes of current breeding ponds under different sea level rise projections. Future work will include additional expert elicitation of predicted climate change temperature, precipitation, and storm surge impacts under different emissions scenarios. We feel the impact of these climate changes on frosted flatwoods salamander populations is best estimated by experts familiar with the species given the lack of data available. We include information on predicted regional climate changes in temperature and precipitation, as well as projected changes in storm surge inundation probabilities under plausible sea level rise scenarios here to lay the foundation for a future expert elicitation and provide information on potential impacts to species populations.

5.2.1 Sea Level Rise

Sea level rise is predicted to reduce frosted flatwoods salamander breeding and upland habitat through direct inundation of coastal areas and coastal habitat changes due to soil and water salinity changes (Carter, 2014). To model the impact of future sea level rise on frosted flatwoods salamander breeding sites, we conducted a GIS analysis to determine which currently active Florida breeding sites would be inundated or unsuitable in the future based on sea level rise and marsh migration projection data from the National Oceanic and Atmospheric Administration's (NOAA) Sea Level Rise Viewer (https://coast.noaa.gov/digitalcoast/tools/slr.html). This tool uses local tide station data and elevation data to provide a range of potential sea level rise scenarios (1–6 feet based on different emissions scenarios) at different time scales. The marsh migration data layers of this tool display changes in the distribution of different coastal habitat types based on different sea level rise scenarios by using habitat thresholds based on elevation data and the relationship of each habitat type to tidal influence.

The relationship between the amount of sea level rise and the various emissions scenarios in the NOAA Sea Level Rise Viewer are based on recently revised global mean sea level rise scenarios presented by the Federal Interagency Sea Level Rise and Coastal Flood Hazard Scenarios and Tools Task Force in Sweet et al. (2017), which forecasts greater sea level rise impacts under the various emissions scenarios than the most recent IPCC report (IPCC, 2014). Multiple recent studies have suggested faster rates of sea level rise under current emissions scenarios or argued for greater consideration of plausible, but less likely predicted levels of sea level rise from existing projections (Hall et al. 2016, Jackson and Jevrejeva 2016, Kopp et al. 2014, Parris et al. 2012). Sweet et al. (2017) argued for the consideration of an extreme worst-case scenario of 2.5m global mean sea level rise by 2100 citing the accelerating loss of Greenland and Antarctic ice sheets and the importance of including worst-case scenarios in adaptation planning. This worst-case scenario has a 0.1% probability of occurring by 2100 (Sweet et al., 2017). In addition, sea level rise along U.S. coasts is predicted to be significantly greater than the global mean sea level rise under higher emissions scenarios (Sweet et al., 2017).

In recognition of the fact that these studies are based on more recent data than the most recent IPCC report and consider a wider range of plausible sea level rise predictions, we have chosen to base our sea level rise analyses on the full range of sea level rise scenarios presented by Sweet et al. (2017) and used in the NOAA Sea Level Rise Viewer to consider the full range of potential impacts to the species. The relationship between the sea level rise scenarios used in the NOAA Sea Level Rise Viewer based on Sweet et al. (2017) and the RCP emissions scenarios used in the latest IPCC report for the Years 2050 and 2100 is provided in Tables 5.8–5.9.

Table 5.8. *Mean and (range) of sea level rise projected for the Florida panhandle by 2050 for different CO_2 emissions scenarios as presented by the most recent IPCC report (IPCC, 2014) and Sweet et al. (2017) as used in the NOAA Sea Level Rise Viewer.*

	Predicted Sea Level Rise 2050	
Emissions Scenario	**NOAA Sea Level Rise Viewer**	**IPCC (2014)**
Low / RCP 2.6	0.16 m	0.24 m (0.17–0.32 m)
Medium Low / RCP 4.5	0.24 m	0.26 m (0.19–0.33 m)
Medium	0.34 m	NA
Medium High / RCP 6.0	0.44 m	0.25 m (0.18–0.32 m)
High / RCP 8.5	0.54 m	0.29 m (0.22–0.38 m)
Extreme	0.63 m	NA

Table 5.9*. Mean and (range) of sea level rise projected for the Florida panhandle by 2081-2100 for different CO_2 emissions scenarios as presented by the most recent IPCC report (IPCC, 2014) and Sweet et al. (2017) as used in the NOAA Sea Level Rise Viewer.*

	Predicted Sea Level Rise 2081–2100	
Emissions Scenario	**NOAA Sea Level Rise Viewer**	**IPCC (2014)**
Low / RCP 2.6	0.3 m	0.44 m (28–0.61 m)
Medium Low / RCP 4.5	0.5 m	0.53 m (0.36–0.71 m)
Medium	1.0 m	NA
Medium High / RCP 6.0	1.5 m	0.55 m (0.38–0.73 m)
High / RCP 8.5	2.0 m	0.74 m (0.52–0.98 m)
Extreme	2.5 m	NA

To examine the potential impact of sea level rise over a range of emissions scenarios and intermediate and long-term time scales, we determined which currently active breeding ponds would potentially be inundated in the future by examining their locations in relation to projected sea levels for the years 2050 and 2100 for the different sea level rise scenarios presented in Sweet et al (2017; see Tables 5.8 and 5.9). These sea level rise projections represent a range of potential sea levels based on different emissions scenarios (low–high emissions). In addition, we also determined if rising sea level would result in habitat changes in currently active breeding wetlands by examining their location in relation to the marsh migration data layers of the NOAA Sea Level Rise Viewer at each sea level predicted for the Years 2050 and 2100. Breeding ponds were considered no longer suitable if marsh migration data layers reflected changes to open water, unconsolidated shore, an estuarine or brackish marsh, or a freshwater marsh that appeared to be connected to brackish or transitional marshes.

Based on our GIS analysis, we found that no currently active frosted flatwoods salamander breeding ponds are projected to be inundated under any of the sea level rise scenarios by the year 2050. However, habitat changes in breeding ponds (conversions to unsuitable habitat as a result of fire suppression/absence, marsh migration, etc.) on St. Marks NWR are predicted to occur under the lowest emissions scenarios beginning at a sea level rise of 0.5 feet (Figure 5.2; Table 5.10). Under the extreme sea level rise scenario (sea level rise of two feet), habitat changes would result in the loss of 30 of the 36 currently active breeding ponds on St. Marks NWR.

Table 5.10. *Number of currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds lost to sea level rise and marsh migration under different emissions scenarios by Year 2050. NOAA Sea Level Rise Viewer marsh migration data was not available for 0.24 m so the number of active breeding ponds affected by sea level rise at the medium low or RCP 4.5 emissions scenario could not be estimated.*

Emissions Scenario	**Predicted Sea Level Rise**	**No. Current Breeding Ponds Lost**	**Reason for Loss**
Low / RCP 2.6	0.16 m (~0.5 ft)	1 (St. Marks NWR)	Habitat changes
Medium Low / RCP 4.5	0.24 m (~ 0.75 ft)	Not Available (>1)	Habitat changes
Medium	0.34 m (~ 1 ft)	9 (St. Marks NWR)	Habitat changes
Medium High / RCP 6.0	0.44 m (~1.5 ft)	26 (St. Marks NWR)	Habitat changes
High / RCP 8.5	0.54 m (~1.75 ft)	29 (St. Marks NWR)	Habitat changes
Extreme	0.63 m (~ 2 ft)	30 (St. Marks NWR)	Habitat changes

A)

C)

D)

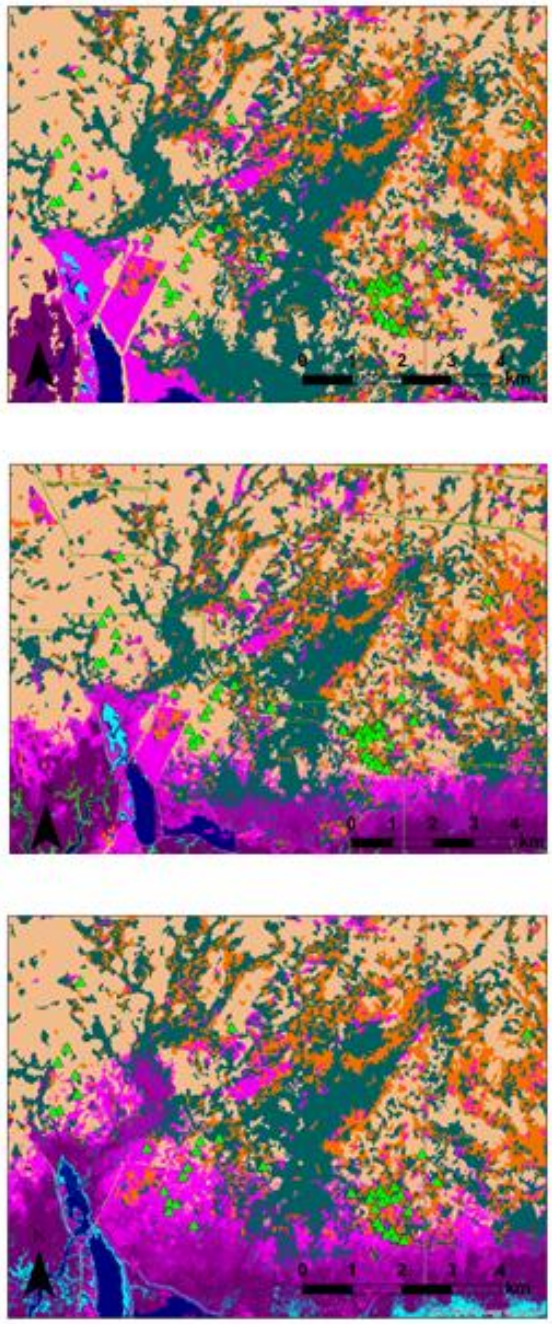

Figure 5.2. *Currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds at St. Marks National Wildlife Refuge and Flint Rock (green triangles) in relation to sea level rise and marsh migration under different emissions scenarios by Year 2050. A) current landcover B) land cover legend C) 0.5 ft sea level rise (low emissions) D) 2 ft sea level rise (extreme scenario) Source: NOAA Sea Level Rise Viewer*

When Year 2100 sea level scenarios were examined, a range of impacts to current breeding ponds on St. Marks National Wildlife Refuge and Flint Rock were observed. Inundation of breeding ponds on St. Marks National Wildlife Refuge was projected to occur under higher emissions levels by 2100 beginning at a sea level rise of 4 feet (Figure 5.3). At a sea level rise of 6 feet, which is possible under high emission levels, all but one St. Marks National Wildlife Refuge and four Flint Rock breeding ponds are projected to be inundated. The NOAA Sea Level Rise Viewer does not provide data for a sea level rise of 6.5 feet or 8 feet, which would be reached under the high and extreme sea level rise scenarios. In addition to direct inundation, rising sea levels will result in habitat changes in coastal natural communities, which may make current breeding ponds unsuitable for species reproduction.

When coastal habitat changes were considered for each sea level rise scenario using the NOAA marsh migration data layers, the loss of current breeding ponds on St. Marks National Wildlife Refuge began at a sea level rise of one foot and increased until the loss of all breeding ponds on the refuge and nearby Flint Rock area at a sea level rise of six feet (Table 5.11; Figure 5.4). None of the currently active Apalachicola National Forest breeding ponds or the remaining breeding pond on Fort Stewart in Georgia were impacted by any of the sea level rise scenarios or associated habitat changes predicted by Year 2100 (Figure 5.5). However, it is important to consider that sea level rise will not cease by 2100 regardless of the emissions scenario, and therefore the number of breeding ponds inundated or lost to habitat changes will be greater than considered here at later time periods (Sweet et al. 2017, Table 5.12).

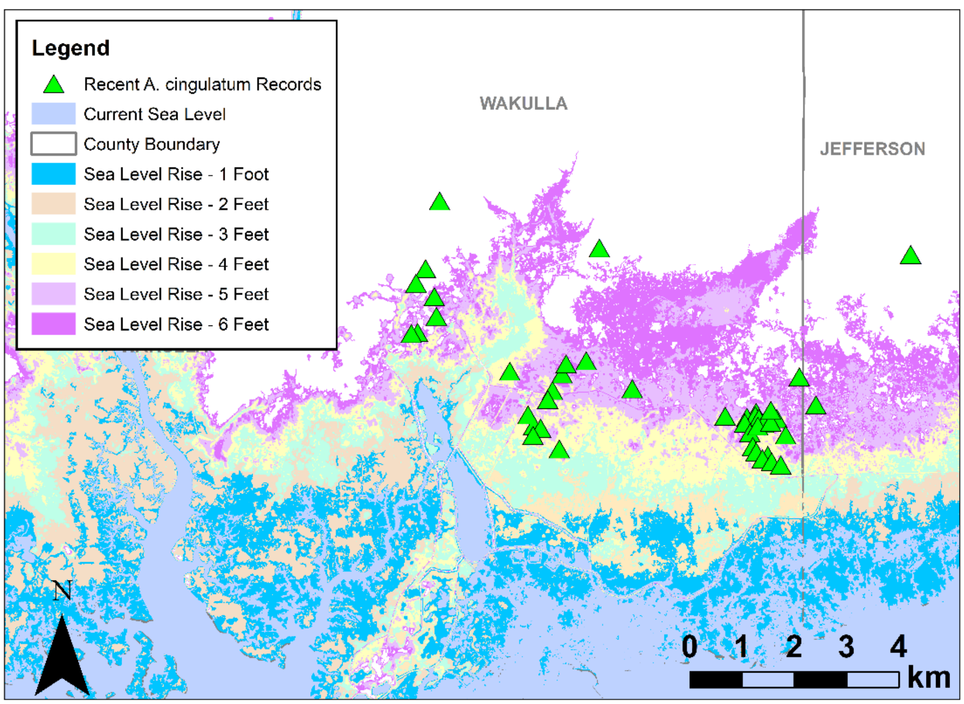

Figure 5.3*. Currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on St. Marks National Wildlife Refuge and Flint Rock (green triangles) in relation to sea level rise (1-6 ft) under different emissions scenarios by Year 2100. Source: NOAA Sea Level Rise Viewer*

Table 5.11. *Number of currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds lost to sea level rise and marsh migration under different emissions scenarios by Year 2100.*

Emissions Scenario	**Predicted Sea Level Rise**	**No. Current Breeding Ponds Lost**	**Reason for Loss**
Low / RCP 2.6	0.3 m (~ 1 foot)	9 (St. Marks NWR)	Habitat changes
Medium Low / RCP 4.5	0.5 m (~ 2 feet)	27 (St. Marks NWR)	Habitat changes
Medium	1.0 m (~ 3 feet)	35 (St. Marks NWR/Flint Rock)	Habitat changes
Medium High / RCP 6.0	1.5 m (~ 5 feet)	41 (St. Marks NWR/Flint Rock)	Inundation and habitat changes
High / RCP 8.5	2.0 m (~6.5 feet)	42 (St. Marks NWR/Flint Rock)	Inundation and habitat changes
Extreme*	2.5 m (~ 8 feet)	42 (St. Marks NWR/Flint Rock)	Inundation and habitat changes

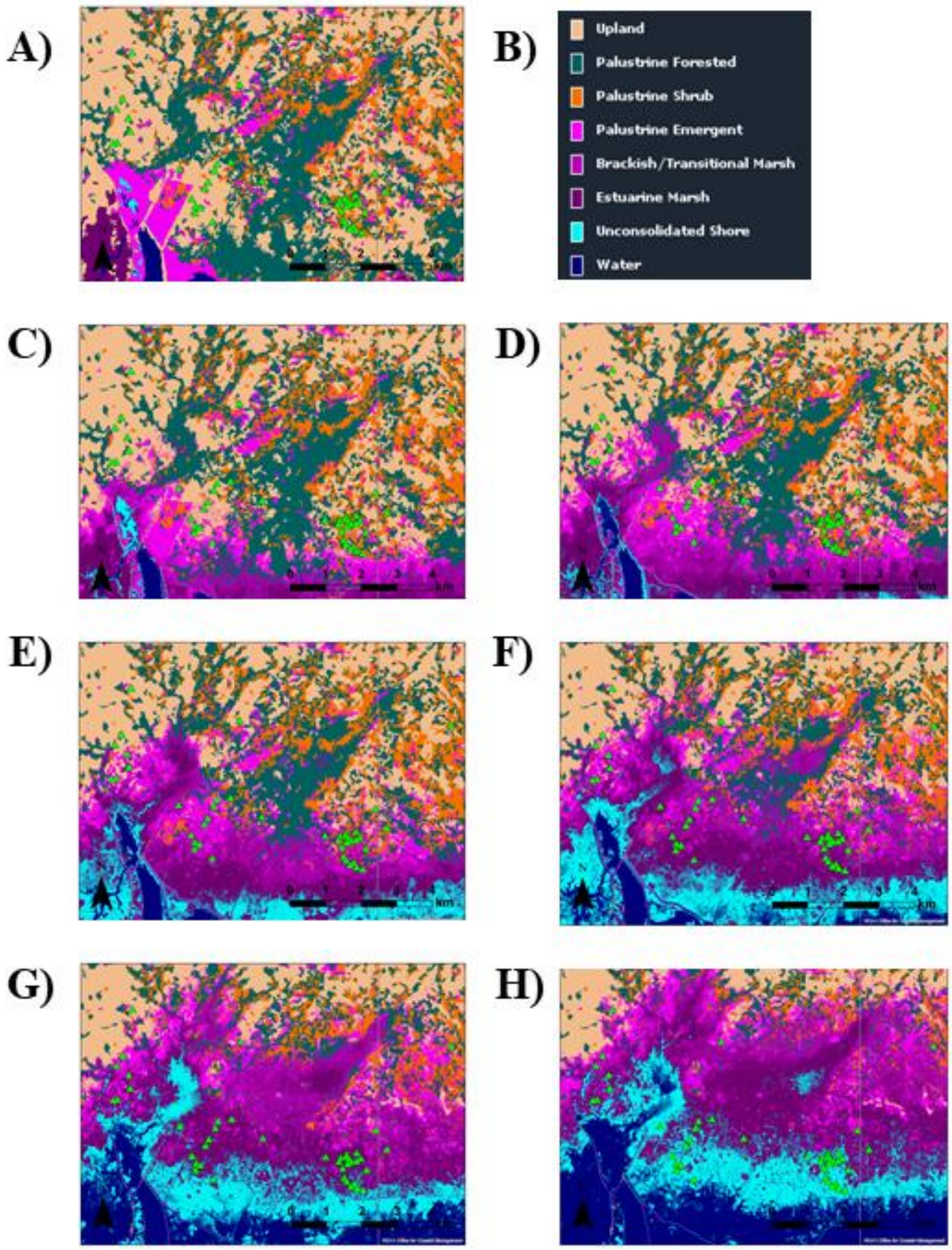

Figure 5.4. *Currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds at St. Marks National Wildlife Refuge and Flint Rock (green triangles) in relation to sea level rise and marsh migration under different emissions scenarios by Year 2100. A) current landcover B) land cover legend C) 1 ft sea level rise (low emissions) D) 2 ft sea level rise (medium low emissions) E) 3 ft sea level rise (medium emissions) F) 4 ft sea level rise G) 5 ft sea level rise (medium high emissions) H) 6 ft sea level rise (high emissions) Source: NOAA Sea Level Rise Viewer*

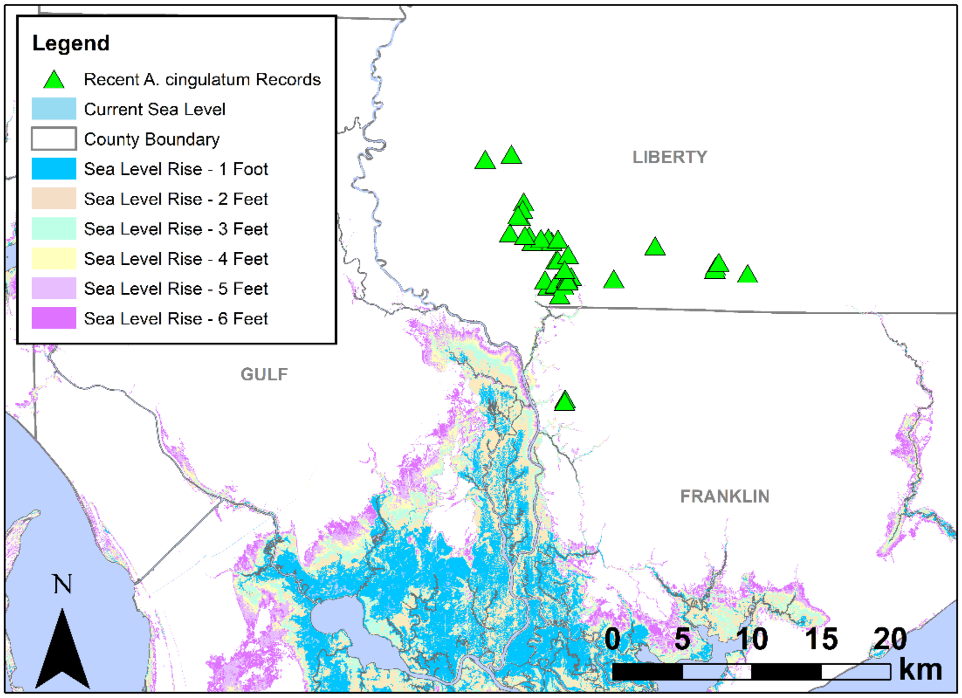

Figure 5.5. *Currently active frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds on Apalachicola National Forest (green triangles) in relation to sea level rise (1-6 ft) under different emissions scenarios by Year 2100. Source: NOAA Sea Level Rise Viewer*

Table 5.12. *Global mean sea level rise from year 2000 in meters under each emissions scenario over time. Only median values are presented. Data from Sweet et al. (2017).*

Sea Level Rise Scenario	**2020**	**2030**	**2040**	**2050**	**2070**	**2100**	**2150**	**2200**
Low	0.06	0.09	0.13	0.16	0.22	0.3	0.37	0.39
Intermediate-Low	0.08	0.13	0.18	0.24	0.35	0.5	0.73	0.95
Intermediate	0.10	0.16	0.25	0.34	0.57	1.0	1.8	2.8
Intermediate-High	0.10	0.19	0.30	0.44	0.79	1.5	3.1	5.1
High	0.11	0.21	0.36	0.54	1.0	2.0	4.3	7.5
Extreme	0.11	0.24	0.41	0.63	1.2	2.5	5.5	9.7

5.2.2 Storm Surge

The NOAA Ecological Effects of Sea Level Rise in the Northern Gulf of Mexico Project modeled simulated 100 and 500-year storm surges (based on 1% and 0.2% annual chance of flooding, respectively) under current conditions and at four different sea level rise scenarios (low, intermediate low, intermediate high, and high carbon emissions) for the Year 2100 based on sea level scenarios presented in Parris et al. (2012). To determine the likelihood that current breeding ponds will be inundated by future storm surges, we used ArcGIS to overlay the storm surge maps from this project over the current breeding locations at each of the modeled sea level rise scenarios. Based on this analysis, all current breeding ponds on St. Marks National Wildlife Refuge and the Flint Rock properties are already in danger of inundation during 1% and 0.2% (annual) chance storm surge events under current sea levels, as well as under all future sea level rise scenarios (Figures 5.6-5.7). In contrast, no active breeding ponds on Apalachicola National Forest are at risk of inundation from storm surge under current or projected future sea level rise scenarios modeled for Year 2100, although unoccupied parts of southern Apalachicola National Forest and historically occupied areas on Tate's Hell State Forest will be inundated under the higher emissions (intermediate high and high emissions) future sea level rise scenarios. However, even though currently occupied areas on Apalachicola National Forest are not projected to be impacted by storm surge by Year 2100, sea level rise will not cease by 2100 under any emissions scenario and therefore it is likely that the currently active breeding ponds on this property will become vulnerable to storm surge inundation at later time periods (Sweet et al. 2017).

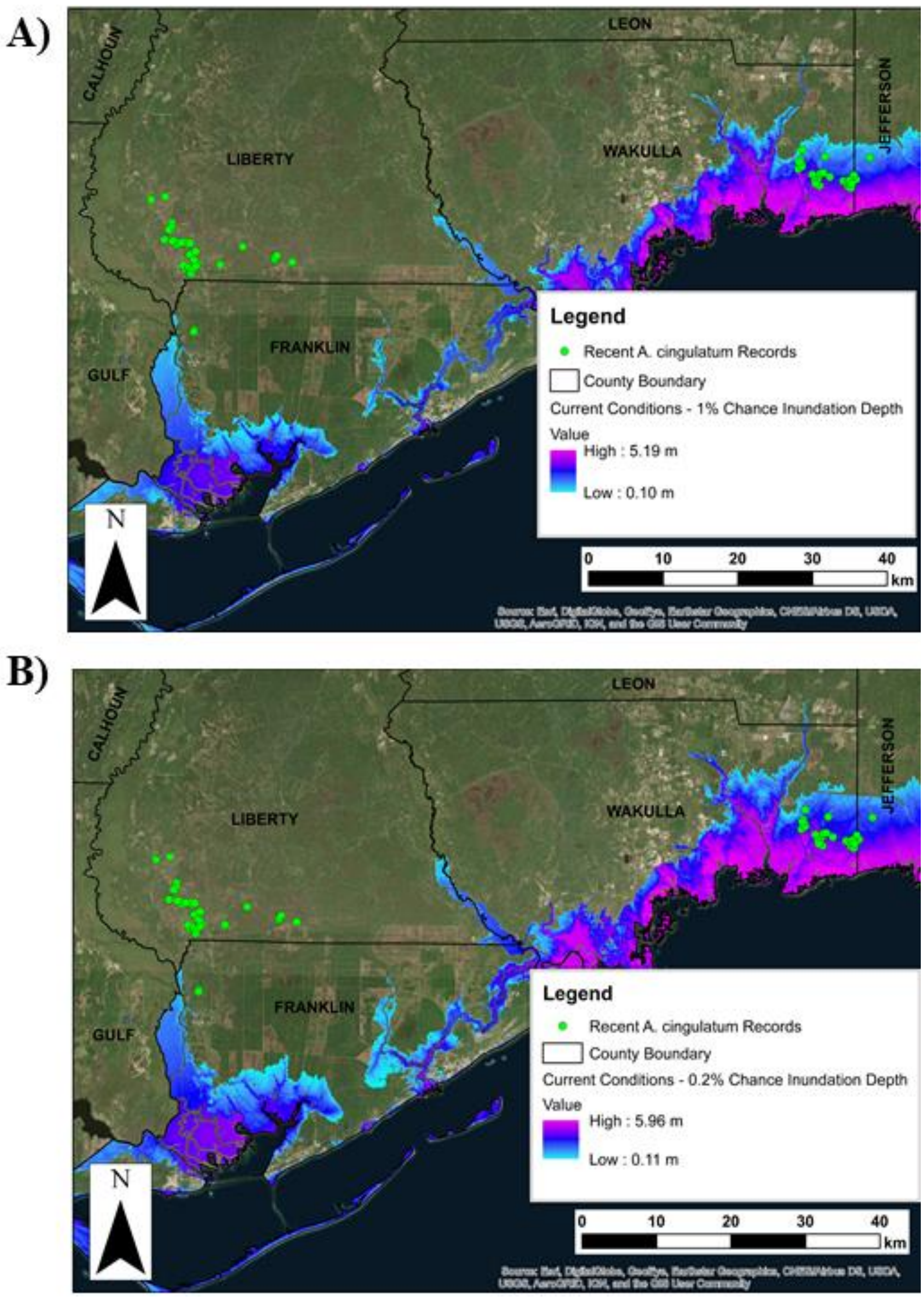

Figure 5.6. *Currently active Florida frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds in relation to A) 1% and B) 0.2% annual chance storm surge inundation depths at current sea levels. Source: NOAA Ecological Effects of Sea Level Rise in the Northern Gulf of Mexico Project*

A)

B)

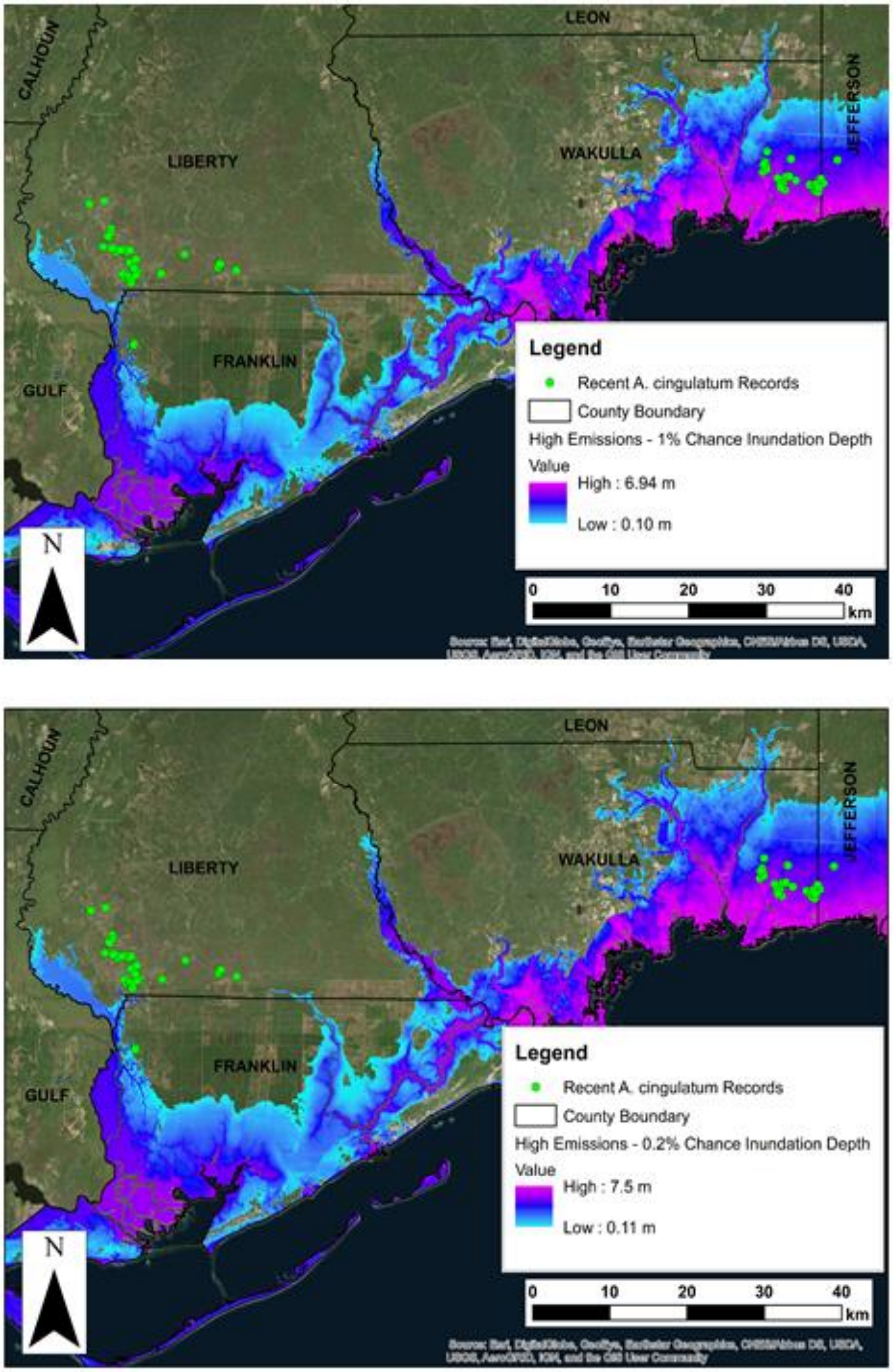

Figure 5.7. *Currently active Florida frosted flatwoods salamander (Ambystoma cingulatum) breeding ponds in relation to future (Year 2100) 1% and 0.2% annual chance storm surge inundation depths. A) high emissions scenario 1% annual probability for Year 2100 B) high emissions scenario 0.2% annual probability for Year 2100 Source: NOAA Ecological Effects of Sea Level Rise in the Northern Gulf of Mexico Project*

5.2.3 Temperature and Precipitation Changes

In the future, we plan to use expert elicitation to determine the effect of projected temperature and precipitation climate changes on the number of active breeding ponds on currently occupied properties by providing the following information to species experts: 1) the projected seasonal temperature and precipitation changes for the Florida panhandle under a range of emissions scenarios (Tables 5.13–5.14); 2), the average annual and average winter and summer temperatures for the Florida panhandle since 1895 (Figures 5.8–5.10); and 3), and the average, minimum and maximum thermal maxima measured for *Ambystoma* species (Figure 5.11). Species experts will be encouraged to consider direct mortality, reduction in breeding pond hydroperiods due to increased evapotranspiration and drought, and changes/degradation in wetland and upland habitat due to increased fire intensity. The currently available data suggest further loss of breeding ponds in addition to those predicted to be impacted by sea level rise under the higher emissions scenarios due to hydrological changes and increased wetland plant evapotranspiration as temperatures increase 2–5°C (Table 5.13). In addition, under higher emissions scenarios by 2050, maximum summer temperatures may exceed the critical thermal maxima of some *Ambystoma* species (Table 5.13, Figures 5.10–5.11) resulting in direct individual mortality. The critical thermal maxima of this species is unknown, but it is likely similar to other species within the same genus. Hutchinson (1961) observed the critical thermal maxima of five southeastern *Ambystoma* species ranging from 36.25–37.77°C.

Table 5.13. *Median projected seasonal temperature increases for the Florida panhandle under various emissions scenarios. Source: IPCC 2013 (This source provides regional climate projections vs. the global predictions in IPCC 2014)*

	Years 2046–2065				**Years 2081–2100**			
Climate Scenario	**Dec–Feb**	**March–May**	**June – Aug**	**Sept–Nov**	**Dec – Feb**	**March–May**	**June–Aug**	**Sept–Nov**
RCP 2.6	0.5–1°C	1–1.5°C	1–1.5°C	1–1.5°C	0.5–1°C	1–1.5°C	1–1.5°C	0.5–1°C
RCP 4.5	1–1.5°C	1.5–2°C	1.5–2°C	1.5–2°C	1.5–2°C	1.5–3°C	1.5–3°C	2–3°C
RCP 6.0	1–1.5°C	1–1.5°C	1–1.5°C	1–1.5°C	1.5–2°C	2–3°C	2–3°C	2–3°C
RCP 8.5	1.5–2°C	2–3°C	2–3°C	2–3°C	3–4°C	3–4°C	3–5°C	4–5°C

Table 5.14. *Median projected precipitation changes for the Florida panhandle under various emissions scenarios. Precipitation changes are in percent change from average seasonal conditions. *Indicates projections are within the current normal range of variation. Only one projection is outside of the current range of variation. Source: IPCC 2013 (This source provides regional climate projections vs. the global predictions in IPCC 2014)*

	Years 2046–2065		**Years 2081–2100**	
Climate Scenario	**October–March**	**April –September**	**October–March**	**April –September**
RCP 2.6	0-10%*	0-10%*	0-10%*	0-10%*
RCP 4.5	0-10%*	0-10%*	0-10%	0-10%*
RCP 6.0	0-10%*	0-10%*	0-10%*	0-10%*
RCP 8.5	0-10%*	0-10%*	0-20%	0-10%*

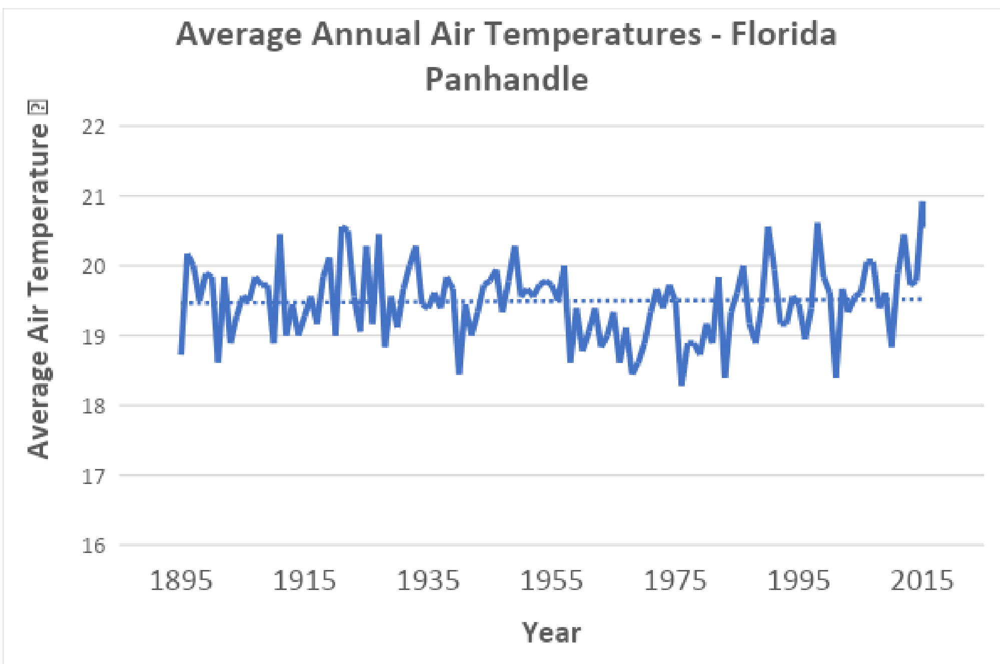

Figure 5.8. *Long-term trends in average annual air temperature in Florida panhandle 1895–2017. Source: NOAA National Climate Data Center.*

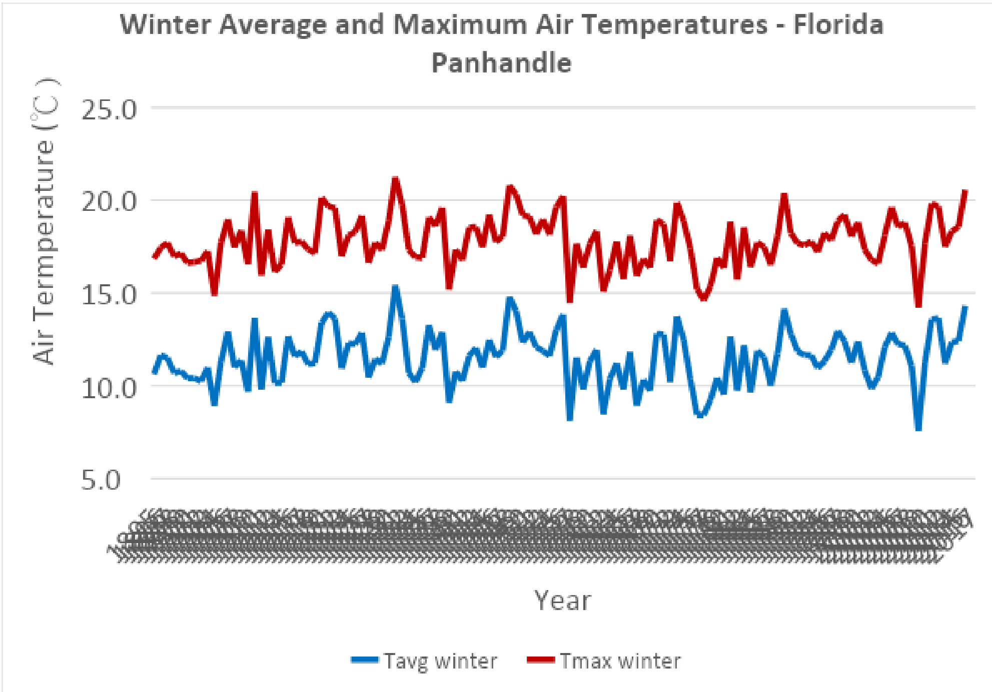

Figure 5.9. *Long-term trends in average winter (December – February) air temperature in Florida panhandle 1895–2018. Source: NOAA National Climate Data Center.*

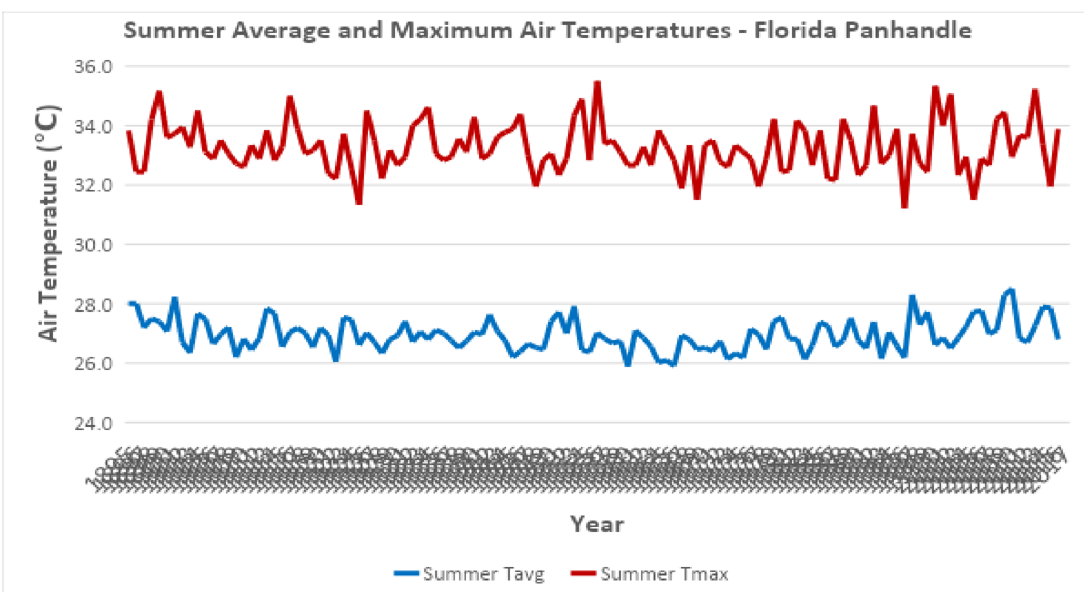

Figure 5.10. *Long-term trends in average summer (June – August) air temperature in Florida panhandle 1895–2017. Source: NOAA National Climate Data Center.*

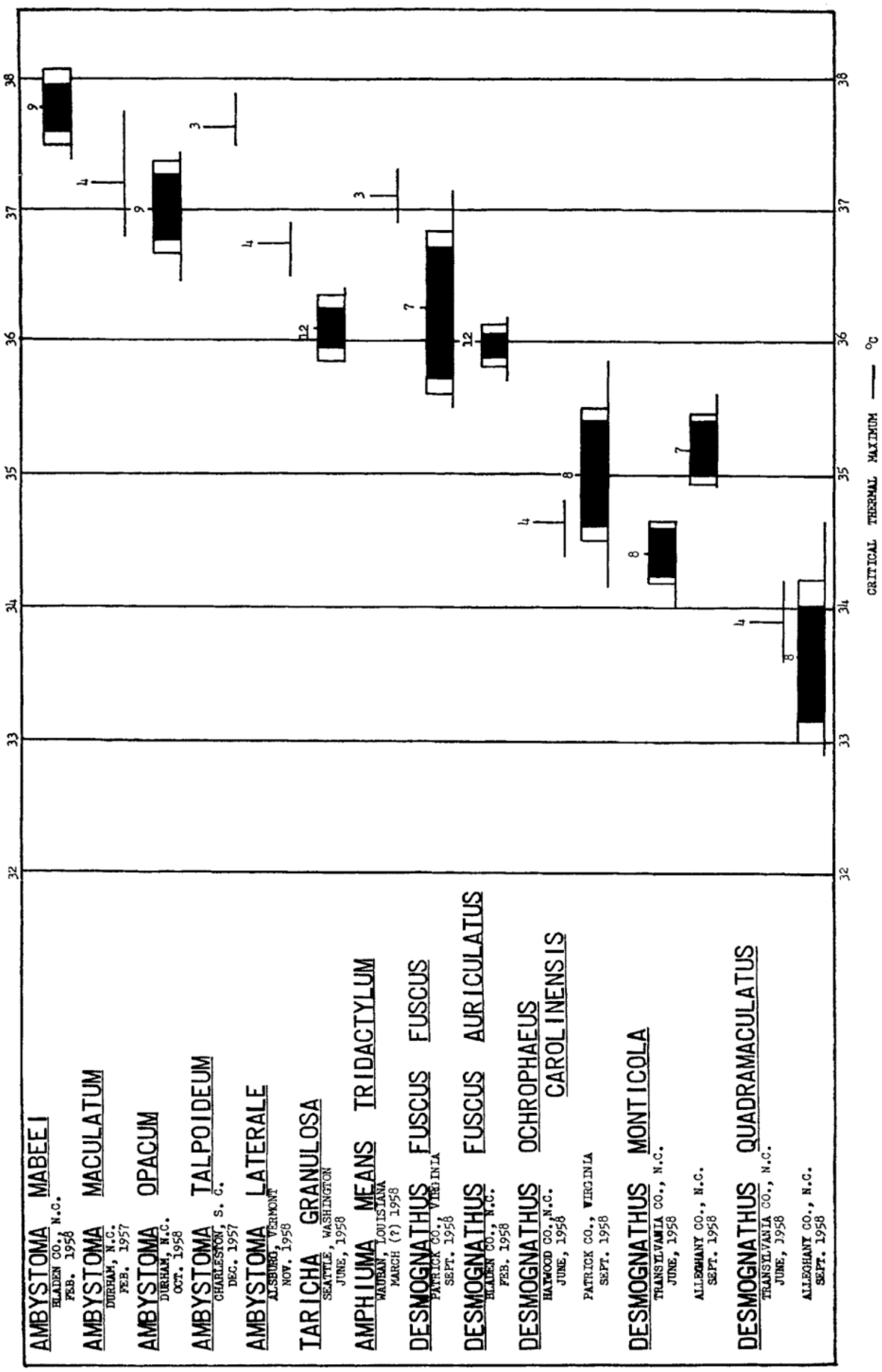

Figure 5.11. *Published data on thermal maxima of salamanders including Ambystoma species. Source: Hutchinson 1961.*

5.3 Creation of Combined Management and Climate Scenarios

We created combined management and climate scenarios to estimate the future impacts of climate change under different wetland management scenarios. We calculated the difference from current conditions for each combined management and climate change scenario by subtracting the number of breeding ponds lost to sea level rise and marsh migration at each emissions scenario from the number of wetlands predicted for Years 2050 and 2100 under each management scenario (Table 5.15). We used total numbers active breeding ponds based on predictions for individual occupied properties (as opposed to range-wide estimates) for these calculations because of the need to subtract inundated ponds from specific properties. The number of wetlands impacted by sea level rise for the wetland restoration scenario was calculated by determining the percentage of currently occupied wetlands projected to be impacted by sea level rise under each climate change scenario on St. Marks National Wildlife Refuge and Flint Rock properties and subtracting this percentage from the projected wetlands for these properties for the wetland restoration scenario. This approach assumes that wetland restored in the future will have the same distribution as currently occupied wetlands, which may not be the case. We did not include potential impacts from storm surge in our estimates because the long-term impacts of storm surge on breeding ponds is unclear, although it is likely that the true number of breeding ponds under each scenario would be less than presented here because of long-term effects of inundation by storm surge in some breeding ponds.

Table 5.15. *Final scenarios 1–12 created from combining climate and management scenarios for each time period.*

	Combined Management/Climate Change Scenarios		
Climate Scenario	**Wetland Loss**	**Wetland Maintenance**	**Wetland Restoration**
Low Emissions (RCP 2.6)	Scenario 1: Low Emissions & Wetland Loss	Scenario 5: Low Emissions & Wetland Maintenance	Scenario 9: Low Emissions & Wetland Restoration
Medium Low Emissions (RCP 4.5)	Scenario 2: Medium Low Emissions & Wetland Loss	Scenario 6: Medium Low Emissions & Wetland Maintenance	Scenario 10: Medium Low Emissions & Wetland Restoration
Medium High Emissions (RCP 6.0)	Scenario 3: Medium High Emissions & Wetland Loss	Scenario 7: Medium High Emissions & Wetland Maintenance	Scenario 11: Medium High Emissions & Wetland Restoration

High Emissions (RCP 8.0)	Scenario 4: High Emissions & Wetland Loss	Scenario 8: High Emissions & Wetland Maintenance	Scenario 12: High Emissions & Wetland Restoration

The projected number of active breeding ponds in Year 2050 for all combinations of management and climate scenarios is presented in Table 5.16. The number of active breeding ponds is predicted to decrease to only three wetlands on ANF under all wetland loss scenarios (Scenarios 1-4; where no species-specific wetland management or restoration is applied) regardless of the climate change impacts. Therefore, for all wetland succession scenarios, there would be no representation of the eastern clade because the one eastern clade pond would have become inactive after ten years due to a lack of wetland management. In addition, there would be no representation or redundancy of the western clade beyond the few ponds left on Apalachicola National Forest because no wetlands would remain on SMNWR. The resiliency of populations in the remaining ponds would also likely be reduced due to the decreased habitat quality of any remaining breeding wetlands. Under the wetland maintenance scenarios (Scenarios 5-8), in which only currently occupied wetlands are the focus of species-directed wetland management, the number of active breeding ponds will decrease from between one to 29 wetlands by the year 2050 depending on the climate change scenario and corresponding sea level rise. Under these scenarios, both the eastern and western clades would be represented, but the redundancy of the western clade would be reduced on St. Marks National Wildlife Refuge due to sea level rise impacts. There would also be no redundancy of populations in the eastern clade as only a single breeding pond would remain on Fort Stewart. Under all wetland restoration scenarios (Scenarios 9-12; where species-specific management and restoration of all potential ponds is maximized), the number of active breeding wetlands is estimated to more than triple regardless of sea level rise impact. However, there is less expert agreement on the numbers of resulting ponds from the wetland restoration scenarios as indicated by the high standard deviation values. Thus, the wetland management scenario plays a profound role in determining the future number of active breeding ponds for this species by the year 2050. However, it should be noted that increased temperatures under the high climate change emissions scenarios (Scenarios 4, 8, 12) will cause additional negative impacts on the number of active breeding ponds that are not considered here. The predicted temperature increases under these high emissions scenarios will likely result in increased drought intensity and frequency, as well as increased evapotranspiration in wetlands, which will result in decreased breeding pond hydroperiods. This will in turn reduce the resiliency of all populations by decreasing larval survival and recruitment. Individual animals may also be heat stressed during the summer months as the projected mean temperature would be 2–3℃ higher, which would put the temperature close to the recorded critical thermal maxima of other *Ambystoma* species on the hottest summer days since the average maximum summer temperature is currently 34℃ (Hutchinson, 1961; Table 5.13, Figure 5.10–5.11).

Table 5.16. *Projected mean (± SD) total number of active breeding ponds for frosted flatwoods salamanders in Year 2050 under different breeding pond management and climate emissions scenarios based on GIS analysis of sea level rise projections and wetland loss/succession rates*

from land managers. NOAA Sea Level Rise Viewer marsh migration data was not available for 0.24 m so the number of active breeding ponds affected by sea level rise at the medium low or RCP 4.5 emissions scenario in Year 2050 could not be estimated.

	Combined Management/Climate Scenarios		
Climate Scenarios	**Wetland Loss**	**Wetland Maintenance**	**Wetland Restoration**
Low Emissions (RCP 2.6)	3 (±2.9)	81 (±0.0)	300 (±154.3)
Medium Low Emissions (RCP 4.5)	3 (±2.9)	Not Available	Not Available
Medium High Emissions (RCP 6.0)	3 (±2.9)	56 (±0.0)	243 (±154.3)
High Emissions (RCP 8.0)	3 (±2.9)	53 (±0.0)	237 (±154.3)

The projected number of active breeding ponds in Years 2100 for all combinations of management and climate scenarios is presented in Table 5.17. For all combinations of the wetland succession scenarios (Scenarios 1–4), there would be no representation of the eastern clade because the one eastern clade pond would have become inactive after ten years due to a lack of wetland management. In addition, there is no redundancy of the western clade under any wetland succession scenario by the Year 2100 because wetland succession would decrease the number of active breeding ponds to only one breeding pond on Apalachicola National Forest by this time regardless of climate change scenario. Under the wetland maintenance scenarios (Scenarios 5–8), the number of currently active breeding ponds would decrease somewhere between nine and 42 wetlands by the year 2100 depending on the climate change scenario and corresponding sea level rise. Under the wetland restoration scenarios (Scenarios 9–12), losses to sea level rise on St. Marks National Wildlife Refuge are mitigated by wetland restoration efforts elsewhere resulting in tripling or quadrupling the current number of active breeding ponds leading to more representative and redundant populations in the eastern and western clades. However, there is less expert agreement on the numbers of resulting ponds from the wetland restoration scenarios as indicated by the high standard deviation values. Under all scenarios, population resiliency in the remaining ponds would decrease due to climate change impacts including habitat degradation in the ponds due to increased drought, extreme fires with higher intensities, and decreased hydroperiods due to increased evapotranspiration within wetlands. In addition, individual animals would potentially be heat stressed during the summer under all but the lowest climate emissions scenarios (Scenarios 1,5,9) as the projected mean temperature would be 1.5–5℃ higher (depending on the emissions scenario), which would put the temperature close to the critical thermal maxima of other *Ambystoma* species since the average maximum summer temperature is currently 34℃ (Table 5.13, Figure 5.10–5.11). Hutchinson (1961) recorded critical thermal maxima of 36.25–37.77℃. for five other *Ambystoma* species However, the thermal maximum of this species is unknown. In addition, this species is believed to be fossorial during the summer months, and the degree to which individuals would be affected by the projected temperature increases during this season is unknown. While the impacts of future temperature and drought stress on this species and its breeding ponds is hard to quantify, these stressors would likely decrease the number of breeding ponds/populations and further reduce the number of breeding ponds below numbers presented in Table 5.17.

Table 5.17. *Projected mean (± SD) total number of active breeding ponds for frosted flatwoods salamanders in Year 2100 under different breeding pond management and climate emissions scenarios based on GIS analysis of sea level rise projections and wetland loss/succession rates from land managers.*

	Combined Management/Climate Scenarios		
Climate Scenarios	**Wetland Loss**	**Wetland Maintenance**	**Wetland Restoration**
Low Emissions (RCP 2.6)	1 (±1.2)	73 (±0.0)	347 (±135.2)
Medium Low Emissions (RCP 4.5)	1 (±1.2)	55 (±0.0)	296 (±135.2)
Medium High Emissions (RCP 6.0)	1 (±1.2)	41 (±0.0)	260 (±135.2)
High Emissions (RCP 8.0)	1 (±1.2)	40 (±0.0)	258 (±135.2)

5.4 Summary of Changes in Future Conditions

The future of frosted flatwoods salamanders greatly depends on how land managers of occupied properties approach wetland management. While both sea level rise and increasing temperatures due to climate change are predicted to decrease the number of breeding ponds and resiliency of populations, particularly by 2100, the choice of management scenario has profound impacts on the number of breeding ponds in both the short and long-term. If species-specific wetland management (regularly burning of breeding ponds when they are dry) is not conducted, most active breeding ponds will become inactive by the Year 2050. However, it is not enough to simply actively manage the breeding ponds that are currently occupied, as sea level rise and associated marsh migration will result in some loss of currently active breeding ponds at St. Marks National Wildlife Refuge under all climate change scenarios by the Year 2050.

To avoid further population declines and ensure populations are as resilient as possible in the face of anticipated changes to the climate, land managers will need to engage in and maximize the active restoration of potentially suitable breeding wetlands to offset anticipated breeding pond losses to sea level rise. Wetland restoration efforts should be primarily focused on Apalachicola National Forest and Fort Stewart, which are not anticipated to be affected by sea level rise in the next 80 years, as well as other inland areas with potentially suitable habitat in the range of the species. Similarly, long-term protection (via acquisition or easements) should focus on this portion of the species range. Currently, many managers lack the resources to maintain all active breeding ponds or the ponds they restore in suitable condition. Therefore, efforts should be made to remove barriers to and provide support for wetland restoration and management on occupied and potentially suitable properties.

In addition to wetland restoration efforts, salamander translocations to restored wetlands may be necessary if salamanders fail to colonize restored wetlands.

These simulations give insights into how to recovery the frosted flatwoods salamander. As discussed earlier in sections 2.8 and 3.4, we estimate this species will require at least 101 resilient metapopulations rangewide (25 per RMU) to persist into the future at least 40 years with a reasonable risk of extinction based on a PVA approach. Thus, the wetland restoration and management efforts simulated here are necessary for recovery. However habitat restoration and management alone will be inadequate to recovery the species. Population management, such as captive breeding, translocation and reintroductions are needed as well because the distribution of the species is so fragmented that re-colonization is not possible in many cases. Finally, these habitat and population management efforts should be clustered to have the greatest chance of success to recover the species.

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Appendix 1. List of future conditions elicitation participants

Respondent	Category	Affiliation	Title	Property Affiliation
Anna Farmer	Species expert	Florida Fish and Wildlife Conservation Commission	Florida Research Lead	All Florida
Mike Sisson	Species expert	Florida Fish and Wildlife Conservation Commission	Wildlife Biologist	All Florida
Pierson Hill	Species expert	Florida Fish and Wildlife Conservation Commission	Research Associate	All Florida
John Jensen	Species expert	Georgia Dept. of Natural Resources	Wildlife Biologist	All Georgia
Joe Reinman	Species/land manager	U.S. Fish and Wildlife Service	Wildlife Biologist	St. Marks National Wildlife Refuge
Jonathan Chandler	Species expert	U.S. Fish and Wildlife Service	Biological Science Technician	St. Marks National Wildlife Refuge
John Dunlap	Species/land manager	U.S. Forest Service	Wildlife Biologist	Apalachicola National Forest
Jana Mott	Land manager	U.S. Forest Service/The Nature Conservancy	Restoration Specialist	Apalachicola National Forest/Flint Rock properties
Jaime Barichivich	Species expert	U.S. Geological Survey	Wildlife Biologist	St. Marks National Wildlife Refuge/Flint Rock properties
Dr. Katie O'Donnell	Species expert	U.S. Geological Survey	Wildlife Biologist	St. Marks National Wildlife Refuge/Flint Rock properties
Dr. Kurt Buhlmann	Species expert	University of Georgia, Savannah	Senior Research Associate	All properties

		River Ecology Laboratory		
Marysa Milinichik	Species expert	U.S. Fish and Wildlife Service	Biological Science Technician	St. Marks National Wildlife Refuge/Flint Rock properties
Roy King	Species manager	U.S. Army	Wildlife Biologist	Fort Stewart