Species Status Assessment for the

Reticulated Flatwoods Salamander

(*Ambystoma bishopi*)
Version 1.0

(Credit: Kelly Jones, Virginia Tech)

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EXECUTIVE SUMMARY

This species status assessment (SSA) reports the results of a comprehensive status review for the reticulated flatwoods salamander (*Ambystoma bishopi*), documents the species' historical conditions, and provides estimates of current and future conditions under a range of different scenarios. This species 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 species was split from the threatened frosted flatwood salamander (*A. cingulatum*), listed as endangered, and critical habitat was designated for both species (74 FR 6700).

Reticulated flatwoods salamanders are moderately-sized (mean snout-to-vent length (SVL) = 59 mm, max SVL = 78 mm, mean total length (TL) = 105 mm, max TL = 143 mm), slender salamanders with relatively short, pointed snouts and stout tails (Martof and Gerhardt, 1965; Palis, 1997a; John Palis, Palis Environmental Consulting, 1995 unpublished data; Gorman and Haas, Virginia Tech, 2014 unpublished data). Their heads are small and only about as wide as the neck and shoulder region (Petranka, 1998). They weigh from approximately 1 to 12 g (adult males and adult gravid [containing mature eggs] females), respectively (Palis, 1997a; John Palis, Palis Environmental Consulting 1995 unpublished data; George Brooks and Carola Haas, 2019 unpublished data). 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 adults, and widely scattered and "lichen-like" in recently metamorphosed individuals. 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.

Reticulated flatwoods salamanders are ephemeral wetland-breeding amphibians with complex life cycles (i.e., there is a terrestrial egg stage, an aquatic larval life history stage, as well as a terrestrial metamorphosed juvenile and adult stage). As adults, flatwoods salamanders migrate to ephemeral (seasonally-flooded) wetlands to breed in the fall, where females lay eggs singly or in small clusters on litter, vegetation, or soil, usually in small depressions near the base of plants, in dry areas that will later fill with water provided by winter rainfall (Anderson and Williamson, 1976; Palis, 1995a, 1997b; Gorman et al. 2014). Well-developed embryos hatch into larvae in the winter and metamorphose between March and May after an 11 to 18 week larval period (Palis, 1995a). Juveniles normally disperse from wetlands shortly after metamorphosing, but may stay

near wetlands 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; 1-2 years for females) and most return to their natal wetland to breed during the fall months (Petranka, 1998; Powell et al., 2015, Tom Gorman and Carola Haas, 2014 unpublished data).

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. In the past, this ecosystem was characterized by open pine woodlands maintained by frequent, lightning-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 reticulated 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 reticulated flatwoods salamander, we assessed a range of conditions to allow us to consider the species' resiliency, representation, and redundancy.

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 population 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 34 metapopulations per each of the 3 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.

Representation characterizes a species adaptive potential by assessing geographic, genetic, ecological, and niche variability. The reticulated flatwoods salamander historically occurred within the western Coastal Plain of the Florida panhandle, extreme southwestern Georgia, and extreme southeastern Alabama (Palis and Means, 2005). The species is currently represented in three separate metapopulations capable of at least short term sustainability. Two of these metapopulations are within Eglin Air Force Base (EAFB) and the other at Escribano Point Wildlife Management Area, outside the western boundary of Eglin, along the eastern coast of Pensacola Bay, Florida. One population at Hurlburt field re-discovered in February 2020 after finding no larvae since 2015, and one population re-discovered in 2018 at Yellow River Marsh Preserve State Park, and one population at Garcon River Water Management Area, but has been at least 2 years since the last detection of animals there. One very small, intermittent population exists on Mayhaw Wildlife Management Area in Southwestern Georgia. No current populations are known to remain in Alabama. 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 34 metapopulation targets in each of the RMUs. This would help prevent potentially clumping too many metapopulations into a confined geographic area within the range.

We have assessed the reticulated flatwoods salamander's levels of resiliency, redundancy, and representation currently and up to 80 years into the future by estimating the persistence of each population on currently occupied properties. Rankings are quantitative assessments of the relative condition of the reticulated 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).

Together, Resiliency, Redundancy, and 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 3R's, we characterized both the species' current viability and forecasted its future viability over a range of plausible future scenarios. To this end, we estimated the current condition and persistence of each population using a combination of the best available scientific information and expert elicitation.

The most significant stressors for individuals and populations of the reticulated 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 lightning 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 wetlands), and inadequate funding to address recovery actions. Genetic bottlenecking 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.

The reticulated flatwoods salamander will continue to 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.

We used the best available information to forecast the likely future condition of the reticulated flatwoods salamander. Our goal was to describe the viability of the species in a manner that will address the needs of the species in terms of the 3 R's. We considered a range of potential 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).

Table ES-1. Summary results of the reticulated 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 and quantity; sufficient herbaceous groundcover and sufficient habitats with suitable soils.	Only 3 metapopulations currently known, one on Eglin AFB one west on Escribano Point WMA, one at Yellow River Marsh Preserve State Park; individual wetlands at an unnamed pond on Santa Rosa County land and Garcon Point Water Management Area. One at Mayhaw WMA in Georgia, capable of at least short term sustainability.	May be limited by available sites for future repatriation. Expansion of occupied habitat into suitable but unoccupied habitat one of the primary focal points on DoD and State managed lands. We estimate that 101 resilient metapopulations are required to ensure the species persistence into the future.
Redundancy: number and distribution of populations to withstand catastrophic events	Multiple resilient populations distributed across the historic range of the species.	As above, currently little redundancy. All but 1 breeding wetland occurs in RMU 2. (Eglin Complex) with the last remaining one at Mayhaw WMA in Southwestern Georgia. Extremely vulnerable to catastrophic events and widespread weather conditions	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 34 resilient metapopulations in each RMU are

			required to ensure persistence of the species into the future.
Representation: genetic and ecological diversity to maintain adaptive potential	Maintenance of genetic variability, adaptive potential, and less population isolation	Preliminary genetic work shows indication of some bottlenecking. More in depth genetic work currently underway.	Use of genetic and adaptive trait 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 reticulated flatwoods salamander (*Ambystoma bishopi*) is a moderately-sized salamander that is endemic to longleaf pine (*Pinus palustris*)-dominated flatwoods/savanna communities in southwest Georgia and Florida west of the Apalachicola/Flint River system. It is currently found only in Santa Rosa and Okaloosa counties in Florida and Miller County in Georgia. 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 revision, the reticulated flatwoods salamander (*A. bishopi*) was listed as endangered, and the frosted flatwoods salamander (*A. cingulatum*) retained threatened status.

The Species Status Assessment (SSA) framework (U.S. Fish and Wildlife Service, 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 threatened or endangered species, to support and inform all functions of the Endangered Species Program from Candidate Assessment, to Listing, to Consultations to Recovery.

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), the 2009 decision to split the species into two and designation of critical habitat (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 reticulated flatwoods salamander SSA is intended to provide a review of the best available commercial and scientific 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 its ecosystem for at least 20 years. We chose 20 years because it represents 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).

- 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 wetlands 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 hydrology (to include hydroperiod (duration of surface water in the basin) and recession rate [Chandler et al. 2017], which is the reduction in wetland water level over time, often expressed in centimeters per day and, in this system, is primarily driven by evapotranspiration and groundwater flux,), and 6) burn frequency and season.

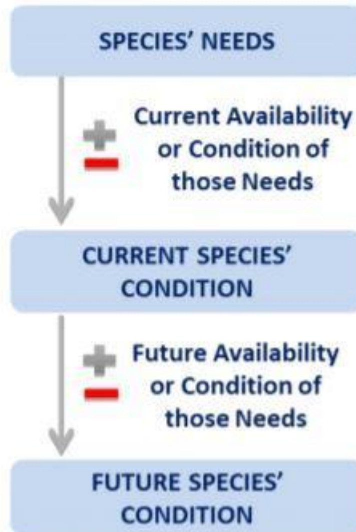

Figure 1.1. Species Status Assessment Framework consisting of three basic stages. From USFWS (2016).

- 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 reticulated 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 reticulated 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 reticulated flatwoods salamander.

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

2.1 Description

Reticulated flatwoods salamanders are moderately-sized (mean snout-to-vent length (SVL) = 59 mm, max SVL = 78 mm, mean total length (TL) = 105 mm, max TL = 143 mm), slender salamanders with relatively short, pointed snouts and stout tails (Martof and Gerhart, 1965; Palis, 1996; John Palis Palis Environmental Consulting 1995, unpublished data; George Brooks and Carola Haas, Virginia Tech, 2019 unpublished data). Their heads are small and only about as wide as the neck and shoulder region (Petranka, 1998). They weigh from 1 to 12 grams (adult males and adult gravid [containing mature eggs] females), respectively (Palis, 1997a; John Palis, Palis Environmental Consulting 1995 unpublished data; George Brooks and Carola Haas, Virginia Tech 2019, unpublished data). 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 reticulated flatwoods salamander is (Integrated Taxonomic Information System, 2018):

Phylum: Chordata
Class: Amphibia
Order: Caudata
Family: Ambystomatidae
Genus: *Ambystoma*
Species: *Ambystoma bishopi*

There are 33 species of *Ambystoma* found 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, and two species, *Ambystoma mavortium* and *A. tigrinum*, are found in all three countries (IUCN, 2018). 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 reticulated flatwoods salamander is a wetland-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 triggered by cold-front associated rains (October to December), where females lay eggs singly or in small clusters on litter, vegetation, or soil in small depressions that later fill with water (Figure 2.1B; Anderson and Williamson, 1976; Palis, 1995a, 1997; Gorman et al., 2014; Brooks et al., 2019a and 2019b). After a period of typically 22-36 days (Anderson and Williamson, 1976) and once inundated, well-developed embryos hatch into larvae (Figure 2.1C) in the winter. Individuals then metamorphose between March and May, typically after an 11 to 18 week larval period (Palis, 1995a). Factors that control larval growth rate and size at metamorphosis are not well known for this species, but for some members of the genus, abundant food supplies coupled with low temperatures and slow drying allow prolonged growth and increased size at metamorphosis (Stewart, 1956, Newman, 1998, Ihli and Beachy, 2016). Juveniles (Figure 2.1D) normally disperse from wetlands shortly after metamorphosing, but may stay near wetlands 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; 1-2 years for females) and return to their natal wetland to breed during the fall months (Petranka, 1998; Powell et al., 2015; Brooks et al., 2019b). Adults are known to regularly live at least 4 years (both in captivity and in the wild), (Palis and Means, 2005; George Brooks, Virginia Tech, 2019 pers. comm.), and Palis et al. (2006) attributed a decline in the number of adults captured in a 4-year drift fence study to attrition without recruitment of juveniles during an extended drought. However, recent evidence suggests that in rare cases individuals can live for as long as 9-12 years in the wild (Kelly Jones, Virginia Tech, 2016 pers. comm.; George Brooks, Virginia Tech, 2019 pers. comm.).

(A)	(B)	(C)	(D)

Figure 2.1. Life stages of the reticulated flatwoods salamander. (A) Adult salamander in wiregrass, Aristida sp. (B) Eggs laid among herbaceous ecotone vegetation. (C) Developing embryo. (D) Aquatic larva. All photographs courtesy of Kelly C. Jones.

2.4 Habitat

Breeding wetlands are located within mesic (moderate moisture) to intermediate-mesic longleaf pine (*Pinus palustris*)-dominated flatwoods/savanna communities where adults and metamorphosed juveniles spend the rest of their life outside of the breeding season. There are some variations in vegetation, geology, and soils among geographic areas within the range of the salamander; however, basic characteristics are similar throughout. Longleaf 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 spring and early summer, these flatwoods historically burned at intervals ranging from one to 4 years (Clewell, 1989). The topography can vary from nearly flat to gently rolling hills (the latter is true especially in the Dougherty Plain of southwest Georgia).

The groundcover of the longleaf pine flatwoods/savanna ecosystem is typically dominated by 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.), pitcher plants (*Sarracenia* spp.), meadow beauties (*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 (Sekerak et al., 1996).

Flatwoods salamanders breed and deposit eggs in wetlands that are, ideally, not yet inundated with water (Anderson and Williamson, 1976; Hill, 2013; Powell et al., 2013; Gorman et al., 2014). Females select areas within breeding wetlands, including the ecotone, that have complex and diverse stands of herbaceous vegetation and concave depressions for egg deposition (Gorman et al., 2009; Jones et al., 2012; Gorman et al., 2014). Such small depressions likely minimize desiccation, freezing, or burning of developing embryos in the otherwise dry wetland (Powell et al., 2013; Gorman et al., 2014). The total area of herbaceous cover suitable for egg-laying and larvae was the best predictor of occupancy by reticulated flatwoods salamanders, and no wetlands on Eglin AFB with less than 0.2 ha of herbaceous breeding habitat were occupied

(Brooks et al., 2019b). As noted, management of breeding wetlands for this species should include a suite of management actions that increase the cover of herbaceous vegetation (Gorman et al., 2014).

When inundated with sufficient precipitation, these wetland basins become acidic (pH 3.4 to 5.6), tannin-stained ephemeral wetlands (swamps or marshes) suitable for continued larval development, that typically range in size from <1 to 10 acres (ac) (0.4 to 4.0 hectares [ha]), but may reach or exceed 37 ac (15 ha) (Palis, 1997; Safer, 2001; Nicholas Caruso and Carola Haas, 2019, Virginia Tech, unpublished data).Wetlands 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 portions of the wetland dominated by herbaceous ground cover provide important habitat for larval development when inundated (Chandler et al., 2015, 2017). Water depth fluctuates greatly, but is usually 0.5 meters (m) or less (Palis, 1997; Bishop, 2005) in areas where larval salamanders are found. Wetlands that fluctuate more slowly (have a lower recession rate) appear to be more likely to support flatwoods salamander reproduction (Chandler et al. 2017; Nick Caruso, Virginia Tech, 2019 unpublished data). Wetlands typically fill in late fall or early winter, and dry in late spring or early summer. Summer thunderstorms may refill some wetlands, but most of these dry again by early fall (See Figure 1 in Chandler et al., 2016).

Suitable wetlands tend to be distributed on the landscape in a clumped fashion. For *A. bishopi* on Eglin AFB 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 (Brooks et al., 2019a and George Brooks Virginia Tech 2019, unpublished data).

Under current conditions, the overstory within breeding wetlands is typically dominated by pond cypress (*Taxodium ascendens* [=*T. distichum* var. *imbricarium*; Lickey and Walker, 2002]), blackgum (*Nyssa sylvatica* var. *biflora*) and slash pine (*Pinus elliotti*), but can also include red maple (*Acer rubrum*), sweetgum (*Liquidambar styraciflua*), sweetbay (*Magnolia virginiana*), loblolly bay (*Gordonia lasianthus*), longleaf pine, and pond pine (*Pinus serotina*). Canopy cover of occupied wetlands 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*). When dry, breeding wetlands 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, as well as litter and duff accumulation, the herbaceous groundcover of breeding sites can vary widely, although larvae and their prey are most often associated with higher amounts of herbaceous cover, (Gorman et al., 2009; Gorman et al., 2013; Chandler et al. 2015; Brooks et al., 2019b) which, in occupied wetlands covers an average of 0.7 ha and was always at least 0.2 ha (Brooks et al., 2019b) and, on average, is greater than (>) 40% coverage of the wetland (Gorman et al., 2009; Gorman et al., 2013). It seems likely that larger amounts of herbaceous cover would support larger populations of salamanders, but that relationship has not been documented. Most, but not all, breeding sites exhibit distinct vegetative zonation, with bands of different herbaceous plant assemblages in shallow vs. deeper portions of the wetland. The groundcover is dominated by graminaceous species, including beakrushes (*Rhynchospora spp.*), sedges (*Carex* spp.), panic grasses (*Panicum* spp.), rosette grasses (*Dicanthelium* spp.), bluestems (*Andropogon* spp.), jointtails (*Coelorachis* spp.), longleaf three-awned grass (*Aristida palustris*), plumegrasses (*Erianthus* spp.), nutrush (*Scleria baldwinii*), and 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.). The basin of breeding sites generally consists of relatively firm mud with little or no peat and little litter accumulation (Sekerak et al., 1996).

Burrows of crayfish (genus *Procambarus*, principally) are a common feature of flatwoods salamander breeding sites, which are typically encircled by a bunchgrass (wiregrass, Curtiss' sandgrass [*Calamovilfa curtissii*] and/or dropseed) dominated graminaceous ecotone. Two wetland basins that have been regularly occupied by breeding reticulated flatwoods salamanders have a burrow density of 0.6 to 2.2 burrows/m^2, a higher density of burrows than documented in crawfish frog habitat (Powell et al., 2015). The density of burrows used by metamorphs and adults in upland habitats is unknown, but their availability is likely to be important. These wetlands often harbor small fishes; the most typical species include pygmy sunfishes (*Elassoma* spp.), Eastern mosquitofish (*Gambusia holbrookii*), American pickerel (*Esox americanus*) and banded sunfish (*Enneacanthus* spp.) (Palis, 1997). Typical amphibian associates of flatwoods salamander larvae include southern leopard frog (*Rana sphenocephala*), ornate chorus frog (*Pseudacris ornata*), southern cricket frog (*Acris gryllus*), and dwarf salamander (*Eurycea quadridigitata*) larvae, as well as larval and adult newts (*Notophthalmus viridescens*) (Palis, 1997; Erwin et al., 2016).

Some of the previous descriptions of flatwoods salamander adult and larval habitats represent high quality conditions. However, populations persist in less than ideal habitat which may differ from what is presented above. Appropriate management will be needed at most of these sites to prevent the populations from disappearing as habitat conditions worsen. For example, fire suppression at many sites has led to greater canopy closure in the overstory of both the flatwoods uplands and ephemeral wetlands (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), allowing extensive accumulation of hardwood litter and duff. This has resulted in a lower cover of herbaceous ground cover that is less diverse. Wetland habitats with less herbaceous cover also had lower diversity and abundance of arthropod prey in the water column (Chandler et al., 2015). Overstocking of woody vegetation in the uplands may also result in reduced hydroperiod (Jones et al., 2018a). In the absence or paucity of lightning-season fire, slash and/or pond pine may become dominant over longleaf pine in the flatwoods uplands. Further, the ecotone between the breeding wetland and associated flatwoods may be obscured or non-existent, replaced with a dense layer of shrubs, such as titi, fetterbush, and dog hobble (*Leucothoe* spp.) due to fire suppression or exclusion (Gorman et al., 2013). Additionally, prescribed burns across the range of the species are conducted most often in winter and early spring when wetlands would typically be flooded and less likely to burn (Bishop and Haas, 2005). Larger, deeper wetlands may have historically provided more suitable habitat but may have been most likely to become overgrown by shrubs as a result of fire suppression (because these large, deep wetlands would only be dry enough to burn throughout their basin during the driest periods when managers are less likely to be able to use prescribed fire). To increase effective burning of flatwoods salamander habitat, land managers should diversify burning strategies (Bishop and Haas, 2005). Other options may include burning uplands during the dormant season (being cautious not to burn when adults may be migrating or active near the surface) and return in the lightning season to burn wetlands when they are dry (Gorman et al., 2009), although exclusive use of this approach may lead to unsatisfactory results in the salamander's upland habitat. Mechanical treatments (including removal of duff layers) 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 that have the physical and biotic characteristics of natural breeding sites may be used by flatwoods salamanders, especially when located near natural breeding wetlands (Anderson and Williamson, 1976; Palis, 1995b; Stevenson, 1999; Tom Gorman and Carola Haas, Virginia Tech, unpubl. data).

2.5 Diet

Because of its complex life cycle, the diet of the reticulated 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 three most common prey in Whiles' et al. (2004) study were isopods, amphipods, and cyclopoid copepods. 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 River. Based on these findings, the Service recognized two distinct species in 2009, reticulated flatwoods salamander (*Ambystoma bishopi*) and frosted flatwoods salamander (*Ambystoma cingulatum*) (74 FR 6700).

Using microsatellite nuclear DNA markers, Wendt (2017) characterized population structure of the reticulated flatwoods salamander at Eglin Air Force Base (EAFB), a locality (an area with multiple known clusters of breeding wetlands) for this species. Wendt’s analysis revealed that the greatest amount of genetic variation at EAFB was among three spatially disconnected regions, even though wetlands < 1 km of each other were also genetically divergent. Wendt’s (2017) analyses suggested that each breeding wetland functions as a semi-independent local population. Estimates of effective population size were fewer than 50 salamanders at most wetlands, and land cover (especially urbanization) was associated with restricted gene flow between populations (Wendt, 2017).

2.7 Ecological Needs

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

1. Individual Resource Needs (Table 2.1)
   a. Eggs/embryos: Flatwoods salamanders breed and deposit eggs in small depressions in wetlands that are not yet inundated with water (Anderson and Williamson, 1976; Hill, 2013; Powell et al., 2013; Gorman et al., 2014). Adults select areas of complex and diverse stands of herbaceous vegetation within breeding wetlands for egg deposition. In this microhabitat, eggs are typically located in small depressions that likely minimize 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 40 ac (15 ha) (Palis, 1997; Safer, 2001). Occurrence of larvae is associated with low conductivity (Jamie Barichivich, USGS pers. comm. 2019). Water depth fluctuates greatly, but is usually 0.5 meters (m) or less (Palis, 1997; Bishop, 2005) 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 0.7 ha (and at least 0.2 ha), often, but not necessarily, covering >40% of the wetland (Gorman et al., 2009, Gorman et al., 2013; Brooks et al. 2019a). A minimum wetland hydroperiod (length of time wetland retains

water) of at least 11-18 weeks is needed to complete metamorphosis (Palis, 1995a). Cool winter temperatures and a slow wetland recession rate may also be important to larval growth and metamorphosis (Newman, 1998, Ihli and Beachy, 2016; Chandler et al., 2017).

c. Juveniles: Juveniles normally disperse from wetlands shortly after metamorphosing, but may stay in or near wetland basins 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; 1-2 years for females). The presence and density of burrows may be important for growth and survival (Powell et al., 2015). Suitable, seasonally-appropriate fire-maintained terrestrial habitat is necessary for dispersal, migration to/from adjacent wetlands, feeding, 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.

Table 2.1. Life history stage and resource needs of the reticulated 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	Exposed soil (not deep duff) in herbaceous ecotone and basin (maintained by frequent fire) between upland and wetland habitats for oviposition and development; seasonally appropriate rain events to expose and then flood oviposition sites and fill wetlands	B	Anderson and Williamson, 1976; Palis 1995a, 1997; Gorman et al., 2014
Larvae	Ephemeral wetlands with herbaceous cover and hydroperiods of at least 11-18 weeks to complete metamorphosis; low recession rates and good larval habitat in deeper sections of wetland	F,S	Palis, 1995a; Gorman et al., 2009; Gorman et al., 2013; Chandler et al., 2017

Juveniles	Fire-maintained upland habitat in longleaf pine-wiregrass ecosystems that is suitable for dispersal, migration to/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 (may need a density of 0.6 to 2.2 burrows/m^2 or more) until they reach sexual maturity (1 year for males; 1-2 years for females)	F,S,D	Petranka, 1998; Powell et al., 2015
Adults	Upland habitat: same as for juveniles; density of burrows necessary for survival is unknown. 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. 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, 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. 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 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, as well as loss and degradation of wetlands, 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, fire-suppressed stands, and pine plantations, leaving many flatwoods salamander populations widely separated from each other by unsuitable habitat.

b. Species-level Redundancy: Processes that increase the number and connectivity of wetlands within populations (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 restoration of degraded wetlands, restoration of diverse hydroperiods by upland stand thinning, construction of new wetlands, acquisition of new habitat, corridor development, and assisted colonization may also increase redundancy of populations (Semlitsch et al., 2017).
c. 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. bishopi*, re-establishment of extirpated populations throughout the species' historical range, particularly in Georgia, Alabama, and at the eastern-most edge of its range along the Apalachicola River in Florida would increase the diversity of habitats and environmental conditions in which this species is found.

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 bishopi* occurred in Mobile, Baldwin, Covington, and Houston counties in Alabama, Escambia, Santa Rosa, Okaloosa, Walton, Holmes, Washington, Bay, Jackson, Calhoun, and Gulf counties in Florida, and Seminole, Decatur, Early, Miller, Baker, Dougherty, and Lee counties in Georgia. Over time and despite recently increased efforts to survey historical locations and find new populations, the combined range of *A. bishopi* 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).

When the 2009 final rule was published (74 FR 6700), there were 20 existing populations of *A. bishopi*. These populations were defined to consist of salamanders that use breeding wetlands within 3.2 km (2 miles [mi]) of each other, barring an impassable barrier such as a perennial stream (64 FR 15691). That definition is not supported by recent information. Ecologically, for other species of *Ambystoma*, the interpond distance used in this legal definition could describe 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. Another member of the same family, *Ambystoma maculatum*, showed significant genetic correlation among wetlands located within 4.8 km of each other (Zamudio and Wieczorek, 2007). The observed dispersal distances of marked individuals 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 wetlands within 2.09 km and 2.51 km of each other were connected, demographically and genetically, respectively. However, for *A. bishopi* at EAFB, Wendt (2017) found that wetlands less than 1 km from each other were genetically divergent and suggested that each breeding wetland functions as a semi-independent local population at this site. Also at EAFB, an indirect measure of connectivity from occupancy models found no evidence of connectivity beyond 1.5 km (the scale at which a metapopulation of flatwoods salamanders could be defined) and an average dispersal distance of only about 230 m (Brooks et al., 2019a). Thus, the number of actual "populations" of *A. bishopi*, as defined by the 2009 final rule, is likely inclusive of several different populations. Therefore, we use a data-based definition of population in this document, but when needing to compare with the previous document for consistency, we use the term "populations recognized in 2009" as defined in the final rule.

Amphibian populations often exist in a metapopulation, and evidence suggests this is likely the case for reticulated flatwoods salamanders. Wetlands occur densely on the landscape in many of the locations where populations of reticulated flatwoods salamanders persist (Figure 2.2). Because populations that have persisted over time have multiple wetlands in close proximity (Brooks et al., 2019a; George Brooks, Nick Caruso, and Carola Haas, Virginia Tech, 2019 unpublished data), we define a resilient population as one that includes at least 3 regularly occupied (larval detections once every 3 years) wetlands within an approximately 500 m radius and connected in a stepping stone fashion to at least 6 suitable wetlands that are within 500 m of each other. A resilient metapopulation would include at least 14 suitable wetlands within a 1.5 km radius of the cluster and connected to 22 or more wetlands through stepping stone arrangement, surrounded by suitable upland habitat. This definition of a metapopulation is based on Brooks et al. (2019b) who indicated that occupied wetlands within 1.5 km of other occupied wetlands were also occupied and genetic data (Wendt 2017) confirmed the small scale of connectivity (<1 km).

Of the 20 populations recognized in 2009, 11 exist on private lands (mainly designated critical habitat areas), and 9 on public lands. Most of the private lands have not been surveyed in recent years. Unfortunately, this has created a gap in the known status of the majority of wetlands located in designated critical habitat on private lands, at least during any recent breeding seasons. Having limited breeding data on private wetlands does not mean those populations have disappeared. However, we present several lines of evidence related to habitat of some private land sites and information on one private land site that has gone locally extinct. Overall, 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. Some former surveys may have been too limited to have a high probability of detection. Previous work suggested that 2 surveys per year would be sufficient (Bishop et al., 2006) but more recent analysis at Eglin AFB found that 4 dipnet surveys per year are required (Brooks et al., 2019a).

Figure 2.2. Outline of flatwoods ponds, roughly delineated from aerial imagery, across a landscape, showing that they vary in size and shape but occur at high density (Data source: Mark Windland, FWC, Image credit Charlie Abeles, Longleaf Alliance).

2.9 Current Range and Distribution

In Alabama, the flatwoods salamander was historically known to occur in four southern counties (Wright, 1935; Mount, 1975; Mount, 1980; Jones et al., 1982). Despite more recent survey effort (Godwin, 1994, 2003), the last observation of the species (of what we now have determined is the reticulated flatwoods salamander) in Alabama was in Houston County in 1981 (Jones et al., 1982). In Georgia, *A. bishopi* was recently found in two wetlands on the Mayhaw Wildlife Management Area (John Jensen, pers. comm., 2015), Miller County. Prior to 2015, this species had not been detected in Georgia since 2001, but in some cases surveys were limited. In Florida, recent surveys have detected *A. bishopi* in Santa Rosa and Okaloosa Counties (Table 2). Within these counties, 16 breeding wetlands on EAFB have had at least one detection from 2010 to 2015, with most being occupied during multiple years within this time frame (Gorman et al., 2013, Haas et al., 2014a). Additionally, larvae were detected at one site on Hurlburt Field and one site on property owned by Santa Rosa County in 2014 (Haas et al., 2014b). Larvae were detected at Garcon Point in 2014 (Pierson Hill, FWC, pers. comm.) and, in 2015, larvae were detected at a breeding wetland at Yellow River Marsh Preserve State Park for the first time since

2006 (K. Enge, FWC, pers. comm., 2015). Lastly, *A. bishopi* was detected at one site on Naval Outlying Landing Field (NOLF) Holley in 2010 and another site in 2011, but no salamanders have been detected since 2011 (Kurt Buhlmann, UGA, pers. comm., 2018). Thus, currently, *A. bishopi* is only known from Santa Rosa and Okaloosa counties in Florida, and Miller County in Georgia (Table 2.2, Figure 2.3).

As of the end of the 2014/15 breeding season, there were six known and currently occupied populations (based on unpublished data from Gorman and Haas, Virginia Tech 2014, K. Enge, FFWC, K. Buhlmann, UGA, and J. Jenson, GADNR). The Mayhaw WMA site, the wetland in which larvae were found in 2001, did not yield any detections, but two other wetlands within the same population did. The two detections in 2015 were from wetlands never previously known to be occupied. These two wetlands are considered separate populations (John Jensen, GADNR, pers. comm. 2017). It is important to note that 11 of the 20 populations (described in 74 FR 6700) are on private land, and nine on public land. However, compared to those included in the 2009 final rule, 3-4 populations (33-45%) on public lands have not had detection in recent years (Table 2.2).

Table 2.2. The number of populations from the 2009 final rule (74 FR 6700) and the status of those populations in 2010 and 2015. (Based on unpublished data from Gorman and Haas, Virginia Tech, K. Enge, FFWC, K. Buhlmann, UGA, and J. Jenson, GADNR).

Property	Populations according to 2009 listing	Occupied in 2010	Occupied in 2015
Eglin AFB/Hurlburt Field	3	2	2
NOLF Holley	1	1	0
Pine Log State Forest	1	0	0
Mayhaw Wildlife Management Area	1	0	2
Northwest FL Water Man. Dist. (NWFLWMD) and Blackwater River State Forest	1	0	0
NWFLWMD and Yellow River Marsh Preserve SP (Garcon Point)	1	1	1
Santa Rosa County	1	1	1
Private property#	11	0	0
Total	20	5	6

 # Many private land populations have not been effectively sampled between 2010 and 2015 to know if they are still occupied

From October 2015 to April 2016, migrating adult and subadult *A. bishopi* were captured at four drift fence locations at Escribano Point Wildlife Management Area (WMA) in Santa Rosa County, FL. Additionally, larvae were dipnetted from four wetlands that were adjacent to two of the drift fence arrays (Surdick, 2016). These captures represent the first detections of this species at this location. As of 2018, six new breeding wetlands have been found, bringing the current total of number of breeding sites at Escribano WMA to 10 (Pierson Hill, FWC, 2017 pers. comm.).

Maps informing range and locality information were developed by selecting all ecoregions that included historic records derived from a museum database query (Figure 2.2 Jamie Barichivich, USGS, pers. comm., 2019). The L4 ecoregions were divided into smaller unit by rivers known to be major faunal breaks. The boundaries of the ecoregions were then dissolved into appropriate polygons into the proposed Recovery Management Units (RMUs). 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.

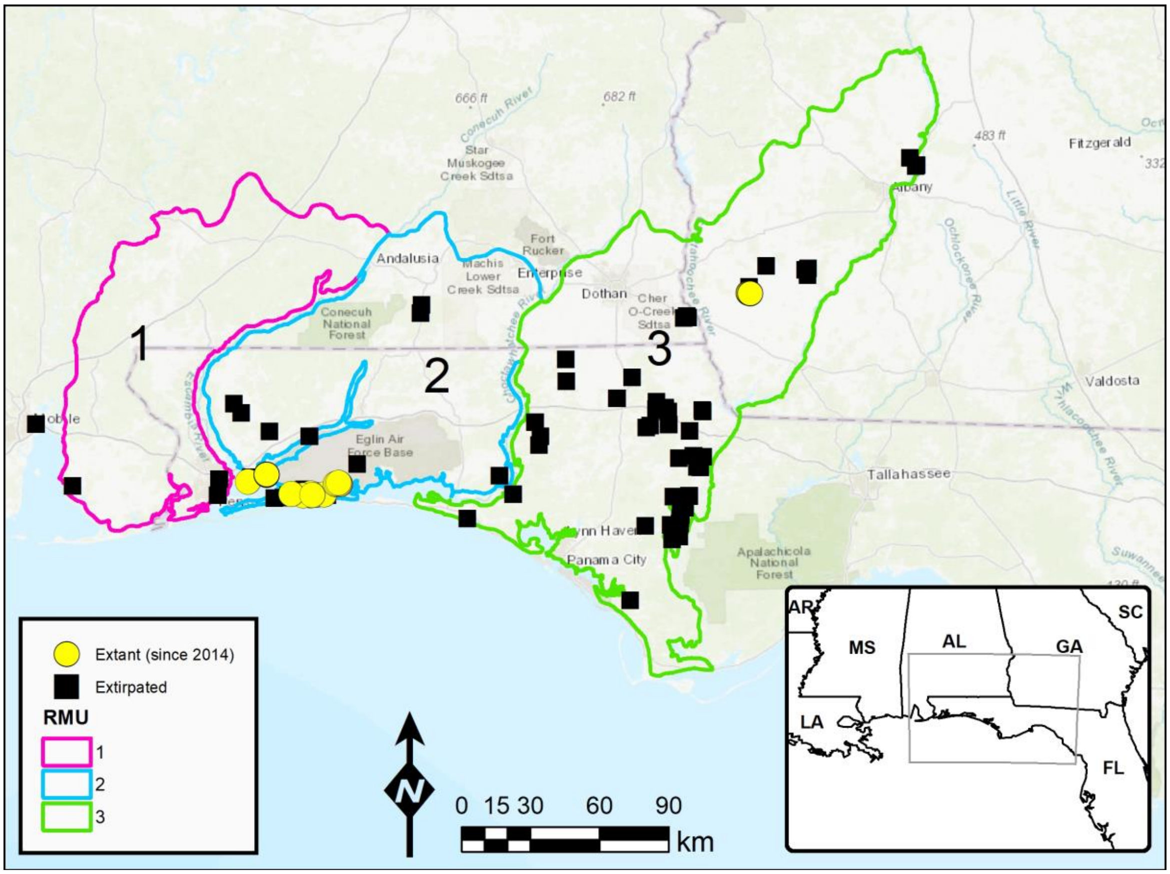

Figure 2.3. Historic and extant locations of reticulated salamanders within the current RMU context.

All but one currently extant metapopulation occurs in RMU 2, and one remains in RMU 3. Our goal is to establish or re-establish roughly 34 metapopulations in each of the 3 RMUs. The extirpated location to the west and outside the boundary of RMU 1 is a very old and possibly misidentified occurrence (Jamie Barichivich, USGS, pers. comm., 2020).

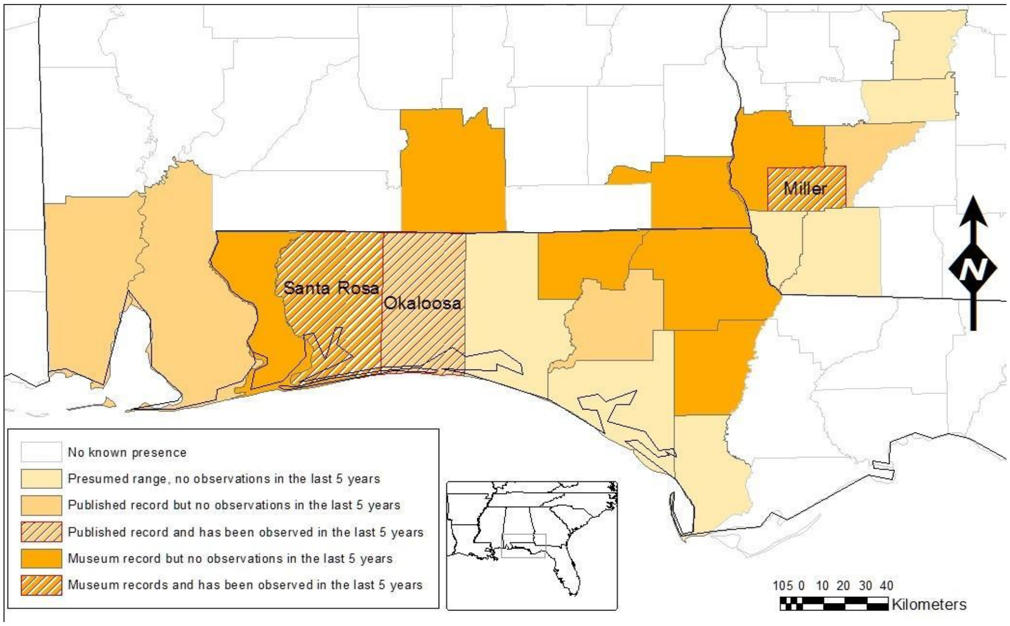

Figure 2.4 Current geographic range of the reticulated flatwoods salamander (J. Barichivich, USGS 2018, pers. comm.).

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 1,803 ha, partitioned among 16 critical habitat units, designated for the reticulated flatwoods salamander, with 18.8% of this habitat located on public lands (Susan Walls USGS et al., unpub. data 2018). An analysis of potentially suitable habitat outside of designated CHUs has not yet been performed for this species. Most of the presumed extirpated populations were located on private lands, but several well-sampled wetlands on protected lands have had no larval detections in over 10 years, after several prolonged droughts (Figure 2.4).

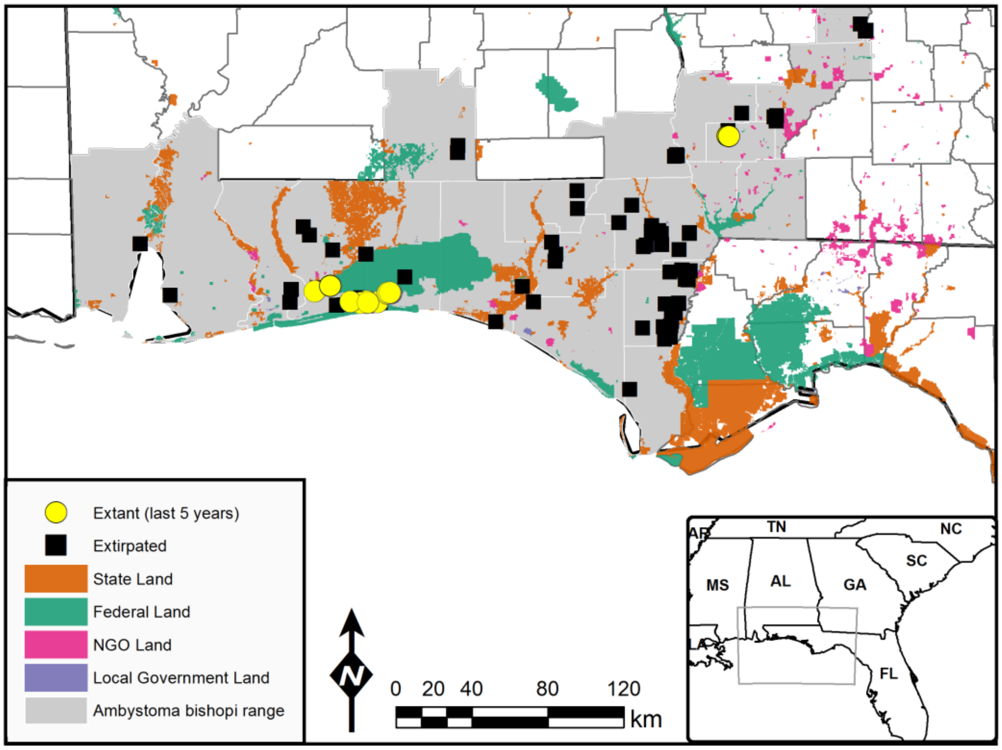

Figure 2.5. Land ownership and historical localities of the reticulated flatwoods salamander (K. O'Donnell, unpubl. Data).

CHAPTER 3 – CURRENT SPECIES CONDITION

3.1 Resiliency

3.1.1 Resiliency Metrics

Resiliency is a reflection of a population's health and its ability to withstand stochastic events (e.g., drought, storms, 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, survival, and reproductive output, 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 flatwoods salamanders, data are not available on demography; thus, alternative measures of resiliency must be used.

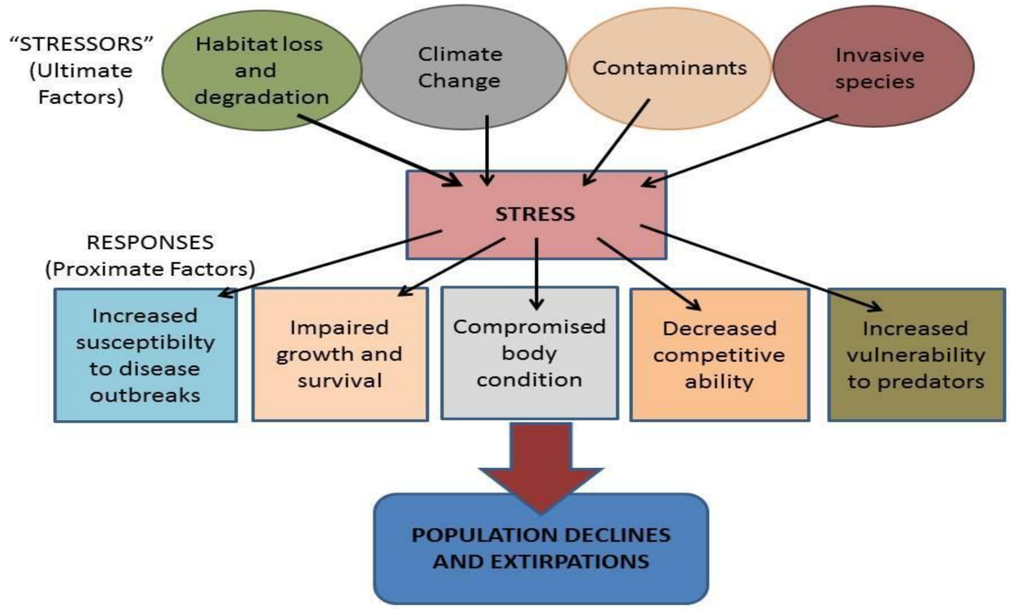

Figure 3.1. Conceptual model of the mechanisms by which key stressors decrease population resiliency, thus leading to population declines. Stressors induce physiological stress which, in turn, suppresses immune responses of members of a population. Responses to such immunosuppression – either individually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

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 reticulated flatwoods salamander based on measures of body size or body condition and effective population size when that information was available and 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). High-quality habitat patches could act as refugia where a species may withstand catastrophes, thus allowing recolonization and re-establishment of redundant populations after a catastrophic event (Earl et al., 2017). Habitat quality may also serve as a proxy for genetic variability – a correlate of representation – as genetically variable populations generally have different environmental conditions or phenotypes (Earl et al., 2017).

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. bishopi* on eight public properties (EAFB, Escribano Point WMA, Mayhaw WMA [Georgia], Garcon Point WMA, Yellow River Marsh State Park, Hurlburt Field, Holley Outlying Field, and Jones Center/Ichauway Plantation). 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 the 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 summarized this information as participants' overall assessment of resiliency of flatwoods salamander habitat on the six properties for which we received results (Table 3.1). We received a total of eight responses for six properties (three for EAFB and one each for the other five properties). We did not receive responses from managers at NOLF Holley and Jones Center/Ichauway Plantation and those properties are not included further. Data were summarized as percentages because individual participants responded only to the wetlands familiar to them and varied by respondent. Overall, 17 breeding wetlands were assessed as highly resilient (either a 4 or 5) and 9 wetlands were moderately resilient. The 4 remaining wetlands 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 wetlands 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. EAFB=Eglin Air Force Base; EPWMA=Escribano Point WMA; MWMA=Mayhaw WMA (Georgia); GPWMA=Garcon Point WMA; YRMSP=Yellow River Marsh State Park; HF=Hurlburt Field.

Property	**Extremely Low**		**Low**		**Moderate**		**High**		**Extremely High**		**Total Ponds**
	Mean	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	**Mean**	**SD**	
EAFB	8	8.3	12	11.5	40	9.6	81	35.3	0	0.0	14
EPWMA	0	0.0	0	0.0	36	36.4	9	9.1	50	50.0	10
MWMA	0	0.0	0	0.0	0	0.0	100	0.0	0	0.0	2
GPWMA	100	N/A	0	N/A	0	N/A	0	N/A	0	N/A	1
YRMSP	0	N/A	0	N/A	100	N/A	0	N/A	0	N/A	1
HF	0	N/A	100	N/A	0	N/A	0	N/A	0	N/A	1
Total											***29***

3.2 Representation

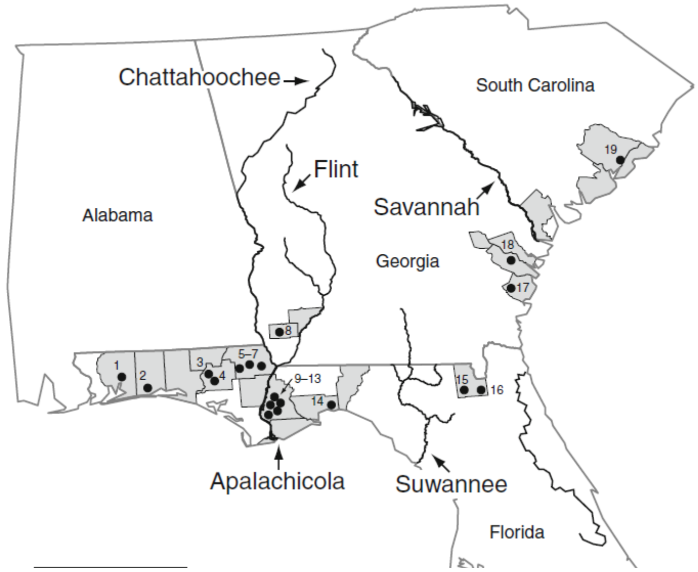

Figure 3.2. Localities (indicated by numbers) at which flatwoods salamanders were collected for molecular and morphological analyses by Pauly et al. (2007, 2012).

Pauly et al. (2007) found that some geographically distant populations of the reticulated flatwoods salamander, located on the same side of the Apalachicola River in Georgia and Florida (Figure 3.2) are phylogenetically very closely related and often share haplotypes. For example, the same haplotype occurs in Populations 1, 5, and 8 (Figure 3.2), which traverses a distance of over 200 km and spans the entirety of the extant distribution of *A. bishopi* (Pauly et al., 2007). Thus, based on Pauly et al.'s (2007) analyses, we consider that there is only one representation unit (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).

The lack of replicate representation units in this species increases its 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. bishopi* to adapt to changing environmental conditions over time is compromised. Re-establishment of extirpated populations throughout the species' historical range, particularly in Georgia, Alabama, and at the eastern-most edge of its range along the Apalachicola River in Florida, would increase the diversity of habitats and environmental conditions in which the reticulated flatwoods salamander is found. Our RMU boundaries (Figure 2.3) reflect this approach.

3.3 Redundancy

Our conclusions about redundancy in *A. bishopi* are based on Semlitsch et al.'s (2017) evaluation of the number of active breeding wetlands (for both species of flatwoods salamanders, combined) and the mean distance of each wetland to the next nearest known breeding wetland.

Semlitsch et al. (2017) compiled locality information across the combined former historical ranges 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. 3.2A-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).

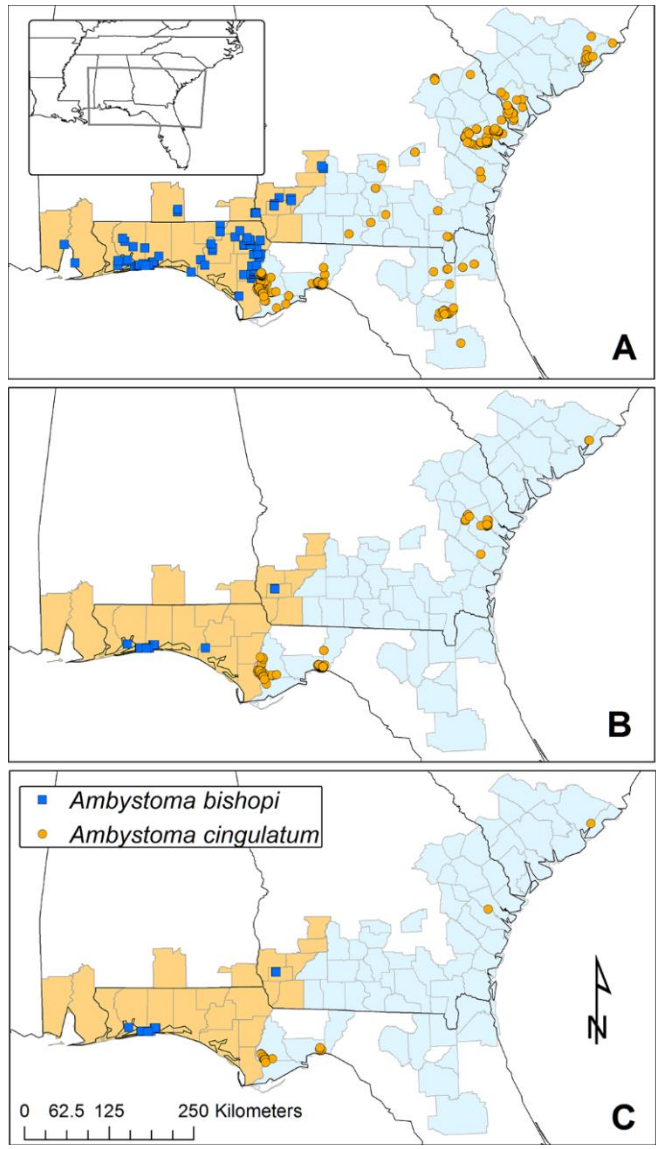

Figure 3.3. Known localities of flatwoods salamanders (Frosted Flatwoods Salamander, Ambystoma cingulatum, and Reticulated Flatwoods Salamander, Ambystoma bishopi) over three time periods. A) All known records, B) 2000-2009 (post-listing), and C) 2010 to 2015 (post-taxonomic split). Orange circles =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

Specific environmental conditions during the breeding season are required for successful recruitment in flatwoods salamander populations; drought years result in catastrophic reproductive failure. A population viability analysis (PVA) conducted for the species revealed a high risk of stochastic extirpation of isolated populations, suggesting that multiple populations distributed across the species range will be necessary to achieve long-term viability of the species (George Brooks, Virginia Tech, 2019, unpublished data).

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 (i.e. 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, 0.000000000000000000000000026697 in 50 years, and 0.0026818 in 50 years.

- For each scenario, we calculated the number of metapopulations required to stay below the desired extinction risk. For example, if the tolerable risk level is 0.0002479 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 34 metapopulations (rounding off remainders of the 3 RMU numbers) in each of the 3 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 compares the potential habitat from the (U.S. Geological Survey - Gap Analysis Project, 2017, Reticulated Flatwoods Salamander *Ambystoma bishopi*) to the RMU boundaries and demonstrates there is enough habitat to support our targets (Table 3.2).

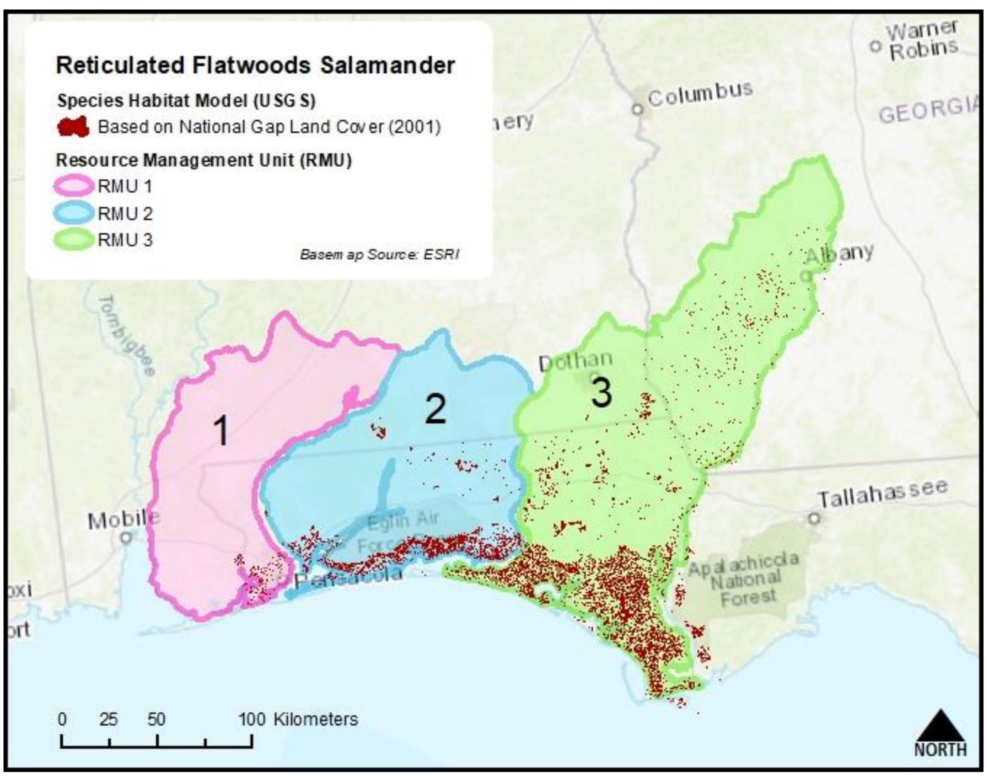

Figure 3.4 potential suitable habitat within each RMU boundary

These data to develop the habitat model were taken from the USGS National Gap Land Cover map (2001). 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 USGS model.

USGS Habitat Data	Area (acres)	Area (sq. m)	Theoretical # of Metapopulations Supported
RFSA Range (USGS)	703,000	2,845,257,300	3,623
RMU 1 (data only for portion, ~10%)	20,674	83,664,900	107
RMU 2 (~90% rmu/data overlap)	167,519	677,927,700	863
RMU 3 (~90% rmu/data overlap)	469,678	1,900,720,800	2,420

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

Our minimum target of 101 metapopulations, when distributed over the 3 RMU's is clearly less than what the potential habitat suggests. However, not all of the dark red shaded habitat may be available for recovery purposes because some portions are privately owned, the habitat has been significantly altered/drained, or other factors. The maps above represents the suitable habitat to support enough resilient metapopulations to be potentially delisted.

3.5 Existing Regulatory Mechanisms

There are no existing regulatory mechanisms for the protection of the upland habitats where reticulated flatwoods salamanders spend most of their lives. Section 404 of the Clean Water Act (CWA) is the primary Federal law that has the potential to provide some protection for the wetland breeding sites of the reticulated flatwoods salamander. However, due to recent case law (Solid Waste Agency of Northern Cook County (SWANCC) v. U.S. Army Corps of Engineers 2001; Rapanos v. U.S. 2006), 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. At the State and local levels, regulatory mechanisms are limited. Although not listed as threatened or endangered in Alabama, the reticulated flatwoods salamander is listed among those nongame species for which it is ‘‘unlawful to take, capture, kill, or attempt to take, capture or kill; possess, sell, trade for anything of monetary value, or offer to sell or trade for anything of monetary value’’.

The 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. In April 2014, the U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers, in response to the SWANCC and Rapanos decisions (Rapanos v United States, 547 U.S. 716

2006), 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 wetlands 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, mechanisms in place to adequately protect ephemeral wetlands, like those necessary for successful flatwoods salamander breeding. This includes the breeding wetlands themselves as well as the surrounding upland habitat. The exceptions to this are the federally designated critical habitat units (CHUs) for *A. bishopi* but this only applies to federal actions and actions with a federal nexus, not to state or private sector actions.

3.6 Current Conservation Measures

Both species of flatwoods salamanders (combined) have experienced an 87% 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 habitat restoration, likely including creation of wetlands, captive breeding, reintroductions and/or translocations to suitable habitat. Additionally, in the light of the value of remnant high-quality habitat patches that allow the species to persist while additional habitat responds to restoration actions, protection from feral swine is critically important through hog exclusion barriers and hog removal (Jones et al., 2018b). In 2014, a structured decision making 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 within its historic range, 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. bishopi* has not been established yet (the only existing captive population for this species is being maintained at the San Antonio Zoo). Moreover, no progress has been made in getting reticulated 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 that was specifically focused on 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., 2017). There was consensus among the workshop participants about how to address this decision problem. The group acknowledged, however, that habitat restoration at SMNWR would likely take several years, and the group decided that waiting to begin reintroductions of frosted flatwoods salamanders into restored habitat would delay conservation efforts for too long, and that actions were needed to maintain current populations on the landscape until restoration could be completed (O'Donnell et al., 2017).

To address this need, the group decided to launch a head-start effort of larval *A. bishopi* at EAFB, along with one with *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 with *A. cingulatum* on that property as well. This *in situ* approach has successfully been employed for the conservation and recovery of at least one other federally listed, endangered amphibian, *Rana sevosa* (USFWS, 2014). The objectives of this ongoing effort are to rear larvae through to metamorphosis, releasing most at their natal wetlands while placing some in captive facilities for future captive breeding. Because low survival of eggs and larvae is normal for ephemeral pond breeding amphibians including other species of *Ambystoma*, (Shoop, 1974; Semlitsch et al., 2014; Anderson et al., 2015), in instances where populations may contain unique genetic variation, bringing some of those individuals into captivity for a captive assurance colony, or headstarting individuals to ensure some metamorphosis even in a drought year could be valuable. Moreover, at low rearing densities in a head-start effort, individuals can metamorphose at a larger body size. Large size at metamorphosis is correlated with increased survival in *A. bishopi* (George Brooks, Virginia Tech, pers. comm. 2019) and has been demonstrated empirically to be related to correlates of 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). However, long-term studies of the effects of captive rearing and stocking of salmonids show that there are serious risks. If adaptation to captivity occurs, head-started or captive-bred individuals may not be well-suited for survival in the wild. In steelhead trout, augmenting a wild population with captive-reared fish reduced population fitness by 8% (Araki et al., 2009). If the genetic diversity of captive-reared or head-started stock is not as high as the wild population (and it is almost impossible to represent a large number of parents in captivity), the effective population size (N_e) of augmented populations can suffer. In steelhead trout, an augmentation program resulted in almost doubling of the census population size but a drop of almost 60% of N_e, a result that could have severely negative effects (Christie et al., 2012).

Thus far, this effort on Eglin has resulted in the production and release of over 800 metamorphs and/or late stage larvae, representing from 54% to 91% of the initial number of larvae/embryos head-started each season (Table 3.3). Of those individuals released, from 5% to 15% have been recaptured at their release site during the same spring season (Table 3.3). However, to date only 4 individuals of the 585 released over a year ago at drift-fenced ponds (0.68%) have been documented returning as adults to the breeding wetlands (Carola Haas, Virginia Tech 2019 unpublished data). The Longleaf Alliance and FWC partnered to set up a REPI funded head starting effort on EPWMA in 2018, with heavy guidance from Virginia Tech and Eglin. In their first year, they released 247 late stage larvae or metamorphs raised from 6 breeding wetlands. A small drift fencing study has been set up to track movement and potential success of released animals in the spring, however it is more likely for success to be determined through ongoing genetic analysis. Currently, there are no plans to heavily increase the head starting effort in the short term until more is known about the success or potential for use in repatriation. However, the LLA field teams plan to greatly expand their egg search and occupancy surveys this year and will adjust their head starting effort accordingly. The EPWMA animals will primarily be used to attempt to bolster existing breeding populations, act as a rescue for stranded individuals, and provide samples for much needed genetic research.

By releasing metamorphosed salamanders at their site of origin, the near-term objective of this effort was 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. There may be a benefit to shifting priorities to salvaging later stage larvae during drought years, and to planning releases for translocations. The definition of a population used in this document based on clusters of wetlands does not mean that artificial movement of animals between wetlands within a population is risk-free. Given observation of genetic structure between wetlands separated by less than 500 m (Wendt, 2017), and the high potential for microgeographic adaptation (Friedenburg and Skelly, 2004; Orizaola and Laurila, 2009; Richardson and Urban, 2013; Richter-Boix et al., 2013), genetic and environmental considerations should be addressed before translocation. Adaptive traits (such as response to temperature or recession rate) may be more useful in identifying source populations than neutral genetic variation or geographic proximity (Mills, 2017). All translocation attempts, even those within a population, must be considered on a case by case basis.

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. It is also not known whether such an increase in abundance would come at a cost of reduced fitness or reduced effective population size as has been documented in stocked fish populations (Araki et al., 2009; Araki and Schmid, 2010; Christie et al., 2012; Willoughby and Christie, 2019), so habitat restoration or creation to increase the size and number of naturally recruited metamorphs, to increase survival of adults in the wild, and to increase connectivity, is clearly an urgent priority.

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 wetland 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 3.3. Results of head-start effort for larval reticulated flatwoods salamanders at Eglin Air Force Base. Animals were reared to release either as metamorphs or late-stage larvae and were released at the edge of the wetland. Survival to release was much lower in 2016-2017 because of the high proportion of animals salvaged from drying wetlands. Values in parentheses are the number of animals in that column as a percentage of the previous column (total input or total released). Snout-Vent Length (SVL) in mm of reticulated flatwoods salamander metamorphs released from cattle tanks at Eglin Air Force Base in the last four seasons reported as mean (standard deviation). Despite reducing average stocking density from 15 to 12 larvae per tank in last two years, there are challenges in consistently rearing large metamorphs.

Season	**Tanks**	**Wetlands**	**Larvae /Eggs Input**	**Larvae/ Metamorphs Released near Pond Edge**	**SVL of metamorphs at release (mm; SD in parentheses)**	**Released Metamorphs Encountered the Same Spring at Drift Fence**
2015-16	6	2	67	47 (70%)	40.5 (2.5)	7 (15%)
2016-17	16	2	350	189 (54%)	33.8 (3.3)	23 (12%)
2017-18	14	2	168	135 (80%)	36.2 (3.3)	7 (5%) limited effort

2018-19	29	4	350	320 (91%)	35.9 (2.4)	15 (5%) limited effort

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 [as legally defined], along with a loss of individual breeding wetlands 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 reticulated flatwoods salamander can be summarized as low to moderate. Only one representation unit exists for this species. The lack of replicate representation units for *A. bishopi* 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. bishopi* to adapt to changing environmental conditions over time is compromised. The species exists primarily in two areas – EAFB and EPWMA. Both locations have redundant populations – EAFB has approximately 14 extant breeding sites and EPWMA has 10 which, on average, are of moderate resiliency and these two areas contain the only populations with high resiliency.

The reticulated flatwoods salamander currently lacks adequate redundancy across its extant range. This species exists only as isolated metapopulations in a few locations (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, important considerations for flatwoods salamanders include (1) a spatial configuration of nearby breeding wetlands, within the range of dispersal, that enhances metapopulation sustainability and (2) an adequate quantity and quality of terrestrial upland habitat surrounding breeding wetlands to facilitate survival, growth, fecundity, and dispersal (Semlitsch et al., 2017).

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) 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 or fall flooding just prior to the egg-laying 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. It is noteworthy that not all of the above mentioned acreage is suitable for flatwoods salamanders.

Many ecologists consider fire suppression to be the primary reason for the degradation of remaining longleaf pine forests. The disruption of the natural fire cycle 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). Ponds surrounded by pine plantations and protected from the natural fire regime may become unsuitable flatwoods salamander breeding sites due to canopy closure and the resultant reduction in herbaceous vegetation needed for egg deposition and larval development sites (Palis, 1993; Gorman et al., 2014). In addition, lack of fire within the wetland during periods of dry-down may result in chemical and physical (vegetative) changes that are unsuitable for the salamander (Bishop and Haas, 2005; Gorman et al., 2013). Large scale prescribed fire is often accomplished in the dormant season, and can have negative effects on salamander habitat (Bishop and Haas, 2005). However, these burns are important for reducing woody fuels and decreasing wildfire danger, but more effort should be placed on

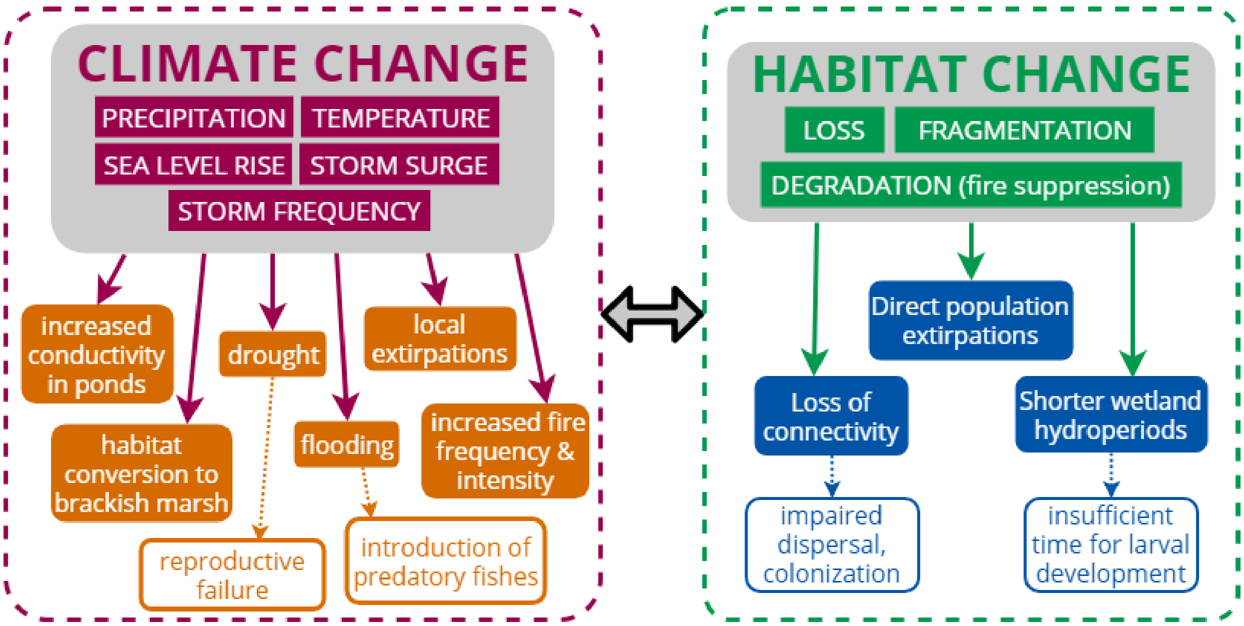

Figure 4.1. Two of the principal stressors, changes in climate and habitat that impact populations of the reticulated flatwoods salamander, along with their ecological and demographic consequences. Dashed lines represent examples of indirect effects from key consequences. The double arrow between the 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 (Walls et al., 2019).

burning the sites when they are dry, avoiding burning when salamanders may be migrating to and from the wetland, and follow-up burns should be used to ensure wetlands benefit from fire. 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 pine plantations. Most flatwoods salamander populations are widely separated from each other by unsuitable habitat.

Road construction in the last two decades destroyed an historic breeding wetland 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 cause the death of flatwoods salamanders during migrations across roads (Means, 1996). Proposed state and local road construction is a recurring threat in the remaining flatwoods salamander habitats on EAFB and Hurlburt Field. Most of these roads are not on Eglin AFB or proposed by the USAF. Roads near base boundaries could cause disruptions to groundwater and sheet flow, and have serious direct and indirect impacts on the breeding wetlands.

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). According to Enge et al. (2014), commercial forestry using silvicultural Best Management Practices (Florida Forest Service, 2012) 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). Urbanization and agriculture resulted in the destruction of flatwoods salamander localities in Mobile and Baldwin counties, Alabama; Jackson and Washington counties, Florida; and Berrien, Chatham, Early, and Effingham counties, Georgia. 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 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 wetlands, 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 wetlands 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. 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, wetland alteration and additional introduction of predatory fish, fire suppression leading to altered forest habitat and crayfish harvesting comprise the most serious threats to *A. bishopi* populations (Palis and Hammerson, 2008).

4.2 Changes in Land Use in Designated Critical Habitat Units

In 2009, the Service designated 16 critical habitat units (CHUs) for the reticulated flatwoods salamander (74 FR 6700), omitting key military lands: EAFB, Hurlburt Field, and NOLF Holley in Florida. These Department of Defense installations each have an Integrated Natural Resource 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 except two CHUs (across both species) were known to be occupied at the time of species' listing in 1999 (64 FR 15691) but all were occupied when critical habitat was designated in 2009.

Means (2013) visited most of the CHUs for this species, including those on private lands, but the year was dry and only limited dipnet sampling could be conducted. Hence, the current status of most of the privately-owned sites within designated critical habitat remains unknown. Means' visits to these CHUs were unable to yield any data on occupancy (due to the time of year she was able to visit the sites outside of the majority of the breeding season) but did provide insight into habitat quality. Many sites were no longer suitable and at least one private land population has had no detections since the early 2000s. Of the 16 critical habitat units, only five appeared to have suitable habitat characteristics for successful breeding by *A. bishop*i following on-site evaluations (Means, 2013) and many were comprised of a single breeding wetland (74 FR 6700). Additionally, a well-managed breeding wetland on private land in Georgia has been sampled several times since 2010 and appears to have experienced a localized extinction of *A. bishopi* (John Jensen, GADNR, pers. comm. 2017).

To quantify the current suitability of CHUs for supporting populations of the reticulated 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). However, several currently and historically occupied wetlands are much bigger than 4.0 ha. In fact, there is a possibility that the larger wetlands were more likely to become overgrown by shrubs during decades of fire suppression, but these larger wetlands (if restored to suitable habitat condition) may have high value for flatwoods salamanders. Earlier work has shown that larger wetlands tend to have more suitable hydrology (Chandler et al., 2017, Jones et al., 2018a). 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. bishopi* are small, ranging from 23.1 ha (RFS-3B) to 354.8 ha (RFS 9B), with a median value of nearly 66 ha. On average, 54% of the designated critical habitats are currently comprised of vegetation types (agriculture/disturbed and plantation habitat) that are not suitable for *A. bishopi* (Table 4.1; Jamie Barichivich, USGS, pers. comm 2017.). These habitat categories include cultivated cropland, developed lands, managed plantations, harvested forests, and other types of land use that is consistent with anthropogenic disturbance. Three CHUs (RFS-2B, 3A and 7A) each have only one wetland of the size (< 4 ha) used for breeding by this species. On average, wetlands are 502 m (using NWI) to 688 m (using NHD) apart, which exceeds the average dispersal distance of *A. bishopi* (230 m; Brooks et al. 2019a) and the maximum dispersal distance (380 m; Scott et al., 2013) that has been reported for eight other species of *Ambystoma* (Table 4.1). Occupancy has not been confirmed in any of the 16 CHUs since 2010, with the year of last observation in the 1990s for most CHUs. This does not reflect existing breeding sites at EAFB and newly discovered sites not encompassed by the CHUs (see "Current Range and Distribution"). Because NWI datasets intentionally exclude small wetlands, direct comparisons are challenging, but wetlands on Eglin (directly observed) are less than 200 m apart on average (Nick Caruso and Carola Haas, Virginia Tech, pers. comm. 2019).

Table 4.1. Summary of key features of designated critical habitat units (CHUs) for the reticulated flatwoods salamander. NHD=National Hydrology Dataset; NWI=National Wetlands Inventory. Susan Walls et al., unpub. data 2017.

CHU	CHU area (ha)	% CHU comprised of agriculture/disturbed and plantation habitat types	Mean distance (m) to nearest neighbor pond (NHD)	Mean distance (m) to nearest neighbor pond (NWI)	Last year confirmed occupancy	Year last surveyed
RFS-1	278.0	0.0	677.4	800.5	2010	2013
RFS-2A	65.6	78.2	170.6	196.0	1993	1993
RFS-2B	65.6	75.6	1644.4	950.1	1993	2016[1]
RFS-3A	60.06	28.0	737.1	565.3	1998	1998

RFS-3B	23.1	59.4	251.3	281.0	1998	2013
RFS-6A	86.2	43.7	315.7	451.2	1993	2013
RFS-6B	65.6	35.7	717.3	318.2	2005	2016[2]
RFS-7A	65.6	65.9	390.4	536.9	1993	2013
RFS-7B	66.6	86.1	820.0	446.7	1993	2013
RFS-8A	44.6	16.2	306.3	835.7	1993	2013
RFS-8B	145.0	81.8	175.0	416.7	1993	2013
RFS-8C	98.8	69.7	691.1	302.0	1993	1993
RFS-9A	65.6	61.7	541.9	417.2	1999	1999[3]
RFS-9B	354.8	71.4	965.1	510.0	1991	2013
RFS-10A	65.6	36.5	561.2	684.6	1998	2016
RFS-10B	251.6	53.5	2047.5	318.5	1999	2016
Median (25^{th}, 75^{th} percentiles)	65.92 (65.610, 133.493)	–	–	–		
Mean ± 1 se		53.96 ± 6.262	688.27 ±128.98	501.91 ± 54.39		

[1]Surveyed by FWC 2 – 3 times per year since 2008
[2] Surveyed annually by FWC since 2002
[3] No access to property since this date

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, the small size of the CHU's for *A. bishopi*, extent of agriculture/disturbed and plantation habitat types (outside of EAFB) within each unit, and the distance between neighboring wetlands may affect abundance, dispersal, survival and genetic variation in this species, all of which are measures of population resilience.

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). More recent work has shown that climate patterns have shifted since the late 1990s to create the shortest wetland hydroperiods on the Florida Panhandle than at any time since the beginning of climate records in 1896 (Chandler et al. 2016). 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 wetland-breeding amphibians (Walls et al., 2013). Flooding, such as that which occurs during increasingly frequent extreme precipitation events, can wash fish predators into breeding wetlands, especially where sheet flow is aggravated by roads and ditching and where flood events are aggravated by impervious surfaces and stream modifications (Snodgrass et al., 1996; Baber et al., 2002; Rahel, 2007). In addition, 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, as can saltwater intrusion through groundwater (Walls et al., 2013). Moreover, phenological shifts in the timing of key climatic events (e.g., wetland-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, in addition to increased temperatures (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 reticulated flatwoods salamanders by altering the timing of fall and winter rains, as well as creating drier winters than historically would have occurred (Chandler, 2015). We have already seen shortened hydroperiods in the last two decades compared to the previous 100 years (Chandler et al., 2016).

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 wetland 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 wetlands, forcing females to oviposit along the outer margins of the wetland basin, which may not be inundated later in the season (Walls et al., 2013). Factors that influence the timing and amount of precipitation during rainfall events, along with duration and recession of wetland hydroperiods and rates of wetland recession (or drying), are likely ***the most important*** constraints on the reproductive success and persistence of flatwoods salamanders (Figure 4.2; Blaustein et al., 2010; Chandler et al. 2016, 2017). 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). 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). Other research on the influence of forest canopy on survival of juvenile and adult ambystomatids found that higher temperatures were related to increased desiccation and mortality, with smaller individuals experiencing more severe consequences of increased temperature (Rothermel and Luhring, 2005; Rothermel and Semlitsch, 2006). These results can help make predictions about possible responses of the reticulated flatwoods salamander to similar conditions, even though these experiments were not conducted with this species.

4.3.2 Sea Level Rise

We examined the potential impact of sea level rise on coastal freshwater wetlands that are breeding sites for *A. bishopi*. We focused on two designated critical habitat units – RFS-1 on Garcon Point and RFS-3A in Navarre, FL. Garcon Point is a peninsula surrounded by Escambia Bay to the west and East Bay to the east. Navarre is located on the Fairpoint Peninsula which is bounded geographically by the East Bay to the north and Santa Rosa Sound to the south.

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 were vulnerable to sea level rise and encroaching marsh. Despite their locations on peninsulas that are bounded by water, neither of these critical habitat units are predicted to be inundated by sea level rise at the 6 ft. level (Figure 4.3). However, at this level marsh habitat (in pink) will encroach upon the southeastern corner of RFS-1 (Figure 4.3).

According to estimated storm surge effects gathered by FEMA, in 2004 Hurricane Ivan storm surge would have potentially inundated approximately 80% of the known occupied wetlands and a large majority of potential/unoccupied wetlands on Escribano Point WMA. Hurricane Ivan made landfall as a category 3 hurricane in Gulf Shores, AL, 45 miles West of Escribano Point. Storm surge effects were estimated to be at 10-16ft above sea level in some places, approximately 11-12 feet throughout most of the coast of Escribano. Several coastal occupied wetlands are within 60m of the shore, one within 15m. At least two of these occupied wetlands have leftover hurricane/storm surge debris throughout the basins. Not much is known about the frequency of storm surge on EPWMA, but it is documented as a potential threat of inundation of the majority of known occupied sites on EPWMA.

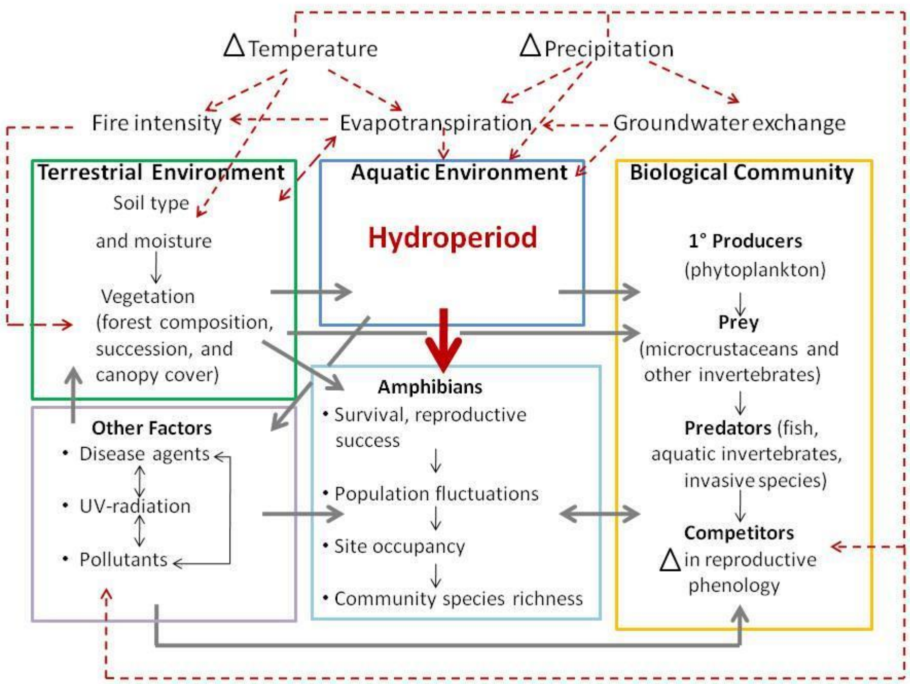

Figure 4.2. Conceptual model of pathways through which changes in temperature and precipitation may impact the resiliency of flatwoods salamander populations. Changes in temperature and precipitation directly affect terrestrial and aquatic habitats, the biological community (of which flatwoods salamanders 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.

4.4 Other Stressors

4.4.1. Disease

Disease is currently unknown in natural populations of reticulated flatwoods salamanders.

However, Whiles et al. (2004) found a parasitic nematode (*Hedruris siredonis*, family Hedruridae) in larvae of the closely related frosted flatwoods salamander from South Carolina and Florida. This parasite has been found in other ambystomatids and can cause individuals to be become undersized and thin, thus reducing their fitness (Whiles et al., 2004). The infestations were not considered heavy and were probably not having a negative impact on the larvae studied (Whiles et al., 2004). However, environmental degradation may change the dynamics between salamander populations and normally innocuous parasites (Whiles et al., 2004). "Red-leg" disease (*Aeromonas hydrophilia*), a pathogen bacterium, caused mortality of the mole salamander (*A. talpoideum*) at the breeding wetland of the reticulated flatwoods salamander in Miller County, Georgia (Maerz, 2006), and reticulated flatwoods salamanders have not been observed at this site since the disease was reported.

Ranaviruses in the family *Iridoviridae* and chytrid fungus may be other potential threats, although the susceptibility of the reticulated 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). This chytrid fungus has been found at an *A. bishopi* breeding wetland on EAFB and at a site near occupied breeding wetlands. Recently, a new species of chytrid fungus, *Batrachochytrium salamandrivorans*, was isolated from a mortality event that caused the near extinction of a population of fire salamanders (*Salamandra salamandra*) in

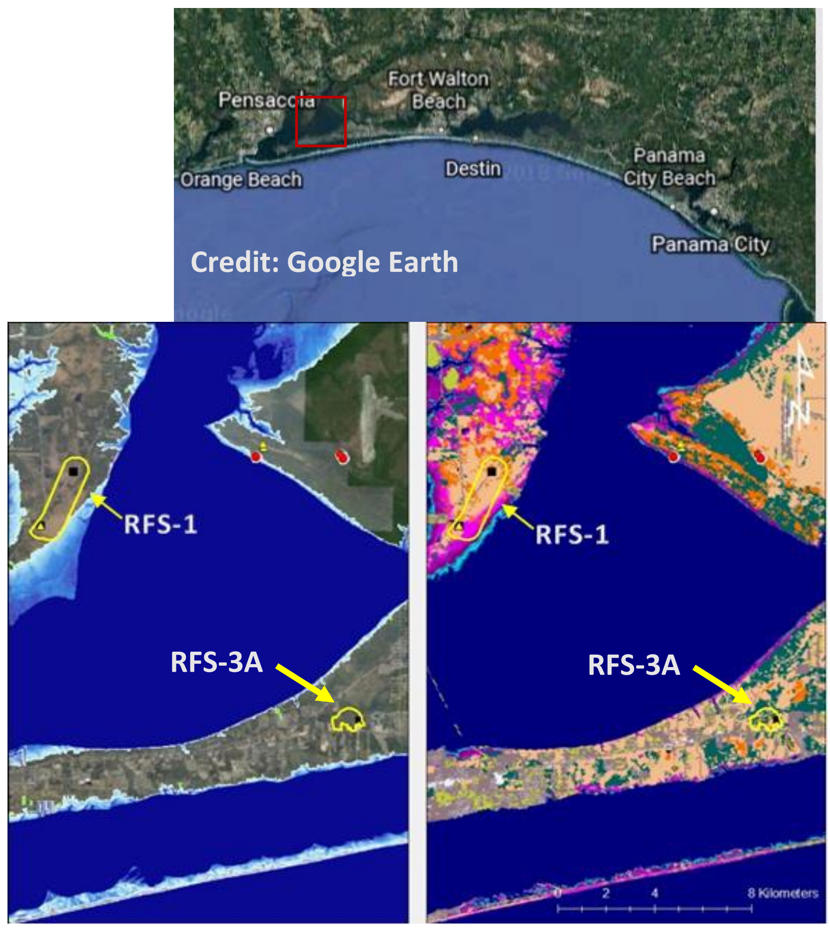

Figure 4.3. Left: Areas of inundation (in light blue) under a scenario of 6 feet of sea level rise for Critical Habitat Units RFS-1 and RFS-3A for A. bishopi. Right: Areas converted to marsh habitat (in pink) as a consequence of sea level rise.

Europe (Martel et al., 2013; Spitzen-van der Sluijs et al., 2013). Currently, it is unknown 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 *Batrachochytrium salamandrivorans* in the U.S. are currently underway. This discussion of disease in other species of closely related salamanders indicates the potential for similar threats to reticulated flatwoods salamander populations for which we will monitor. A graduate student at Louisiana State University is currently examining immune gene diversity in reticulated flatwoods salamanders (Williams, S.T. et al. unpublished abstract, 2018 SEPARC).

4.4.2 Predation

Exposure to increased predation by fish is a potential threat to the reticulated 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. Sea level rise may increase the likelihood of tidal rivers to flood flatwoods wetlands as well. Studies of other ambystomatid species have demonstrated a decline in larval survival in the presence of predatory fish (Semlitsch, 1987; 1988). Snake activity around breeding wetlands increased greatly when flatwoods salamanders and other amphibian metamorphs were emerging from the wetlands, and they may be important predators of flatwoods salamanders (Erwin et al., 2016).

Red imported fire ants (*Solenopsis invicta*) are potential predators of reticulated flatwoods salamanders, especially in disturbed areas. This species is attracted to disturbance from seasonal ponding at known breeding sites (T. Gorman, pers. comm., 2015). 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 or their invertebrate prey is recommended because the severity and magnitude, as well as the long term effect of fire ants on reticulated flatwoods salamander populations is currently unknown. We consider predation to be a threat to the reticulated flatwoods salamander at this time. Both snakes and fire ants are likely to increase foraging activity in warmer weather, so if temperatures during metamorphosis increase, metamorphs may be increasingly vulnerable. Over half the documented wetlands within the range are on private land, increasing the probability of fish being introduced into a breeding site.

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 (Burraco and 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, 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 reticulated flatwoods salamander breeding sites through rooting, so intensive approaches (e.g., control measures and fencing) may be needed to avoid degradation to occupied sites and sites going through restoration (Jones et al., 2018b).

Invasive plant species such as cogongrass (*Imperata cylindrica*) and torpedo grass (*Panicum repens*) 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). Reticulated flatwoods salamander habitat management plans will need to address threats posed by invasive plants and develop strategies to control them, especially in light of the fact that most herbicides are not selective enough to distinguish between invasive and native grasses and other native herbaceous plants. Prevention is difficult where official or unofficial off-road vehicle use occurs, but is simpler than treatment after the fact. It has been documented that cogongrass can displace most of the existing vegetation except large trees. Especially threatening to the reticulated flatwoods salamander is the ability of cogongrass to outcompete wiregrass (*Aristida* sp.), a key vegetative component of the species' natural 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 reticulated flatwoods salamanders by modeling the number of active breeding wetlands under different management and climate scenarios at two timescales (30 years and 80 years) in the future. Scenarios were selected based on the time frames of the 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 wetland to be representative of a single population. This approach is supported by studies in this and other closely-related *Ambystoma* species that suggest strong fidelity to natal wetlands, relatively short dispersal distances and limited dispersal ability, and genetic differentiation between neighboring wetlands (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 Breeding Wetland Management Scenarios

Three types of management scenarios were developed based on the current number of active breeding wetlands observed during recent surveys (2014–2018) and breeding wetland succession and restoration rates elicited from knowledgeable land managers and species experts. A wetland loss scenario estimated the loss of active breeding wetlands over time due to a reduction of oviposition and larval habitat from natural habitat succession in which wetland herbaceous vegetation is reduced by shrub encroachment and organic matter accumulation over time. This scenario assumed that no species-specific management of breeding wetlands would occur and no measurable or successful restoration of potentially suitable (but currently degraded) breeding wetlands would offset the loss of breeding wetlands over time. This represents a worst-case wetland 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 wetlands were maintained in suitable condition by species-specific wetland management activities, but without successful efforts to restore additional potential breeding wetlands. This scenario would reflect a situation where all species-specific management was focused on currently active breeding wetlands. Finally, we modeled a wetland restoration scenario in which all currently active breeding wetlands are being maintained in suitable condition and nearby unsuitable breeding wetlands 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 wetlands are maintained by appropriate species-specific management such that no succession and loss of active breeding wetlands occur, and all restored breeding wetlands are colonized by the species. This scenario is not currently achievable due to a variety of habitat management challenges including a lack of resources and trained personnel, a lack of incentives for wetland burning, the high cost of restoration activities in degraded wetlands, regulatory and administrative barriers, smoke management and liability concerns, and land manager concerns about and lack of experience with wetland burning (K.C. Ryan et al., 2013; Leda Kobziar et al., 2015; data). However, if current barriers to species management were resolved and species management was considered a top priority for land managers, this scenario might be possible. In reality, the management of breeding wetlands on most currently occupied properties lies somewhere between the wetland loss and wetland maintenance scenarios where the loss of breeding wetlands over time due to wetland succession is offset, at least to some degree, by the addition of new breeding wetlands from active restoration programs.

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 with knowledge of the species to two webinars held on May 29, 2018 and June 12, 2018 and received surveys from a total of 11 participants (26 % response rate). Responding participants included regional amphibian experts, representatives from all currently occupied states and most properties, as well as state wildlife agencies responsible for statewide species management (Appendix 1). Participants were asked to estimate the number of inactive breeding wetlands that could be restored to suitable habitat conditions for the wetland restoration scenario, as well as the number of currently active breeding wetlands that would become unsuitable for successful reproduction due to natural habitat succession without species-specific management for the wetland loss scenario within one, ten, 20, 30, and 80-year time frames. Participants were also asked to provide the lowest and highest number of breeding wetlands for each scenario (the range of possible outcomes), as well as their level of confidence for each answer 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 1-5 respondents that were familiar with the species and habitat on that property. For the sake of simplicity, it was assumed that restored wetlands would be naturally colonized by the species, although this would be unlikely if restored wetlands were greater than the presumed dispersal distance of 500-m from occupied wetlands. We also assumed that all restored wetlands 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 wetlands that could be restored based on the number of suitable wetlands on their property, or if responding for the species rangewide, an upper limit of restorable wetlands on all the currently occupied properties. The wetland maintenance scenario assumed that the current number of active breeding wetlands would remain constant over time and was not based on expert elicitation.

5.1.1. Wetland loss scenario

The number of active breeding wetlands 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 at the 30 and 80-year time intervals, indicating greater uncertainty among respondents for this scenario at these time scales. Under the wetland loss scenario, the number of active breeding wetlands decreased rapidly resulting in a mean of 0 (±0.0 SD) active breeding wetlands after 30 years for participants who responded for the species range-wide and 2 (±2.1) active wetlands after 80 years when all estimates were added for participants who responded by property (Figure 5.1, Tables 5.1–5.2). Three of the six range-wide respondents thought that all breeding wetlands in the species' range would be inactive after 20 years under this management scenario, and all but one of the range-wide respondents though that all breeding wetlands would be inactive after 30 years under this scenario (Table 5.1). Each occupied property had 2-5 respondents that were familiar with the species and habitat on that property. Under this scenario, population redundancy would be expected to decrease as the number of active breeding wetlands (each representing a population) decreased on each property. Population resiliency would also be expected to decrease under this scenario since it would lead to the degradation of breeding habitat within all wetlands resulting in less egg laying habitat and lower larval survival rates due to decreased larval cover from predators within breeding wetlands. Under this management scenario, the reproductive rate of populations would diminish as areas of breeding wetlands became gradually overgrown and unsuitable over time. At the later time scales, any remaining active breeding wetlands would have little existing egg laying habitat and extremely low population resilience due to this steady, long-term decline in reproductive success.

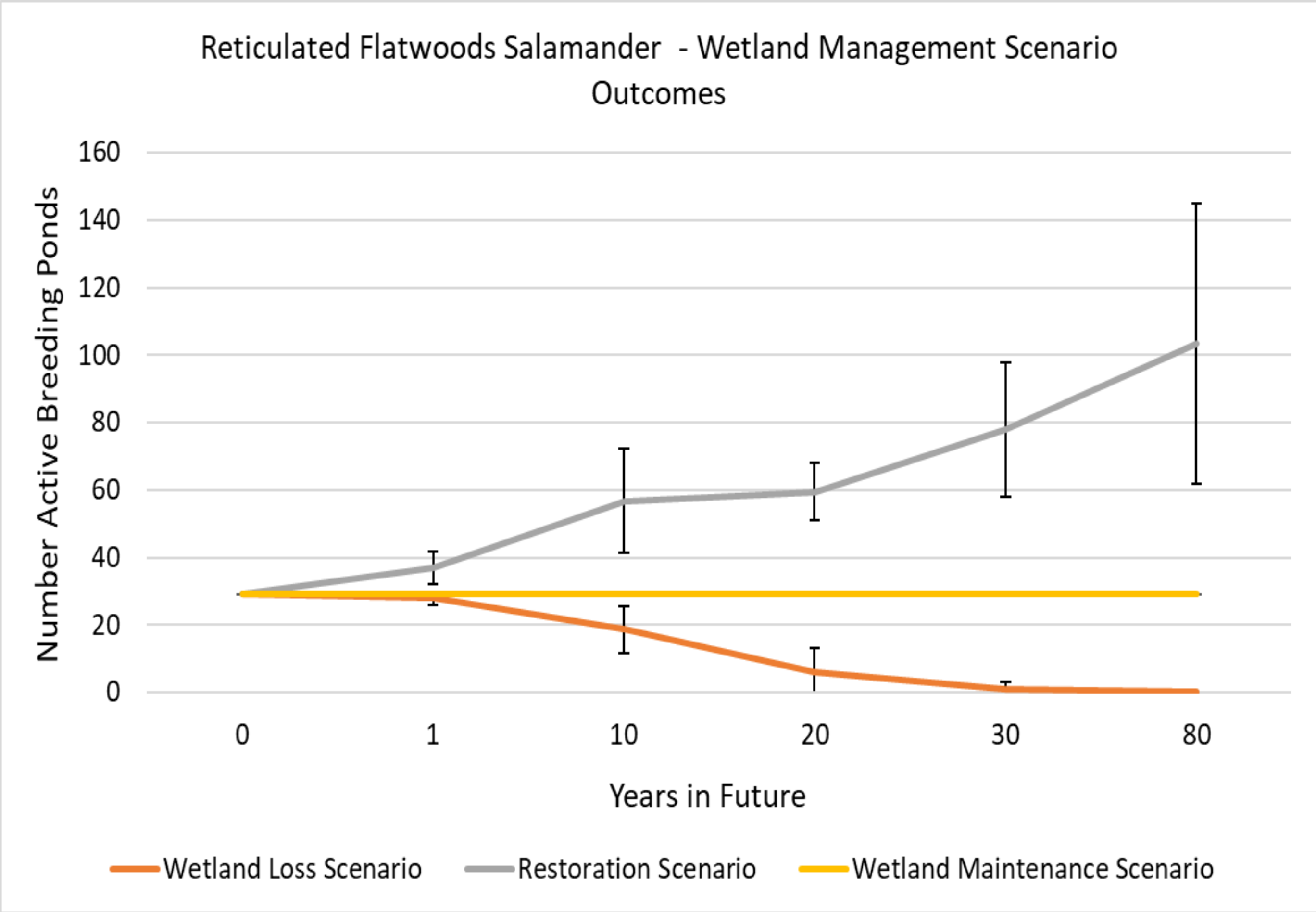

Figure 5.1. Predicted change in mean ± SD active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands under three wetland management scenarios over time. Wetland succession and restoration scenarios based on expert elicitation. Wetland maintenance scenario assumes the current number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

Table 5.1. Predicted mean (± SD) total number of active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands 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	28	20	12	-	-
Respondent 2	29	20	0	0	0
Respondent 3	24	19	0	0	0
Respondent 4	28	5	0	0	0
Respondent 5	29	24	10	0	0
Respondent 6	29	24	15	5	0
Mean (±SD)	28 (±1.9)	19 (±7.0)	6 (±6.9)	1 (±2.2)	0 (±0.0)

Table 5.2. Predicted mean (± SD) number of active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands on currently occupied properties 1, 10, 20, 30, and 80 years in the future based on elicited estimates under the wetland loss scenario for respondents responding by property. Number of respondents for each property provided in parentheses after the property name.

	Number Active Breeding Ponds				
Property	**1 Year**	**10 Years**	**20 Years**	**30 Years (Year 2050)**	**80 Years (Year 2100)**
Eglin Air Force Base, FL (3)	13 (±0)	11 (±2.1)	5 (±1.5)	2 (±2.1)	2 (±2.1)
Escribano Point WMA, FL (4)	9 (±0.5)	7 (±3.9)	4 (±2.9)	3 (±2.9)	0 (±0)
Garcon Point WMA, FL (2)	1 (±0.7)	0 (±0)	0 (±0)	0 (±0)	0 (±0)
Hurlburt Field, FL (2)	1 (±0)	1 (±0.7)	0 (±0)	0 (±0)	0 (±0)
Mayhaw WMA, GA (5)	2 (±0)	1 (±0)	0 (±0)	0 (±0)	0 (±0)
Santa Rosa County, FL (2)	1 (±0)	1 (±0.7)	0 (±0)	0 (±0)	0 (±0)

Yellow River Marsh State Park, FL (2)	1 (±0.7)	0 (±0)	0 (±0)	0 (±0)	0 (±0)
Total	**28 (±1.1)**	**21 (±4.6)**	**9 (±3.3)**	**5 (±3.6)**	**2 (±2.1)**

5.1.2. Wetland maintenance scenario

Currently, there are 29 active breeding wetlands for this species (Table 5.3). Under this scenario, the number of breeding wetlands 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 wetlands with no effective attempt to restore additional potentially suitable breeding wetlands. Under this scenario, there is little redundancy of populations on most properties as currently most properties only contain one breeding wetland. Furthermore, the resilience of populations would be maintained or potentially slightly improved on properties that currently lack adequate habitat management of active breeding wetlands. However, in general, the representation, redundancy, and resiliency of existing populations would not increase over current levels as no significant attempts would be made to restore degraded historical or potentially suitable breeding wetlands, which would limit population growth to levels that can be sustained by the current number of breeding wetlands.

Table 5.2. Number of active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands on currently occupied properties used for wetland maintenance scenario.

Property	Number Active Breeding Ponds
Eglin Air Force Base, FL	13
Escribano Point WMA, FL	10
Garcon Point WMA, FL	1
Hurlburt Field, FL	1
Mayhaw WMA, GA	2
Santa Rosa County, FL	1
Yellow River Marsh State Park, FL	1
Total	**29**

5.1.3. Wetland restoration scenario

Under the wetland restoration scenario, the number of active breeding wetlands increased rapidly resulting in a mean of 103 (±41.6 SD) active breeding wetlands after 80 years for participants who responded for the species range-wide (Figure 5.1, Table 5.4) and 213 (±11.2) active wetlands after 80 years when all estimates were added for participants who responded by property (Table 5.5). The differences between the number of breeding wetlands estimated by each calculation method (range-wide vs. by property) reflects uncertainty among experts about the maximum number of wetlands that can be restored on currently occupied properties. However, these expert predictions reflect broad agreement that the number of active breeding wetlands could be dramatically increased on occupied properties if wetland restoration efforts were maximized. Under this scenario, population resiliency and redundancy would be expected to increase as the habitat in each wetland improved and the number of active breeding wetlands (each representing an individual population) increased on each property as newly restored wetlands are colonized.

Table 5.4. Predicted mean (± SD) total number of active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands 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	35	54	55	-	-
Respondent 2	34	49	69	89	150
Respondent 3	44	79	-	-	-
Respondent 4	39	64	64	90	90
Respondent 5	32	38	50	55	70
Mean (±SD)	**37 (±4.8)**	**57 (±15.5)**	**60 (±8.6)**	**78 (±19.9)**	**103 (±41.6)**

Table 5.5. Predicted number of active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands 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. A mean and SD are provided for properties with multiple respondents. Number of respondents for each property provided in parentheses after the property name.

	Number Active Breeding Ponds				
Property	**1 Year**	**10 Years**	**20 Years**	**30 Years (Year 2050)**	**80 Years (Year 2100)**

Eglin Air Force Base, FL (2)	14 (±0)	29 (±11.3)	43 (±26.2)	76	150
Escribano Point WMA, FL (3)	14 (±1.2)	18 (±5.8)	20 (±8.7)	20 (±8.7)	20 (±8.7)
Garcon Point WMA, FL (1)	2	2	2	2	2
Hurlburt Field, FL (2)	7 (±7.1)	8 (±5.7)	9 (±4.9)	12	12
Mayhaw WMA, GA (4)	5 (±4.8)	8 (±3.9)	11 (±6.9)	13 (±7.0)	13 (±7.0)
Santa Rosa County, FL (1)	1	1	1	1	1
Yellow River Marsh State Park, FL (1)	15	15	15	15	15
Total	**58 (±8.6)**	**81 (±14.4)**	**101 (±28.8)**	**139 (±11.2)**	**213 (±11.2)**

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 this species. 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 wetlands 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 reports. Climate change scenarios in this version are based on a GIS analysis of potential inundation and vegetation changes of current breeding wetlands 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 reticulated 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 will cause some reductions in reticulated 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 reticulated flatwoods salamander breeding sites, we conducted a GIS analysis to determine which current 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 (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 in order 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 5.6–5.7.

Table 5.6. Mean and (range) of sea level rise projected for the Florida panhandle by 2050 for different CO2 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 5.7. Mean and (range) of sea level rise projected for the Florida panhandle by 2081-2100 for different CO2 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 wetlands 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 5.6 and 5.7). These sea level rise projections represent a range of potential sea levels based on different emissions scenarios (low–extreme 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 wetlands 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 reticulated flatwoods salamander breeding wetlands are projected to be inundated or affected by marsh migration under any of the sea level rise scenarios by the year 2050. In contrast, breeding wetland loss is projected to occur by Year 2100 beginning at the medium high emissions level (Table 5.8). Two breeding wetlands at Escribano Point WMA are projected to be inundated by 2100 under the medium high emissions scenario (Figure 5.2). Under the high emissions scenario, 3 breeding wetlands on Escribano Point WMA and one breeding wetland on Garcon Point WMA will be lost to a combination of direct inundation and habitat changes (Figures 5.2-5.3). Under the extreme emissions scenario (8 foot sea level rise), 6 breeding wetlands on Escribano point WMA and one breeding wetland on Garcon Point WMA will be lost to a combination of direct inundation and habitat changes (Figures 5.2-5.3). The loss of these breeding wetlands will decrease the redundancy of populations (each breeding wetland represents a population) for the species. The resiliency of the remaining Escribano Point WMA breeding wetlands might also be negatively affected if habitat conversions (salinity changes) result from their decreased distance to the coastline. It is also important to consider that sea level rise will not cease by 2100 regardless of the emissions scenario, and therefore the number of breeding wetlands inundated or lost to habitat changes will be greater than considered here at later time periods (Sweet et al., 2017, Table 5.9).

Table 5.8. Number of currently active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands 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)	0	No loss
Medium Low / RCP 4.5	0.5 m (~ 2 feet)	0	No loss
Medium	1.0 m (~ 3 feet)	0	No loss
Medium High / RCP 6.0	1.5 m (~ 5 feet)	2 (Escribano Point WMA)	Habitat changes

High / RCP 8.5	2.0 m (~6.5 feet)	4 (3 Escribano Point WMA, 1 Garcon Point WMA)	Inundation and habitat changes
Extreme	2.5 m (~ 8 feet)	7 (6 Escribano Point WMA, 1 Garcon Point WMA)	Inundation and habitat changes

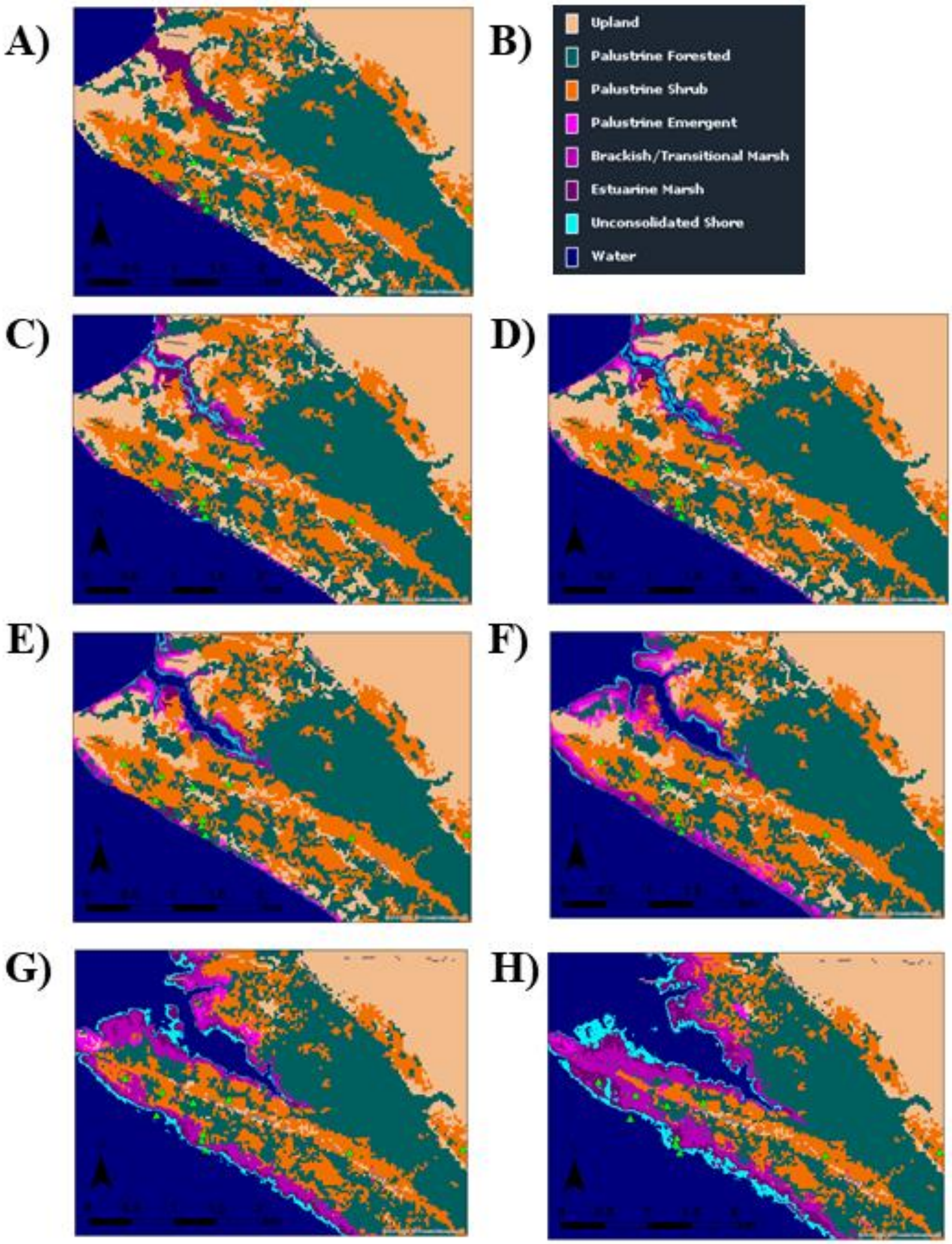

Figure 5.2. Currently active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands at Escribano Point WMA (green triangles) in relation to sea level rise and marsh migration under different emissions scenarios by Year 2100. A) current landcover B) land cover legend C) 1 ft sea level rise (low emissions) D) 2 ft sea level rise (medium low emissions) E) 3 ft sea level rise (medium emissions) F) 5 ft sea level rise (medium-high emissions) G) 6.5 ft sea level rise (high emissions) H) 8 ft sea level rise.

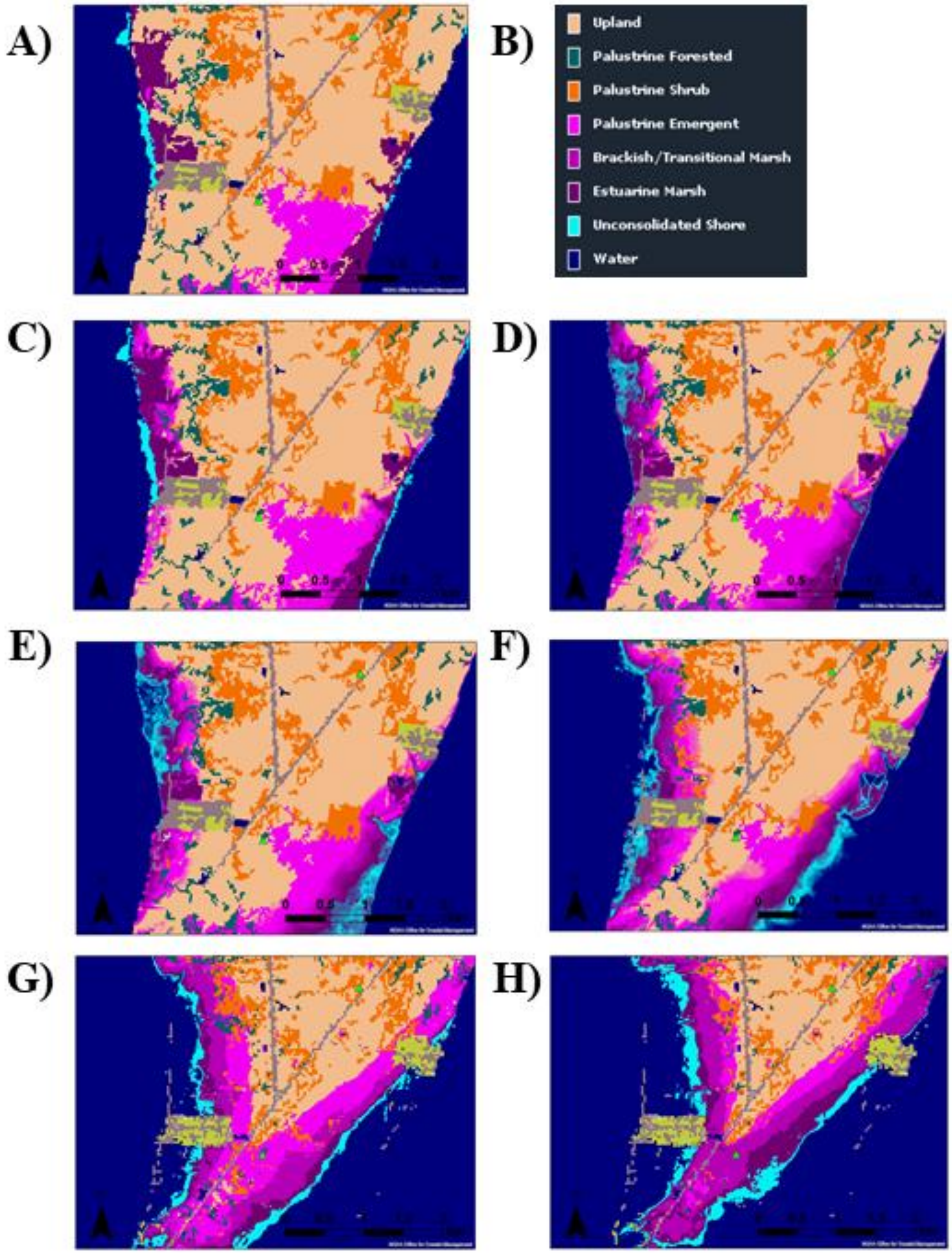

Figure 5.3. Currently active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands at Garcon Point WMA (green triangles) in relation to sea level rise and marsh migration under different emissions scenarios by Year 2100. A) current landcover B) land cover legend C) 1 ft sea level rise (low emissions) D) 2 ft sea level rise (medium low emissions) E) 3 ft sea level rise (medium emissions) F) 5 ft sea level rise (medium-high emissions) G) 6.5 ft sea level rise (high emissions) H) 8 ft sea level rise (extreme emissions) Source: NOAA Sea Level Rise Viewer.

Table 5.9. 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 wetlands 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, a minimum of one current breeding wetland on Escribano Point WMA is 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 (Figure 5.4). A maximum of nine current breeding wetlands on Escribano Point WMA and one breeding wetland on Garcon Point WMA are at risk of storm surge inundation due to rising sea levels by the Year 2100, depending on the emissions scenario (Figure 5.4). Given the lack of certainty of the long-term impacts of storm surge on species populations, we did not consider the impact of storm surge further in our future scenarios. We simply provide this information here to illustrate that storm surge will likely have additional negative, though currently poorly understood, impacts on future populations on Escribano Point and Garcon Point WMAs. Ongoing studies of the recent inundation of frosted flatwoods salamander breeding wetlands on St. Marks National Wildlife Refuge from Hurricane Michael should provide useful information on the short and long-term impacts of storm surge on population demographics and breeding wetland suitability. Once storm surge impacts are better understood, we recommend this SSA be updated to take them into account.

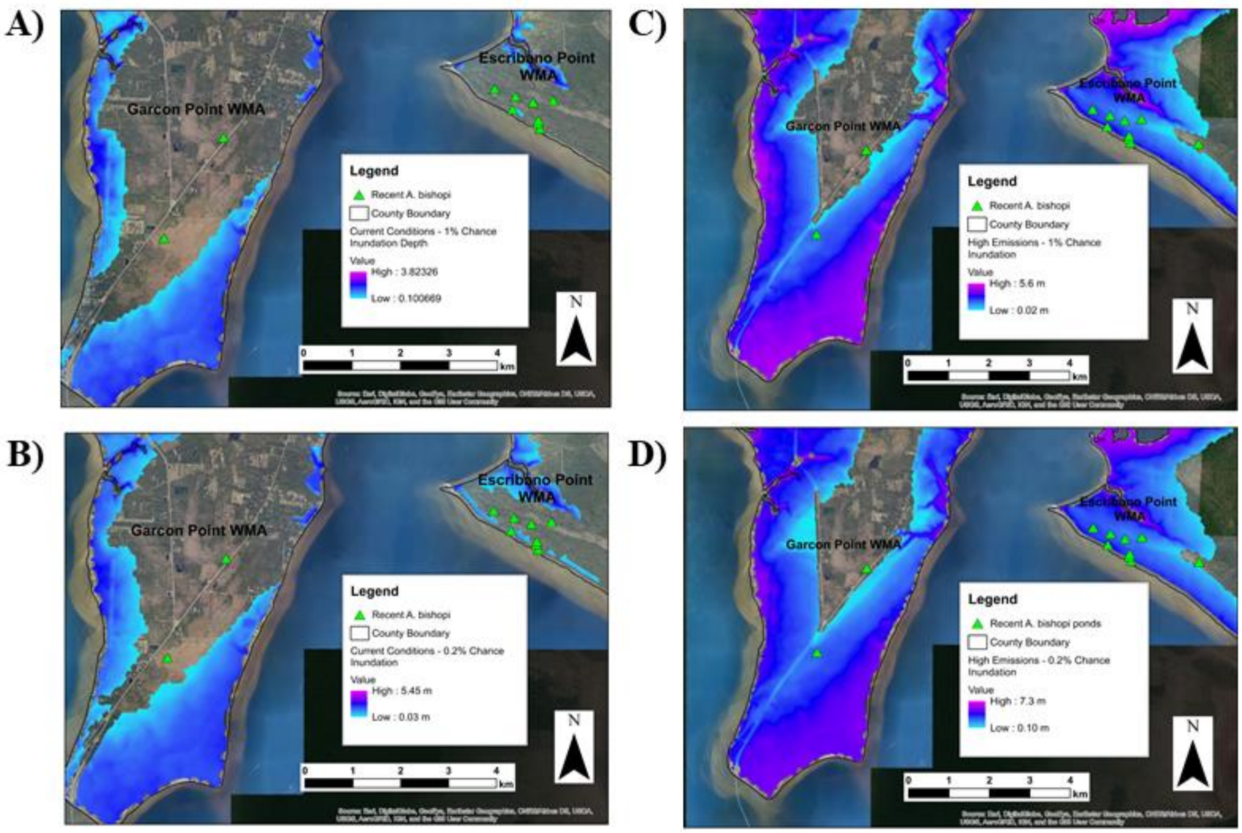

Figure 5.4. Currently active reticulated flatwoods salamander (Ambystoma bishopi) breeding wetlands in relation to A) 1% annual chance storm surge at current sea levels B) 0.2% annual chance storm surge at current sea levels C) high emissions scenarTable 6io 1% annual probability storm surge for Year 2100 and D) high emissions scenario 0.2% annual probability storm surge 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 wetlands 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 5.10–5.11), 2) the average and maximum winter and summer temperatures for the Florida panhandle since 1895 (Figures 5.5–5.6), and 3) measurements of the critical thermal maxima for various *Ambystoma* species (Figure 5.7). Species experts will be encouraged to consider direct mortality, reduction in breeding wetland hydroperiods due to increased evapotranspiration and drought and precipitation changes, and changes/degradation in wetland and upland habitat due to increased fire intensity. The currently available data suggest further loss of breeding wetlands under the higher emissions scenarios in addition to those impacted by sea level rise due to hydrological changes and increased wetland plant evapotranspiration as temperatures increase of 2–5°C (Table 5.10). In addition, under higher emissions scenarios by 2050, maximum summer temperatures may exceed the critical thermal maxima of some *Ambystoma* species (Table 5.10, Figures 5.6–5.7) 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. Maximum summer air temperatures in excess of 35°C have already been recorded in some panhandle locations (Figure 5.6). Soil and fossorial temperatures are being investigated for similar effects.

Table 5.10. 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. Regional climate predictions are generally considered more accurate for local assessments.)

	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 (low emissions)	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 (medium/low emissions)	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 (medium/high emissions)	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 (high emissions)	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 5.11. 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 two projections are outside of the current range of variation. Source: IPCC 2013. (This source provides regional climate projections vs. the global predictions in IPCC 2014. Regional predictions are generally considered more accurate for local assessments.)*

	Years 2046–2065		**Years 2081–2100**	
Climate Scenario	**October–March**	**April – September**	**October–March**	**April – September**
RCP 2.6 (low emissions)	0-10%*	0-10%*	0-10%*	0-10%*
RCP 4.5 (medium/low emissions)	0-10%*	0-10%*	0-10%	0-10%*
RCP 6.0 (medium/high emissions)	0-10%*	0-10%*	0-10%*	0-10%*
RCP 8.5 (high emissions)	0-10%*	0-10%*	0-20%	0-10%*

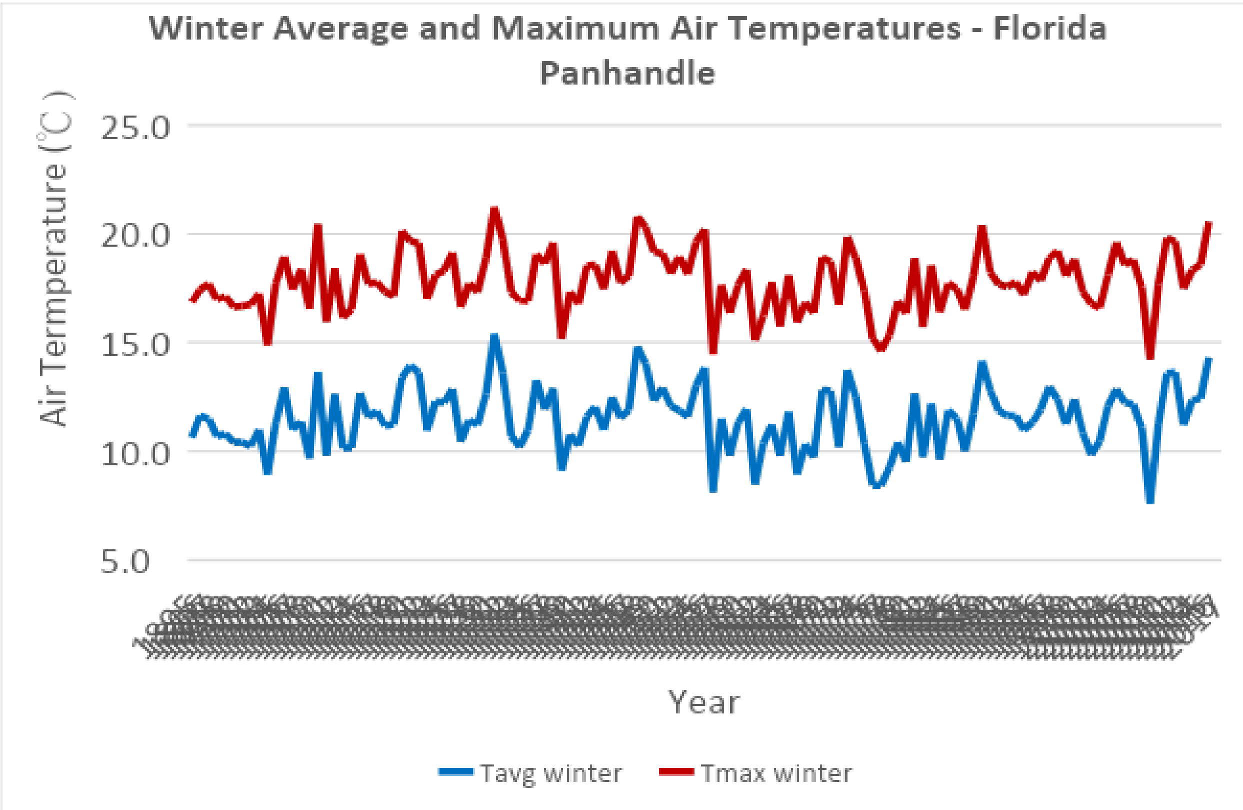

Figure 5.5. Long-term trends in average winter (December – February) air temperature in Florida panhandle 1895–2018. Source: NOAA National Climate Data Center.

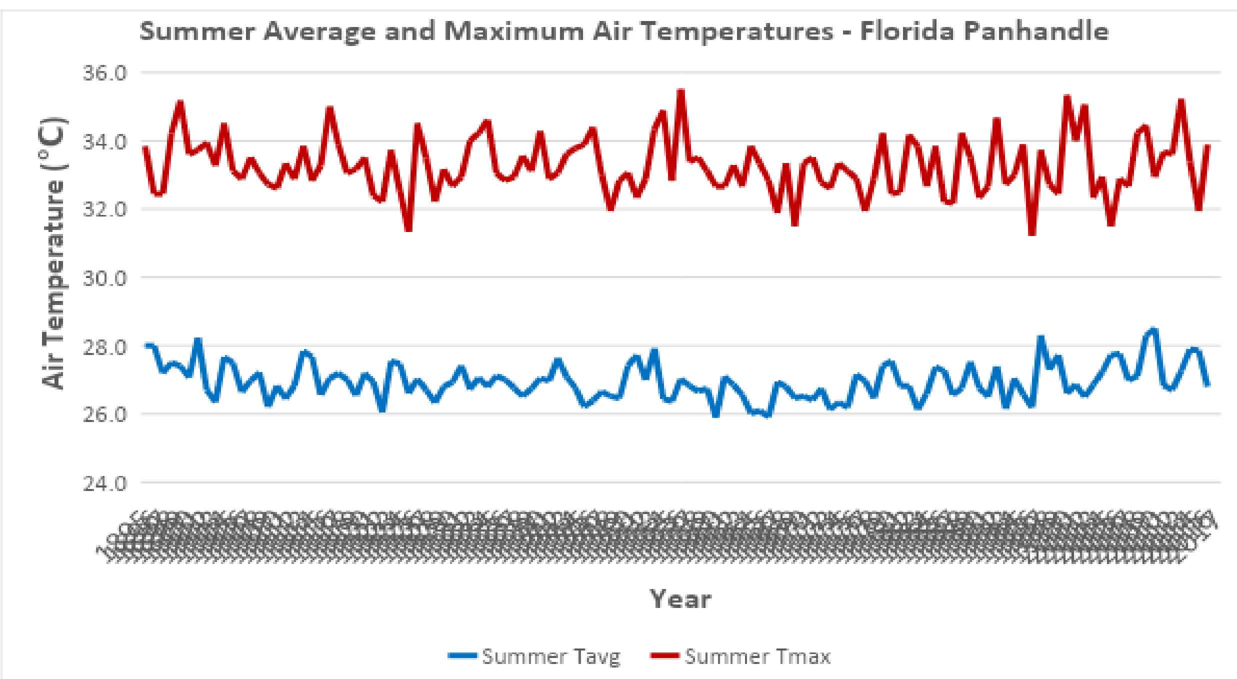

Figure 5.6. Long-term trends in average summer (June – August) air temperature in Florida panhandle 1895–2017. Source: NOAA National Climate Data Center.

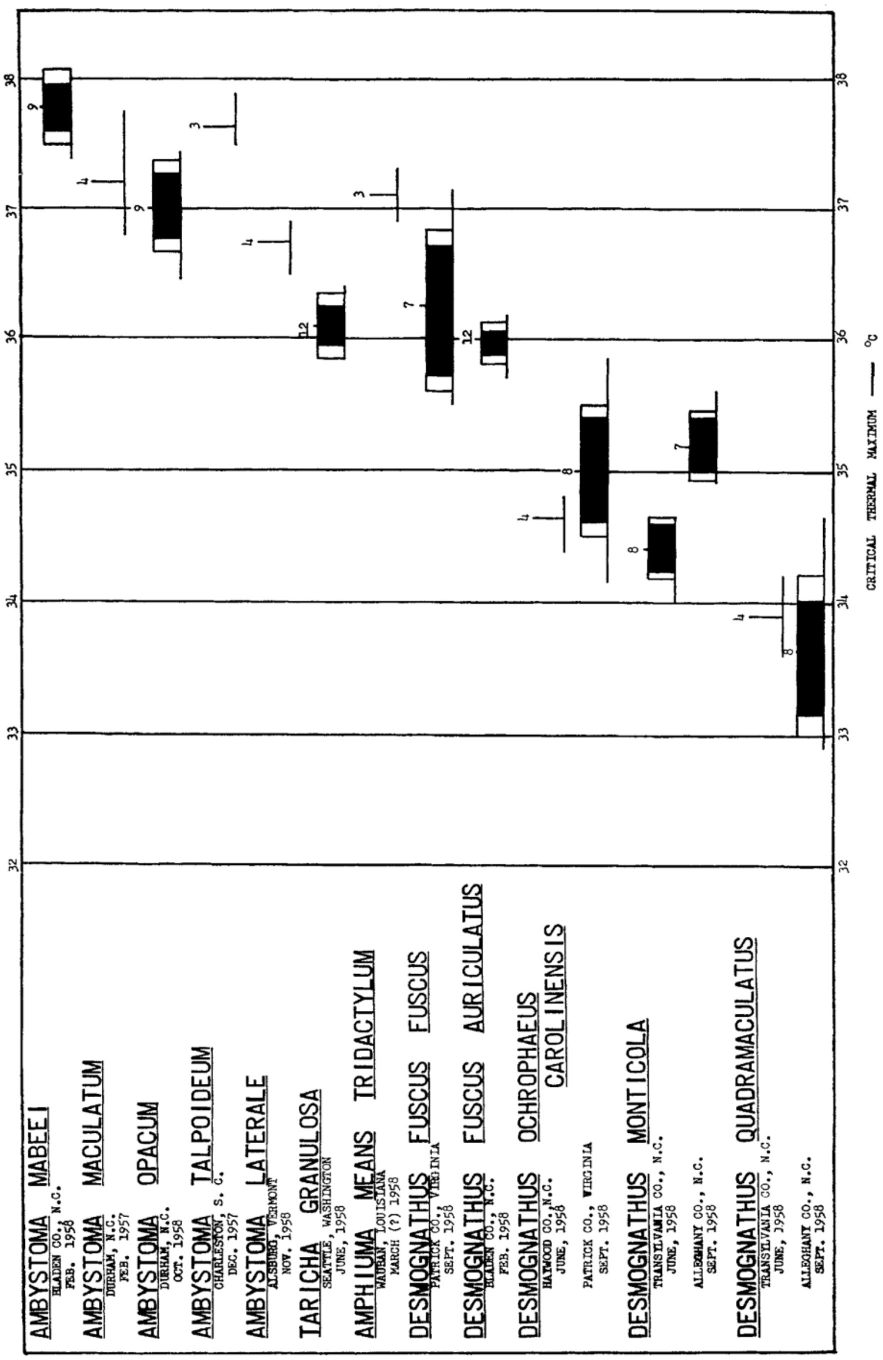

Figure 5.7. Published data on thermal maxima of salamanders including Ambystoma species. Source: Hutchinson 1961.

5.3. Creation of Combined Management and Climate Scenarios

We calculated the difference from current conditions for each combined management and climate change scenario by subtracting the number of breeding wetlands 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 5.12). We used the total number of active breeding wetlands based on predictions for individual occupied properties (as opposed to range-wide estimates) for these calculations because of the need to subtract inundated wetlands 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 Escribano Point and Garcon Point WMAs and subtracting this percentage from the projected wetlands for these properties. This approach assumes that wetlands 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 wetlands under each scenario would be less than presented here because of long-term effects of inundation by storm surge in some breeding wetlands.

Table 5.12. 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 &	Scenario 12: High Emissions & Wetland Restoration

		Wetland Maintenance	

The projected number of active breeding wetlands in Year 2050 for all combinations of management and climate scenarios is presented in Table 5.13. Given the lack of impact of sea level rise on the breeding wetlands by this time period, the combined scenarios reiterate the management scenarios at the 30-year interval described earlier. The number of active breeding wetlands is predicted to decrease to only two remaining wetlands on Eglin Air Force Base and three wetlands on Escribano Point WMA 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 in Year 2050, the representation and redundancy of species populations would be dramatically decreased. The resiliency of populations in the remaining wetlands would also likely be decreased 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 wetlands will remain steady at 29 active breeding wetlands for the Year 2050 regardless of climate change scenario. Under these scenarios, population representation and redundancy would remain the same, but population resiliency would likely decrease due to negative impacts from climate changes such as direct mortality and habitat degradation from temperature increases and increasing intensity and frequency of flood and drought events. Resiliency decreases would be greater under the higher emission scenarios. Under all wetland restoration scenarios (Scenarios 9-12; where species-specific management and restoration of all potential wetlands is maximized), the number of active breeding wetlands is estimated to more than quadruple regardless of predicted sea level increases leading to greater representation and redundancy in populations. However, there is less expert agreement on the numbers of resulting wetlands from the wetland restoration scenarios as indicated by the high standard deviation values. Thus, the wetland management scenario plays a profound role and predicted sea level increases play no role in determining the future number of active breeding wetlands 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 wetlands 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 wetland hydroperiods as has been shown over the past 119 years (Chandler et al., 2016). 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 thermal maxima of other *Ambystoma* species on the hottest summer days since the average maximum summer temperature is currently 34℃ (Hutchinson, 1961; Table 5.10, Figures 5.6–5.7). Thus, population resiliency would be maximized under the lower emissions wetland restoration scenarios (Scenarios 9-10).

Table 5.13. Projected mean (± SD) total number of active breeding wetlands for reticulated flatwoods salamanders in Year 2050 under different wetland 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)	5 (±3.6)	29 (±0.0)	139 (±11.2)
Medium Low Emissions (RCP 4.5)	5 (±3.6)	29 (±0.0)	139 (±11.2)
Medium High Emissions (RCP 6.0)	5 (±3.6)	29 (±0.0)	139 (±11.2)
High Emissions (RCP 8.0)	5 (±3.6)	29 (±0.0)	139 (±11.2)

The projected number of active breeding ponds in Years 2100 for all combinations of management and climate scenarios is presented in Table 5.14. For all combinations of the wetland loss scenarios (Scenarios 1–4), there would be little representation or redundancy of populations because wetland succession would decrease the number of active breeding wetlands to only two breeding wetlands on Eglin Air Force Base by this time regardless of climate change scenario. Population resiliency would also be low under these scenarios because of the poor habitat quality in the two remaining wetlands, the lack of population resilience due to long-term reductions in reproductive success concomitant with habitat degradation, and the lack of immigration of individuals from other populations. Furthermore, the standard deviation for these scenarios reflects disagreement among experts as to whether any active breeding wetlands would remain. Semlitsch et al. (2017) argued that many recent amphibian declines are the result of extinction debt, a situation where the loss of individual populations reaches a tipping point in which too few reproductively successful populations remain to support functioning metapopulations. Within this publication, this species was used as an example of a species in rapid decline due to an extinction debt caused by the destruction of wetland habitat. If this argument is true, the five breeding wetlands remaining in Year 2050 under the wetland loss scenarios may not be enough to sustain the species until Year 2100. In addition, under the wetland loss/high emissions scenario (Scenario 4), population resiliency in the remaining wetlands will be further reduced due to habitat degradation of wetlands from increased drought, extreme fires with higher intensities, and decreased hydroperiods caused by increased evapotranspiration within wetlands. In addition, individual animals would likely be heat stressed during the summer months as the projected mean temperature would be 4–5℃ higher, which would put the temperature close to the recorded thermal maxima of other *Ambystoma* species (range 33–38℃; Hutchinson, 1961) on the hottest summer days since the average maximum summer temperature is currently 34℃ (Table 5.10, Figures 5.5–5.6). Under the wetland maintenance scenarios (Scenarios 5–8), the number of currently active breeding wetlands would either decrease slightly under the medium high and high emissions scenarios due to sea level rise or stay the same under the low and medium low emissions scenarios by the year 2100. Thus, population representation and redundancy would either slightly decrease or stay the same. However, given that there are only 29 remaining populations for this species throughout its range, any further population declines should cause concern for species persistence. In addition, population resiliency would likely decrease under these scenarios due to negative impacts from climate changes such as direct mortality and habitat degradation from temperature increases and increasing intensity of flood and drought events. Resiliency decreases would be greatest for the highest emission scenarios. Under the wetland restoration scenarios (Scenarios 9–12), losses to sea level rise on Garcon Point and Escribano Point WMAs at the higher emissions scenarios are mitigated by wetland restoration efforts elsewhere resulting in large numbers of active breeding wetlands (each representing an individual population) leading to greater population representation and redundancy. Overall, population resiliency would be maximized under the lower emissions wetland restoration scenarios (Scenarios 9–10) as the habitat in each breeding wetland improved and the number of active breeding wetlands (each representing an individual population) increased on each property as newly restored wetlands are colonized. Under the higher emissions scenarios for each of the wetland management scenarios (Scenarios 2–4, 6–8, and 10–12) population resiliency in the remaining wetlands would decrease due to climate change impacts including habitat degradation in the wetlands due to increased drought, extreme fires with higher intensities, 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 recorded thermal maxima of other *Ambystoma* species since the average maximum summer air temperature is currently 34℃ (Table 5.10, Figures 5.5–5.6). 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. Despite the unknowns, given the likelihood of climate change impacts and the certainty of wetland management impacts, the representation, redundancy, and resiliency of this species will be maximized by Scenario 9.

Table 5.14. Projected mean (± SD) total number of active breeding wetlands for reticulated flatwoods salamanders in Year 2100 under different wetland management and climate emissions scenarios based on GIS analysis of sea level rise projections, elicited climate change impacts from subject matter experts, 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)	2 (±2.1)	29 (±0.0)	139 (±11.2)
Medium Low Emissions (RCP 4.5)	2 (±2.1)	29 (±0.0)	139 (±11.2)

Medium High Emissions (RCP 6.0)	2 (±2.1)	27 (±0.0)	135 (±11.2)
High Emissions (RCP 8.0)	2 (±2.1)	25 (±0.0)	130 (±11.2)

5.4 Future Changes Summary

The future of reticulated flatwoods salamanders is dependent on wetland management. While both sea level rise and increasing temperatures due to climate change are predicted to decrease the number and resiliency of populations by 2100, the choice of management scenario has profound impacts on the number of breeding wetlands and salamander populations in both the short and long-term. If species-specific wetland management (regularly burning of breeding wetlands when they are dry) is not conducted, most active breeding wetlands will become inactive by the Year 2050. However, it is not enough to simply actively manage the breeding wetlands that are currently occupied, as sea level rise and associated marsh migration are projected to result in the loss of currently active breeding wetlands at Escribano Point and Garcon Point WMAs under half of the climate change scenarios by the Year 2100. Climate changes in temperature and precipitation extremes (floods and droughts) will also negatively impact all populations under all but the lowest emissions scenario. 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 wetland losses to sea level rise and other climate changes. Wetland restoration efforts should be primarily focused on inland areas with potentially suitable habitat in the range of the species, which are not anticipated to be affected by sea level rise in the next 80 years. Similarly, long-term protection (via acquisition or easements) should focus on this portion of the species range. Currently, many managers face challenges or lack the resources to maintain all active breeding wetlands or restore additional wetlands. 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 recover the reticulated flatwoods salamander. As discussed earlier in sections 2.8 and 3.4, we estimate this species will require at least 101 resilient metapopulations rangewide (34 per RMU) to persist into the future at least 40 years with a reasonable risk of extinction. Each of these metapopulations would include at least 14 suitable wetlands within a 1.5 km radius of a cluster that includes at least 3 regularly occupied (larval detections once every 3 years) wetlands within an approximately 500 m radius and connected to 22 or more wetlands through stepping stone arrangement, surrounded by suitable upland habitat. Thus, the wetland restoration and management efforts simulated here are necessary for recovery. However, habitat restoration and continued 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 most 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
Barbara Almario	Land manager	Florida Fish and Wildlife Conservation Commission	Wildlife Biologist	Escribano Point Wildlife Management Area
Dr. Kurt Buhlmann	Species expert	University of Georgia, Savannah River Ecology Laboratory	Senior Research Associate	Holley Airfield
Anna Farmer	Species expert	Florida Fish and Wildlife Conservation Commission	Amphibian Research Lead	All Florida
Dr. Carola Haas	Species expert	Virginia Tech	Professor, Wildlife Ecology	Eglin Air Force Base
Pierson Hill	Species expert	Florida Fish and Wildlife Conservation Commission	Research Associate	All Florida
Brent Howze	Land manager	Georgia Dept. of Natural Resources	Senior Wildlife Biologist	Mayhaw Wildlife Management Area
John Jensen	Species expert	Georgia Dept. of Natural Resources	Wildlife Biologist	All Georgia
Kelly Jones	Species expert	Virginia Tech	Wildlife Biologist	Eglin Air Force Base
Lorraine Ketzler	Land manager	U.S Fish and Wildlife Service	Natural Resources Manager	Hurlburt Field

Dr. Lora Smith	Species expert	Joseph W. Jones Ecological Research Center	Scientist	Ichauway/Mayhaw WMA
Mark Winland	Land manager	Florida Fish and Wildlife Conservation Commission	Wildlife Biologist	Escribano Point Wildlife Management Area