Species Status Assessment Report for the Florida Pinesnake (*Pituophis melanoleucus mugitus)*

Version 1.1

Photo Credit: P. Hill

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Version Updates**:**

The changes from Version 1.0 (January 2022 – Peer Review) included parameter changes to how the model predicted current and future conditions based on peer review comments, a re-run of the current and future conditions resulting in changes to how we summed up the 3R's, addition of pertinent information throughout the document, as well as minor grammatical changes.

EXECUTIVE SUMMARY

This report summarizes the results of a Species Status Assessment completed for the Florida pinesnake (*Pituophis melanoleucus mugitus*), a subspecies of the eastern pinesnake, to assess the species' overall viability. We considered what the subspecies needs to maintain viability by characterizing the status of the species in terms of its resiliency, representation, and redundancy (3R's). We provide a thorough assessment of ecology and individual needs of the species, followed by a description of the factors influencing viability, and then a description of the species' current condition and predicted future condition.

The Florida pinesnake is a large, non-venomous, heavy-bodied snake that is associated with xeric, open-canopied southern pine uplands, including longleaf pine (*Pinus palustris*), savanna, flatwoods, and sandhills from southeastern South Carolina, Georgia, Florida, and west to Alabama. The Florida pinesnake is diurnal and highly fossorial, spending much of its time underground and hibernate during the winter and may estivate in the summer depending on environmental conditions. Pinesnakes are active foragers that hunt both above and below ground. These non-venomous, powerful constrictors consume a variety of prey. Small mammals, including rabbits, southeastern pocket gophers (*Geomys pinetis*), and other rodents, make up a large portion of the pinesnake's diet.

The potential factors that could be affecting the viability of the Florida pinesnake include the following: (1) habitat loss, conversion, and fragmentation from loss of longleaf pine savanna habitat; (2) road mortality; (3) invasive species, such as the red imported fire ant and feral hogs; (4) effects of climate change resulting in increased temperatures, decreased precipitation, increased severe weather such as drought, flooding, or storms, changes in wildfire frequency and intensity, decreased ability to conduct prescribed burns, and sea level rise (SLR); (5) the collection of individual snakes for the pet trade and persecution by humans; and (6) impacts that a potential disease outbreak, such as snake fungal disease may have on existing populations.

For this assessment, we defined populations as contiguous areas surrounding known Florida pinesnake occurrences with habitat conducive to survival, movement, and inter-breeding among individuals within the area. We used 1,989 species records from the year 1881-2021 to define 153 populations.

To assess the current condition of the Florida pinesnake, we estimated persistence probability for 133 of the populations. The current persistence probability could not be estimated for 2 populations in the analysis area that represented two records that had no date associated with them or for an additional 18 populations representing 65 records that are outside the currently accepted range of the subspecies, which was used as the analysis area. Current resiliency was assessed by modeling persistence probability using occurrence records, habitat suitability, protected areas, roads, and population connectivity. We measured Florida pinesnake population resiliency by using current persistence probabilities (between 0 and 100%) and summarized

results by grouping populations into categories representing ranges of persistence probabilities. The categories we chose were unlikely or extirpated (< 50%), more likely than not on landscape (50 – 79%), very likely on landscape (80 – 94%), and extremely likely on landscape or extant (95 – 100%). For a population to be highly resilient, it must have a relatively high current persistence probability.

Representation reflects the ability of a species to adapt to changing environmental conditions and can be measured by the breadth of genetic or environmental diversity within and among populations. Redundancy reflects the ability of a species to withstand catastrophic events over time by having multiple, widely distributed populations. Since we did not have species-specific genetic and ecological diversity information, we captured representation and redundancy by grouping EPA IV ecoregions into seven representative units, which captured different ecological settings within the species' range, and assessing the number and geographic distribution of resilient populations across and within each unit and across the entire range. To have high representation, the species must have highly resilient populations located in each of the representative units, and resilient populations should span the latitudinal and longitudinal extent of historical populations. To have high redundancy, the Florida pinesnake would need to have multiple resilient populations within a representative unit and throughout its range.

Current resiliency, as measured by current persistence probabilities, for the Florida pinesnake has decreased from historical conditions. Range-wide, we estimated that we have likely lost 30.8% (41/133) of total Florida pinesnake populations (i.e., populations with a less than 50% probability of current persistence), with 92 populations considered currently extant. Conversely, 69.2% of total populations had a greater than 50% probability of current persistence as of 2021. Moreover, we estimated that 49.6% of the total populations were very likely or extremely likely to persist as of 2021 (moderate and high resiliency), and 21.8% of total populations were extremely likely to persist as of 2021 (high resiliency). When looking at only currently extant populations (n=92), 71.7% (66/92) are very likely or extremely likely to persist (≥80% probability of persistence), and 31.5% (29/92) are extremely likely (>95% probability of persistence) to persist as of 2021.

For representation and redundancy, we estimated that across all populations over time, 69.2% of populations are more likely than not to currently exist on the landscape as of 2021. We estimated there has likely been some range-wide loss of redundancy. Out of 133 populations, we estimated that 92 are more likely than not (>50% probability) persisting as of 2021. We considered these to be extant populations. Populations with higher estimated persistence probabilities were spread evenly across the subspecies' range. We estimated there is a minimum of at least one Florida pinesnake population within each of the representative units that are very likely to persist on the landscape today (≥80%, moderate to high resiliency).

Table ES-1. Number and percentage of Florida pinesnake populations in each persistence category, cumulative number of populations at or above each threshold for all populations (n=133) and for currently extant populations (n=92) in 2021, and area occupied.

Population Persistence	Number of populations in each category	% of total	Cumulative number of populations at or above each threshold	% of total (% of extant populations)	Area occupied (acres/ km2)	% of total area occupied
Extremely Likely on Landscape (Extant) 95-100%	29	21.8%	29	21.8% (31.5%)	10662434.5/ 43167.8	69.6%
Very Likely on Landscape 80-94.9%	37	27.8%	66	49.6% (71.7%)	2232475.2/ 9038.4	14.6%
More Likely than Not 50-79.9%	26	19.5%	92	69.2% (100%)	941039.6/ 3809.9	6.2%
Unlikely < 50% (Extirpated)	41	30.8%	-	-	941039.6/ 5971.4	9.6%
Total	133	100%			15310880.8/ 61987.4	100%

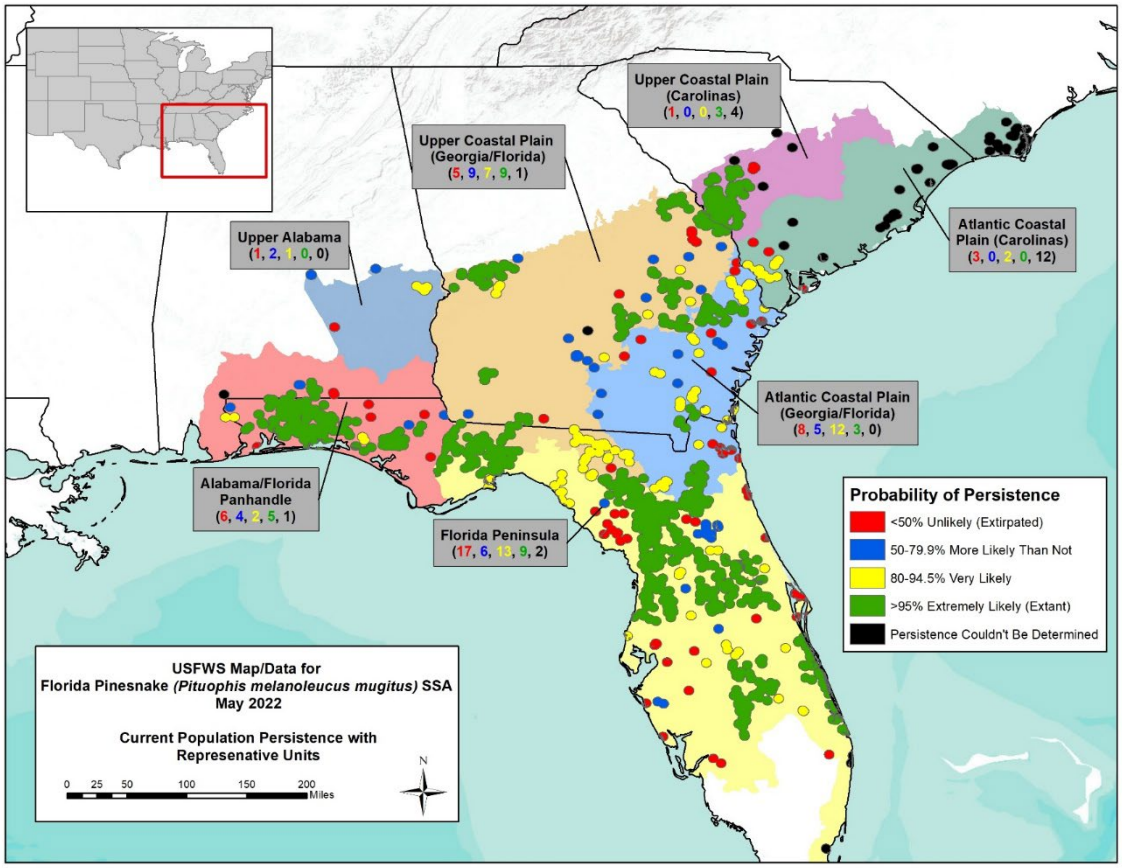

Figure ES-1. Florida pinesnake populations across representative units based on probability of persistence. Green populations are extremely likely to currently occur on the landscape or be extant; yellow populations are very likely; blue more likely than not; red populations are unlikely to currently

In evaluating future conditions for the Florida pinesnake, we developed a simulation model that predicted population persistence through 2080 using current persistence and habitat conditions under six scenarios. We used the same metrics from the current conditions analysis to characterize future resiliency, representation, and redundancy, which we summarized at 2040, 2060, and 2080. Because the simulation model accounted for year-to-year variation and uncertainty, predictions varied each time we ran the model. Therefore, we additionally characterized future conditions by calculating the mean (the most likely prediction) and 95% confidence intervals for the number of persisting populations in each representative unit and range-wide each year through 2080. All six scenarios yielded nearly identical predictions of the number and percentage of resilient populations. These patterns were seen across all future time horizons (2040, 2060, and 2080). To simplify the reporting of our results since there was so little difference between the scenarios, we only show the results for the Low Urbanization/RCP 4.5 (best case) and High Urbanization/RCP 8.5 (worst case).

For all scenarios, we predicted that future population persistence decreased from current conditions and most populations that fell within the extremely likely on landscape (≥95%, high resiliency) threshold under current conditions were predicted to have lower persistence probabilities in the future. The number and percentage of populations likely to be extirpated (<50%) increased for all scenarios and future time horizons, relative to current conditions. For example, the percentage of total populations likely to be extirpated (<50%) in 2021 is 30.8% of the total populations, leaving 92 currently extant populations. Under the Low Urbanization/RCP 4.5 scenario by 2040, that total percentage increased to 42.9%, which represents 16 additional populations predicted to be extirpated. For the currently extant populations (n=92), this represents 17.4% of populations, covering 3.9% of the total currently occupied range area. By 2060, the percentage increased to 58.6% total populations over time (an additional 21 populations predicted to be extirpated), and by 2080 we predicted the loss of an additional 18 populations for a total of 96 populations over time (72.2% of total populations over time) predicted to become extirpated.

We summarized trends in the future condition of the Florida pinesnake for representation and redundancy using the number of populations likely to persist and their spatial distribution. Overall, we predicted future representation and redundancy to decrease from current conditions. The number of resilient populations and the number of representative units with resilient populations decreased under all the scenarios. In our current condition analysis, we determined that 5 of our 7 representative units have at least 3 populations extremely likely to be on the landscape (persistence probability ≥ 95%, high resiliency), and the other two units (Atlantic Coastal Plain (Carolinas) and Upper Alabama) had at least 1 population very likely to be on the landscape (persistence probability ≥ 80%, moderate and high resiliency). By 2080, we estimate that only 1 of the 7 representative units will still contain one population at the ≥ 95% threshold, for the “worst case” High Urbanization/RCP 8.5 scenario. Two representative units are predicted

to have zero populations at or above the more likely than not to persist threshold (≥ 50%) by 2080.

This assessment shows that there have been range-wide declines for this subspecies from its historical number of populations of 133 to current conditions with 92 extant population. Our analysis indicates that future resiliency, representation, and redundancy, as measured by future population persistence, for the Florida pinesnake is predicted to further decline from current conditions under all scenarios considered.

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CHAPTER 1 – INTRODUCTION

The Florida pinesnake (*Pituophis melanoleucus mugitus*), a subspecies of the eastern pinesnake, *(Pituophis melanoleucus)*, is associated with xeric, open-canopied southern pine uplands, including longleaf pine (*Pinus palustris*), savanna, flatwoods, and sandhills from southeastern South Carolina, Georgia, Florida, and west to Alabama. The U.S. Fish and Wildlife Service (USFWS) was petitioned to list the Florida pinesnake as endangered or threatened under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531-1543) (ESA), in July 2012 as a part of the Petition to List 53 Amphibians and Reptiles in the United States by the Center for Biological Diversity (Center for Biological Diversity 2012, pp. 156–163). On July 1, 2015, the USFWS published a 90-day finding that the petition presented substantial scientific or commercial information indicating that listing may be warranted for 30 species, including the Florida pinesnake (80 FR 56423, September 18, 2015). A review of the status of the species was initiated to determine if the petitioned action is warranted. Based on the status review, the USFWS will issue a 12-month finding for the Florida pinesnake.

We conducted a Species Status Assessment (SSA) to compile the best available data regarding the subspecies' biology and factors that influence the subspecies' viability. The Florida pinesnake SSA Report is a summary of the information assembled and reviewed by the USFWS and incorporates the best scientific and commercial data available. This SSA documents the results of the comprehensive status review for the Florida pinesnake and serves as the biological underpinning of the USFWS's forthcoming decision (12-month finding) on whether the species warrants protection under the ESA.

The SSA framework (Figure 1-1; USFWS 2016, entire) is 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 SSA is intended to be easily updated as new information becomes available and to support all functions of the Endangered Species Program. It may be used to inform Endangered Species Act decision making, such as listing, recovery, Section 7, Section 10, and reclassification decisions (the latter four decision types are only relevant should the species warrant listing under the ESA). We developed this SSA to summarize the most relevant information regarding life history, biology, and factors influencing viability for the Florida pinesnake. In addition, we describe the current condition and forecast the possible response of the subspecies to various future factors and environmental conditions using plausible scenarios to formulate a risk profile for the Florida pinesnake.

The objective of this SSA is to thoroughly describe the viability of the Florida pinesnake based on the best scientific and commercial information available. Through this description, we determined what the species needs to support viable populations, its current condition in terms of those needs, and its forecasted future condition under plausible future scenarios. In conducting

this analysis, we took into consideration past, current, and plausible future conditions to help inform the viability of the subspecies.

For the purpose of this assessment, we define **viability** as a description of the ability of a species to sustain populations in the wild over time. Viability is not a specific state, but rather a continuous measure of the likelihood that the species will sustain populations over time (USFWS 2016, p. 9). Using the SSA framework (Figure 1-1), we consider what the species needs to maintain viability by characterizing the status of the species in terms of its **resiliency**, **representation**, and **redundancy** (Shaffer and Stein 2000, pp. 308–311; USFWS 2016, entire).

To sustain populations over time, a species must have the capacity to withstand:

(1) environmental and demographic stochasticity and disturbances (Resiliency),
(2) novel changes in its biological and physical environment (Representation), and
(3) catastrophes (Redundancy).

A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is better able to adapt to novel changes and to tolerate environmental stochasticity and catastrophes. In general, species viability will increase with increases in resiliency, redundancy, and representation (Smith et al. 2018, p. 306).

Resiliency is the ability of a species to withstand environmental stochasticity which includes normal, year-to-year variations in environmental conditions such as temperature or rainfall and periodic disturbances within the normal range of variation (fire, floods, storms), and to withstand demographic stochasticity (normal variation in demographic rates such as mortality and fecundity (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

Resiliency is assessed by evaluating population level characteristics such as: demography (abundance and the components of population growth rate – survival, reproduction, and migration), genetic health (effective population size and heterozygosity), connectivity (gene flow and population rescue), and habitat quantity, quality, configuration, and heterogeneity. For species prone to spatial synchrony (regionally correlated fluctuations among populations), distance between populations and degree of spatial heterogeneity (diversity of habitat types or microclimates) are also important considerations.

Resiliency is measured using metrics that describe population condition and habitat; in the case of Florida pinesnake, we used estimates of current persistence probabilities to assess population resiliency.

Representation is the ability of a species to adapt to both near-term and long-term changes in its physical environment (e.g., climate conditions, habitat conditions, habitat structure) and biological environment (e.g., pathogens, competitors, predators). This

ability to adapt to new environments, referred to as adaptive capacity, is essential for viability, as species need to continually adapt to their continuously changing environments (Nicotra et al. 2015, p. 1269) . Species adapt to novel changes in their environment behaviorally (e.g., moving to new, suitable environments) or through genetic changes that underlie physical or behavioral phenotypes better suited to new environmental conditions (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270). The latter occurs via the evolutionary process of natural selection and is affected by gene flow, mutations, and genetic drift (Crandall et al. 2000, pp. 290–291; Zackay 2007, p. 1; Sgro et al. 2011, p. 327).

Representation is assessed by examining the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas in light of evolutionary influences. In assessing the breadth of variation, it is important to consider both larger-scale variation (such as morphological, behavioral, or life history differences which might exist across the range and environmental or ecological variation across the range), and smaller-scale variation (which might include measures of genetic diversity within and among populations). In assessing the dispersal ability, it is important to evaluate the ability and likelihood of the species to track suitable habitat and climate over time. Lastly, to evaluate the evolutionary processes that contribute to and maintain adaptive capacity, it is important to assess (1) natural levels and patterns of gene flow, (2) degree of ecological diversity occupied, and (3) effective population size. In our species status assessments, we assess all three facets to the best of our ability based on available data.

The number and distribution of resilient populations across a spatially explicit unit based on ecoregions was used to assess representation for the Florida pinesnake.

Redundancy is the ability of a species to withstand catastrophic events. Catastrophes are defined here as rare, destructive events or episodes that may have impacts on a population or multiple populations and are expected to lead to population collapse regardless of population health and for which adaptation is unlikely (Mangel & Tier 1993, p. 1083).

Redundancy is about spreading risk among populations, and thus, is best assessed by analyzing the number and distribution of populations relative to the scale of anticipated species-relevant catastrophic events. The analysis entails assessing the cumulative risk of catastrophes occurring over time. Redundancy can be analyzed at a population or regional scale, or for narrow-ranged species, at the species level. The more resilient populations the species has distributed over a larger area, the better the chances that the species can withstand catastrophic events.

For the Florida pinesnake, we used the number and distribution of resilient populations within the representative units and across the range of the species to measure redundancy.

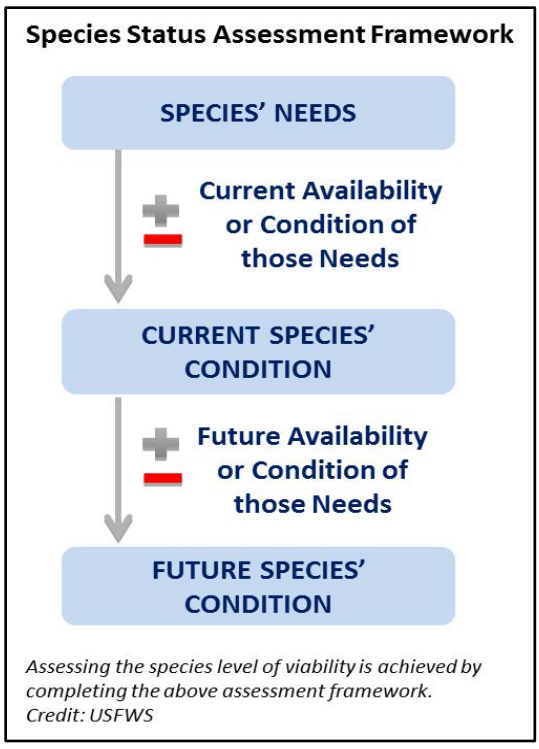

Figure 1-1. Species Status Assessment Framework.

This SSA Report includes the following chapters:

1. Introduction
2. Subspecies Ecology and Individual Needs. The life history of the subspecies and resource needs of individuals.
3. Factors Influencing Viability. A description of likely causal mechanisms, and their relative degree of impact (positive or negative), on the status of the subspecies.
4. Population and Subspecies Needs and Current Condition. A description of what the species needs across its range for viability, and estimates of the species' current resiliency, representation, and redundancy.
5. Future Conditions and Viability. Descriptions of plausible future scenarios and predictions of their influence on Florida pinesnake resiliency, representation, and redundancy.

Cited literature can be found after the final chapter. Additional supplemental information and analysis were used to complete this SSA Report. Details for the current and future condition analysis is presented in Appendix A.

This SSA Report provides a thorough assessment of what is known of the biology and natural history and assesses demographic risks, stressors, and limiting factors in the context of determining the viability and risks of extinction for the Florida pinesnake. Importantly, this SSA

Report does not result in, nor predetermine, any decisions by the USFWS under the ESA. In the case of the Florida pinesnake, the SSA Report does not determine whether the subspecies warrants protections of the ESA, or whether it should be proposed for listing as a threatened or endangered species under the ESA. That decision will be made by the USFWS after reviewing this document, along with the supporting analysis, any other relevant scientific information, and all applicable laws, regulations, and policies. The results of the decision will be announced in the *Federal Register*. The contents of this SSA Report provide an objective, scientific review of the available information related to the biological status of the Florida pinesnake.

CHAPTER 2 – SUBSPECIES ECOLOGY AND INDIVIDUAL NEEDS

In this chapter, we provide biological information about the Florida pinesnake, including its taxonomic history, morphological description, historical and current distribution, and range, and known life history. We then outline the resource needs of individuals.

2.1 Taxonomy

The eastern pinesnake (*Pituophis melanoleucus*) was first described by Daudin as *Coluber melanoleucus* in 1803 (pp. 409–410). The species was then reassigned to the genus *Pituophis* by Holbrook (1842, p. 8), who went on to describe the species in great detail. The Florida pinesnake subspecies was then described in 1921 by Barbour (pp. 117–118) based on a specimen collected ten miles north of West Palm Beach, Florida (Barbour 1921, pp. 117–118). The Florida pinesnake (*P. m. mugitus*) is one of three currently recognized subspecies of *P. melanoleucu*s distributed across the eastern United States (Rodríguez-Robles & De Jesús-Escobar 2000, p. 35; Crother et al. 2017, p. 74). The other subspecies include the northern pinesnake (*P. m. melanoleucus*) and the black pinesnake (*P. m. lodingi*) and much of the delineation for the three subspecies comes from differences in morphology. Currently, areas of intergradation between the Florida and northern pinesnake subspecies are recognized based on morphology; however, the taxonomy of the eastern pinesnake is an area of active research, which could have implications for how the Florida pinesnake subspecies is delineated. There are two recent genetic analyses of eastern pinesnakes that indicate support for *P. m. lodingi* as a distinct subspecies and multiple, geographically structured lineages within *P. m. mugitus* (Apodaca et al. 2021, entire; Nikolakis et al. 2022, entire). Those analyses also indicate introgression among *P. m. mugitus* and *P. m. lodingi* as well as among *P. m. mugitus* and *P. m. melanoleucus*. Despite historic and potentially contemporary introgression, there remains reasonable support for *P. m. mugitus* as a subspecies distributed across the coastal plains from south central Alabama to – potentially – southeastern North Carolina and peninsular Florida. Therefore, for the purposes of this SSA, we describe the Florida pinesnake and its distribution based on this current genetic information in conjunction with the widely accepted distribution among experts with the acknowledgment that new information might refine our understanding of the taxonomy in the future.

The current recommended standard name is Florida pinesnake (Crother et al. 2017, p. 74). Other names include Florida pine snake and southern pine snake (Franz 1992, p. 254).

The currently accepted classification of Florida pinesnake is:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Squamata
Suborder: Serpentes
Family: Colubridae

Genus: *Pituophis*
Species: *Pituophis melanoleucus*
Subspecies: *Pituophis melanoleucus mugitus*

* (Integrated Taxonomic Information System, 2021, unpaginated)

2.2 Subspecies Description

Figure 2-1. Adult Florida pinesnakes (*Pituophis melanoleucus mugitus*) showing variation in coloration. Photo on the left was taken of an individual in the panhandle of Florida, while the individual on the right was photographed in peninsular Florida. Photos by P. Hill, Florida Fish and Wildlife Commission

Florida pinesnakes are large, heavy-bodied snakes with adult specimens typically ranging from 122 to 168 centimeters (cm) (4.0 – 5.5 feet [ft.]) with a maximum total length of 229 cm (7.5 ft.) (Ashton & Ashton 1981, p. 118; Conant & Collins 1998, p. 364; Ernst & Ernst 2003, p. 280; Enge et al. 2016, p. 7). The head is spade-shaped with a pointed snout with an enlarged rostral scale that extends upward between the internasals (Stull 1940, p. 68; Conant & Collins 1998, p. 363). These two physical characteristics make pinesnakes adept at excavating soil (Ernst & Barbour 1989, pp. 100–101). The body scales are keeled, and the anal plate is undivided (Conant & Collins 1998, p. 364; Ernst & Ernst 2003, p. 280).

The Florida pinesnake displays a variety of morphological scale patterns and coloration but the dorsum is typically gray, brown, or rusty brown on a light gray or tan to cream background (Stull 1940, p. 68; Conant & Collins 1998, p. 361; Ernst & Ernst 2003, p. 281). The contrasting of colors tends to be less pronounced near the head and the contrast between the pattern and background color becomes more robust approaching the tail (Ernst & Ernst 2003, p. 281; Miller et al. 2016, p. 1). The ventral side is white to yellow with moderate to little spotting (Tuberville & Mason 2008, p. 388). Coloration can vary regionally, with some being very pale or dark, and young Florida pinesnakes have very distinct dark patches in their dorsal pattern (Miller et al. 2016, p. 1). Overall, what is recognized as the Florida pinesnake subspecies has markings that tend to not be as sharply defined as what are currently recognized as northern pinesnake (Conant & Collins 1998, p. 364).

Pinesnakes are known for their loud hissing defensive behavior when threatened. A Florida pinesnake may coil up, inflate its body, hiss loudly and create a rattling noise by rapidly shaking its tail in dry leaves (Figure 2-2; Ashton and Ashton, 1981, p. 119; Enge et al., 2016, p. 8). To produce the loud hissing sound, exhaled air is passed over a flap of tissue that stretches across the air opening in the mouth, which will be partially open (Ashton & Ashton 1981, p. 119; Enge et al. 2016, p. 8). This visual and audible display partnered with their coloration can cause a Florida pinesnake to be misidentified as an eastern diamondback rattlesnake (Miller et al. 2016, p. 2).

Figure 2-2. Florida pinesnake displaying defensive behavior (*Pituophis melanoleucus mugitus*). Photo by Kodiak Hengstebeck

Recent data on minimum longevity for free-ranging Florida pinesnake has just been reported. A mark/recapture study at Ichauway, in Baker County Georgia recorded 12.9 years for a male and 10.1 years for a female Florida pinesnake (Smith, 2022, pers. comm). In captivity, the longevity record for the subspecies is from a subadult (1-2 years old) Florida pinesnake that was captured from the wild in 1987 and lived 32 years in captivity before succumbing to cancer (Dodd 2019, p. 519). Prior to this record, a wild-caught snake was held in captivity at Zoo Atlanta for just under 19 years prior to death (Snider & Bowler 1992, p. 29; Dodd 2019, p. 519). These reports reveal that Florida pinesnakes have a capacity for a long lifespan.

2.3 Range and Distribution

The Florida pinesnake (as currently described) is endemic to the Coastal Plain of the southeastern United States. The Florida pinesnake's recognized range spans from southeastern South Carolina, Georgia, to south Florida, and west into the Florida panhandle and into the southern part of Alabama (Figure 2-3; Franz, 1992, p. 254). In Florida, the subspecies historically occurred throughout the state except for the Everglades and the Florida Keys; museum records indicate that the distribution in south Florida was patchy and may now be extirpated from the southern portion of their range (Franz 1992, p. 254; Florida Fish and Wildlife Conservation Commission 2013, p. 4; Enge et al. 2016, p. 25).

Within the Florida panhandle, historically the subspecies' range extended west to the Escambia River. Beyond the river, within Escambia and Baldwin Counties, Alabama, it was thought that the Florida pinesnake intergraded with the black pinesnake (Franz 1992, p. 254; Mount 1975, p. 227). The black pinesnake is currently considered geographically isolated from all other eastern pinesnakes but there is evidence of introgression between *P. m. lodingi* and *P. m. mugitus* in Baldwin and Escambia Counties, 12 Alabama, and Escambia County, Florida (Conant 1956, pp. 10–11). These intergraded snakes are now separated from current populations of the black pinesnake by the extensive Tensaw-Mobile River Delta and the Alabama River. However, they are still connected to current populations of Florida pinesnake and thus we include them within the Florida pinesnake's range.

In the northeastern portion of the Florida pinesnake's range there is introgression occurring between the Florida pinesnake and the northern pinesnake where the two subspecies overlap. In the eastern portion of South Carolina, pinesnakes within the coastal plain south of the Congaree and Santee River have been considered intergrades of these two subspecies (Camper 2019, p. 184); expert judgment based on recent genetic analyses from specimens across the range recommends classifying pinesnake populations across the coastal plains of South Carolina and into southeastern North Carolina as the Florida pinesnake subspecies (Apodaca et al. 2021, entire; Nikolakis et al. 2022, entire). Therefore, we have modified the Florida pinesnake's range to include all the coastal plain of South Carolina and the southeastern coastal plain of North Carolina (Figure 2-3).

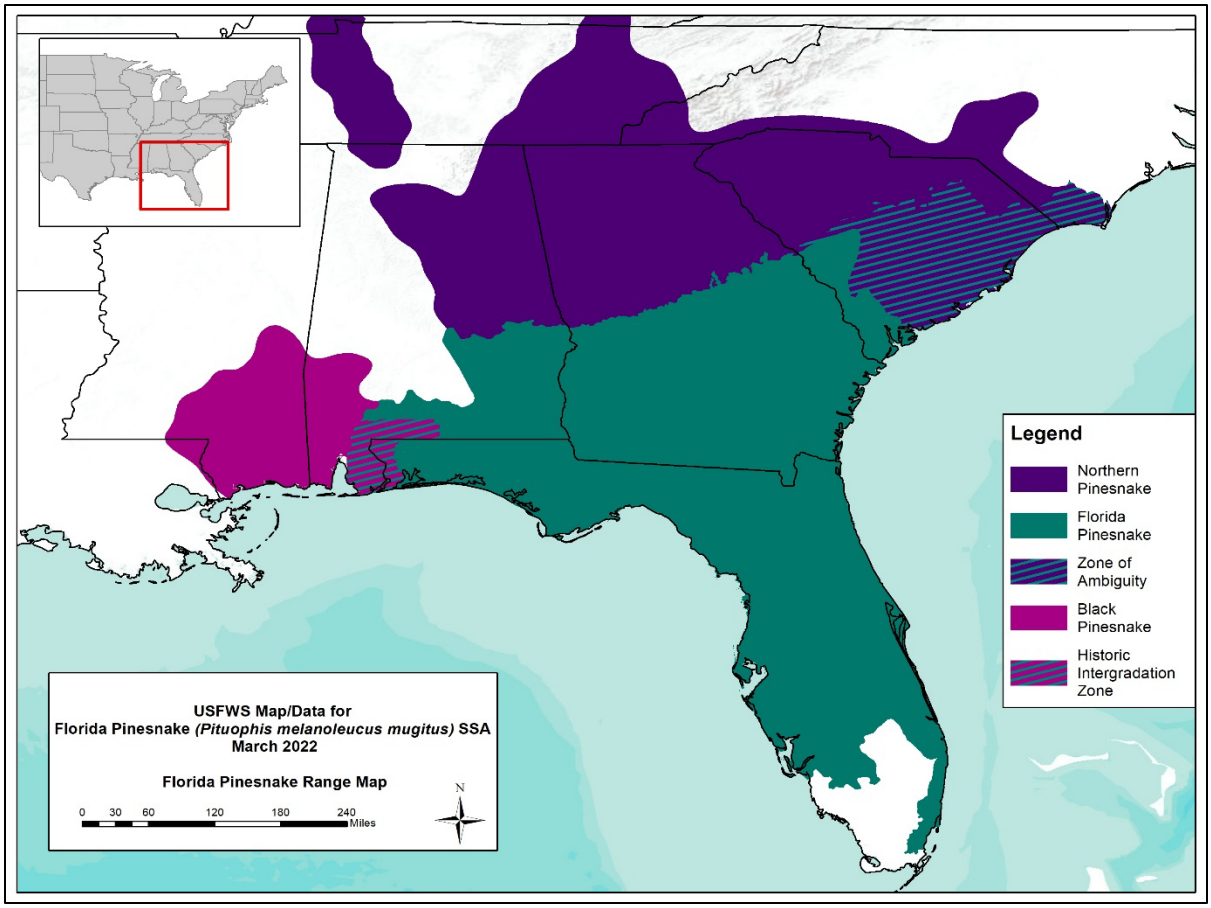

Figure 2-3. Eastern pinesnake range map, showing northern pinesnake (purple), the Florida pinesnake (teal), and the black pinesnake (pink) ranges. The most northern extent of the northern pinesnake range is not shown on the map. Also depicted, the historically considered intergrade zone between the black pinesnake and Florida pinesnake and a zone of ambiguity where recent genetic studies suggest that pinesnakes in this area are a mix of northern pinesnakes and Florida pinesnakes.

Individual Needs

2.4.1 Life History

Life history of a species includes events and characteristics that affect the likelihood that an individual will survive and contribute to the growth and persistence of a population from one year to the next. The Florida pinesnake has three life stages: egg, juvenile, and adult. Life history information for the eastern pinesnake is limited, particularly for the Florida pinesnake subspecies. Available data suggests that life history traits are similar among subspecies. We report life history information specifically for the Florida pinesnake subspecies where available and supplement that with information from those reported for the whole eastern pinesnake species.

The Florida pinesnake is diurnal and highly fossorial, spending much of its time underground. In two telemetry studies individuals were observed underground during 76-84% of observations (Franz 2005, p. 123; Miller 2008, p. 48).

In general, pinesnakes hibernate during the winter and may estivate in the summer depending on environmental conditions (Ernst & Ernst 2003, p. 282). It is possible that the Florida pinesnake is active throughout the year in the southern portion of their range. Occurrence records of Florida pinesnakes collected from cooperating agencies, organizations, and individuals (see 4.2.1), indicate that Florida pinesnakes are active across all months but are observed most often March to October with the greatest number of observations in May and June (Figure 2-4). In Florida, 50.3% of records were collected April through June, which coincides with the breeding and egg-laying season (Enge et al. 2016, p. 7). This pattern of documented records is consistent with other studies that report Florida pinesnakes as being more active above ground from March through July and secondarily in September and October (Franz 1992, p. 255; Miller 2008, p. 47).

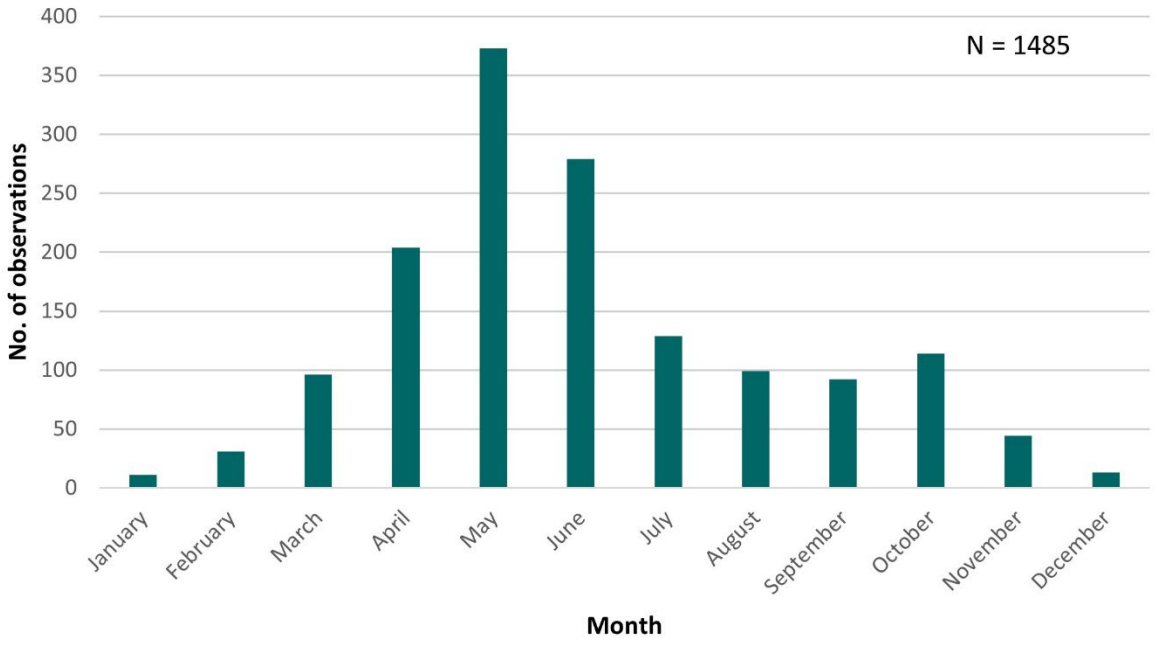

Figure 2-4. Number of Florida pinesnake records by month. There is an increase in number of observations during the months of March – June, with a seasonal peak in activity in May and then a slight increase in observations in October. Data are aggregated for entire defined range of the subspecies.

2.4.2 Foraging Ecology

Florida pinesnakes are non-venomous, powerful constrictors that consume a variety of prey. Small mammals, including rabbits, southeastern pocket gophers (*Geomys pinetis*), and other rodents make up a large portion of the pinesnake's diet but they are also known to eat ground-dwelling birds, bird eggs, and lizards (Ashton & Ashton 1981, p. 120; Ernst & Ernst 2003, p. 284; Franz 1992, p. 255; Miller 2008, p. 47). Florida pinesnakes have also been documented

consuming turtle eggs, most likely those of the gopher tortoises (*Gopherus polyphemus*; (Mortensen et al. 2019, p. 811). Pinesnakes are active foragers that hunt both above and below ground (Ernst & Ernst 2003, p. 284).

As evidenced by their pointed snout and enlarged rostral scale (the scale at the tip of their snout), they are accomplished burrowers capable of tunneling in loose soil, potentially digging nests or excavating rodents for food (Ernst & Barbour 1989, pp. 100–101). With this capability, Florida pinesnakes are adept at excavating the sand plugs of pocket gopher burrows to gain access to their tunnels (Franz 1992, p. 255; Franz 2005, p. 125). Upon encountering prey underground, pinesnakes are believed to constrict their prey by pressing them against tunnel walls (Hisaw & Gloyd 1926, p. 203; Ernst & Ernst 2003, p. 284; Miller 2008, p. 12).

2.4.3 Reproduction and Sex Ratios

For the Florida pinesnake, both sexes become sexually mature when they reach a snout-vent length (SVL) of ~ 90 cm (3 ft.), which can be three years (Franz 1992, p. 254; Ernst & Ernst 2003, p. 282; Enge et al. 2016, p. 7). In spring and early summer, male Florida pinesnakes search large areas for reproductive opportunities (Ernst & Barbour 1989, p. 4; Miller 2008, p. 26; Miller et al. 2012, p. 6). Upon finding a mate, the male engages in a courtship ritual by holding the female's neck in his jaws, coiling around her, and rubbing her body with his body (Ernst & Ernst 2003, p. 283; Miller et al. 2016, p. 4). This courtship and subsequent copulation can last for hours (Ernst & Ernst 2003, p. 283).

Courtship and mating occur April through May with female pinesnakes laying eggs May through July in underground burrows (Ernst & Ernst 2003, p. 283). It is possible that timing of courtship and mating can vary based on latitude, but this is not currently documented in the literature. It is unknown how frequently a female Florida pinesnake can reproduce (ex. annually or skip years) and it is presumed that a female can only produce one clutch in a breeding year. Florida pinesnakes excavate cavities for nests and sometimes do so within existing burrows of other animals, such as southeastern pocket gophers, gopher tortoises, and nine-banded armadillos (*Dasypus novemcinctus*; (Stevenson 2017, p. 680; Holland et al. 2019, p. 398). In captivity, clutch sizes have been reported containing 4-8 very large eggs (Lee 1967, p. 241). More recently, wild nests have been excavated containing 5-7 eggs (Stevenson 2017, p. 680; Holland et al. 2019, p. 398). Florida pinesnake eggs are ovular, leathery, lightly adhere to one another and have light pigmentation ranging from cream to white (Lee 1967, p. 242; Ernst & Barbour 1989, p. 101).

Figure 2-5. Florida pinesnake eggs with hatchling emerging hatchling emerging in left photo. Hatching Florida pinesnakes in right photo. Photo K. Enge, Florida Fish and Wildlife Conservation Commission (right).

Hatchlings emerge from eggs from August through October after an approximate 67-75 day incubation period (Figure 2-5; (Franz 1992, p. 255; Ernst & Ernst 2003, p. 283). Hatchlings resemble adults, but they are duller in color. Hatchling total body length ranges from 33.0 to 55.5 cm (13 to 21.9 in.; (Ernst & Ernst 2003, p. 283; Stevenson 2017, p. 680; Holland et al. 2019, p. 399) and are large compared to other North American snakes. Hatchling sex ratios are believed to generally favor males (Ernst & Ernst 2003, p. 284). Recent reports indicate variable hatchling sex ratios between two clutches collected from the wild (one clutch had a ratio of 3 males:5 females; the second clutch had 4 males: 0 females; (Holland et al. 2019, p. 399).

2.4.4 Individual Subspecies Needs (Habitat)

The Florida pinesnake is commonly associated with the longleaf pine ecosystem that once covered a large portion of the coastal plain in the southeastern United States. They occupy xeric (dry), upland habitat with well-drained, sandy soils, characterized by pine-dominated or pine-oak woodland. Pinesnakes can also be found in pine flatwoods, oak scrub, dry oak forests, old fields (abandoned agricultural or pasture land) and other ruderal habitats, particularly if southeastern pocket gophers are present (Enge et al. 2016, p. 25). Florida pinesnakes exhibit a strong preference for pine forests with open-canopy and frequent fire (Franz 2005, p. 129; Miller et al. 2012, p. 710; Howze & Smith 2021, p. 4). In a mature mesic (moderate amounts of moisture) longleaf pine forest in southwestern Georgia, Florida pinesnakes most often used stands with a pine basal area of 5.92 (± 4.11) m^2/ha, herbaceous ground cover of 32.9% (±22.8), canopy cover of 56.8% (±25.1), and shrub cover of 9.96% (±9.66) (McIntyre et al. 2019, p. 287). A range-wide habitat suitability analysis showed that suitable habitat and likely subspecies presence was strongly and positively associated with well-drained, sandy soil, compatible land cover (i.e., land classified as evergreen or mixed forest, scrub/shrub, grasslands, or similar land cover types), topographic position index (i.e., locally elevated patches, such as sandhills), and fire frequency (Crawford et al. 2020a, pp. 9, Supplemental Text S1, Table S2c [p. 13], Figure S2c [p. 23]).

Florida pinesnakes have the capability to excavate their own burrows and opportunistically use existing underground burrows and tunnels created by other species. They are most often reported using southeastern pocket gopher burrows in Florida and Georgia (Franz 2005, p. 130; Miller et al. 2012, p. 709). Outside the range of the southeastern pocket gopher and to a lesser extent within the range, Florida pinesnakes will use stump holes, nine-banded armadillo burrows, mouse burrows, gopher tortoise burrows, and various other refugia (Miller et al. 2012, p. 709). In a study from Georgia within fire-maintained longleaf pine forests, Florida pinesnakes were found in both gopher tortoise burrows and stump holes, however they were more frequently associated with gopher tortoise burrows (Murphy et al. 2021, p. 5). Florida pinesnakes are also highly associated with southeastern pocket gophers and pocket gopher habitat. Southeastern pocket gopher burrows provide shelter and gophers serve as a source of prey for Florida pinesnakes. Densities of pocket gopher mounds in natural pine and mixed pine-hardwoods have been documented to be four times higher than in hardwoods or pine plantations; however, pinesnakes within the study area did not select the mixed pine-hardwood habitat suggesting that other habitat factors are important (Miller et al. 2012, p. 711). The Florida pinesnake tends to select southeastern pocket gopher tunnel systems over gopher tortoise burrows as overwinter sites as well (Franz 2005, p. 130) and they also demonstrate some refugia site fidelity. A study that tracked two individuals for two consecutive fall seasons found that though they did not use the same refugia between the years, they did overwinter within 335 meters (0.2 mi.) of the previous year's site (Miller 2008, p. 48). Additionally, another study reported that after an early springtime drop in temperatures several Florida pinesnakes returned to the same pocket gopher burrows that had served as their overwinter refugia one month earlier (Orton et al. 2018, p. 762). When temperatures increased, site fidelity weakened but when temperatures began to drop again the following fall, several tracked snakes relocated to a previously used burrow (Orton et al. 2018, p. 762).

In two telemetry studies, the mean home range size for the Florida pinesnake was 57 hectares (ha) (141 acres [ac]) and 59.2 ha (146 ac) (Franz 2005, p. 121; Miller et al. 2012, p. 709). One of the studies also reported home range sizes of 18.6 – 156.8 ha (46 – 387.5 ac) (Miller et al. 2012, p. 709). No significant difference was found between male and female annual ranges; however, male spring seasonal range was larger than that of females, which is attributed to males searching for mates at that time of year (Miller et al. 2012, pp. 708, 710).

It has also been reported, that Florida pinesnakes infrequently cross paved and graded dirt roads but will cross harrowed dirt roads (Miller et al. 2012, p. 709) suggesting roads may influence movement patterns and decrease habitat suitability for the subspecies. When observed above ground, the Florida pinesnake is more frequently found in areas with higher shrub and woody vine coverage and less bare ground (Hayman 2019, pp. 6–7; Miller et al. 2012, p. 709). It is speculated these conditions provide good cover and basking opportunities for the snakes.

The maximum annual movement or dispersal distance for the Florida pinesnake has been reported as 4.26 kilometers (km) (2.65 miles [mi]) with the total mean annual distance per snake as 1.54 – 4.26 km (0.96 - 2.65 mi) from April through August (Franz 2005, p. 124).

We used existing life history literature and expert judgment to identify specific ecological needs for individuals to survive and reproduce (Figure 2-6; as well as factors influencing viability that are discussed in Chapter 3). Five main habitat elements appear to be essential to the survival and reproductive success of individuals: well-drained soils, suitable vegetation structure and composition, nearby road density, fire return interval, and presence of prey.

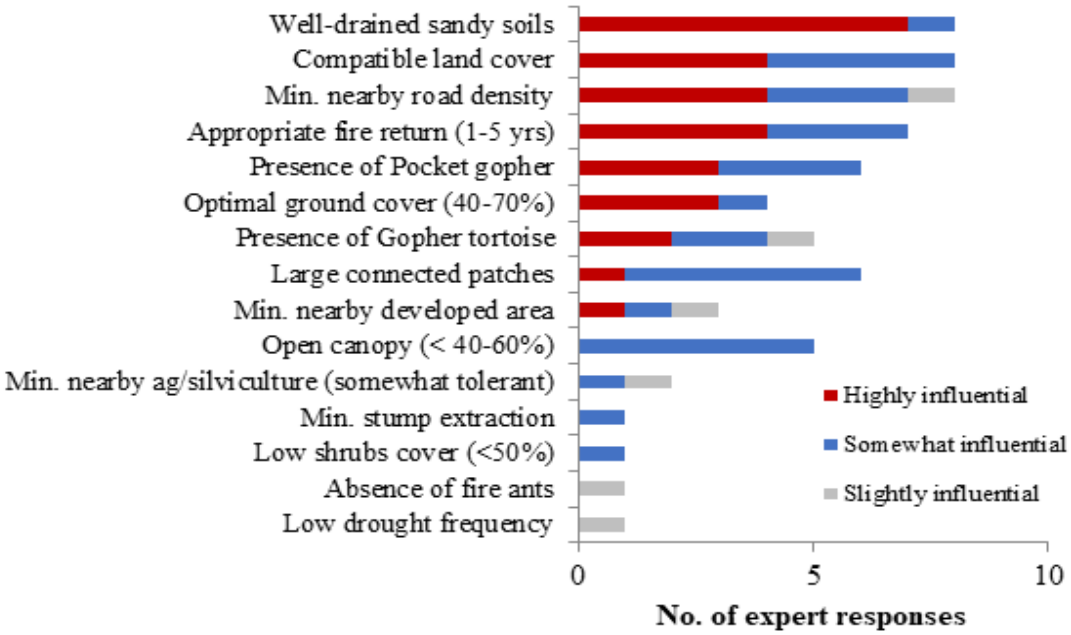

Figure 2-6. Influential habitat, landscape, and biophysical attributes for Florida pinesnake presence at a site, as identified by 9 species experts. Some attributes reflect species needs while others reflect factors influencing viability (discussed in Chapter 3). Experts were asked to list attributes they associate with ideal habitat for the species. Definitions for habitat rankings: Highly – attributes must occur at a site for the species to be present; Somewhat – attributes occurring on the landscape greatly increase the likelihood of species being present, but species may occasionally use landscapes without these attributes; Slightly – attributes occurring on the landscape slightly or variably increase the likelihood of species being present, but species may use landscapes without these attributes. Source: (Crawford et al. 2020a, p. Supplemental Text S1, Figure S1c, Page 19)

CHAPTER 3 – FACTORS INFLUENCING VIABILITY

The following discussion provides a summary of the factors that are affecting or could be affecting the current and future condition of the Florida pinesnake throughout some or all its range. Risks that are not known or not suspected to have effects on Florida pinesnake populations are not discussed in this SSA Report.

3.1 Habitat Loss, Conversion, and Fragmentation

Loss of the Longleaf Ecosystem

The Florida pinesnake is associated with longleaf pine and other open pine ecosystems, particularly xeric uplands that were historically maintained by fire. These pine ecosystems are fire-dependent and once dominated the Coastal Plain of the Atlantic and Gulf coast regions, from Virginia to Texas (Ware et al. 1993, p. 447). The longleaf pine uplands once covered an estimated 92 million acres (Frost 1993, p. 20). By the 21st century, the longleaf pine community had declined to less than three million acres due to forest clearing and conversion for agriculture, silviculture, and development (Jensen et al. 2008, p. 16; Landers et al. 1995, p. 39). Little old-growth longleaf remains, and of the uplands that remain, only about 3% are in relatively natural condition due to the exclusion/suppression of naturally occurring wildfires (Frost 1993, p. 17; Simberloff 1993, p. 3).

Original longleaf pine communities were composed of multi-aged stands, were open-canopied, with sparse but diverse hardwood trees (e.g., turkey oak [*Quercus laevis*], southern red oak [*Q. falcata*]), and contained a structure of two layers: canopy and diverse herbaceous groundcover. Fire tolerant hardwood midstory and shrubs were relatively sparse. Frequently burned, the natural condition was a canopy cover that rarely exceeded 60% and permitted a grassy groundcover to flourish (Noss 2013, p. 9). In contrast, much of today's pine forests are young, dense, and dominated by slash pine (*Pinus elliottii*) or loblolly pine (*P. taeda*), with a substantial hardwood component and little or no herbaceous groundcover (Noel et al. 1998, pp. 534–535). These pine species do not produce the high-quality resinous stumps and deep taproots that longleaf pine does, likely making longleaf pine more important to fossorial species that rely on stumpholes for refugia, such as the Florida pinesnake (Means 2005a, pp. 76–78). Commercial harvest of pine stumps is occurring in parts of the southeast and a recent study of vertebrate use of pine stump holes found that Florida pinesnakes use these as refugia (Murphy et al. 2021, p. 5), but to a lesser extent than other refugia (Miller et al. 2012, p. 706).

The longleaf pine ecosystem was first heavily altered by exploitation for naval stores and then virtually eliminated by widespread logging (Frost 1993, pp. 24–37). Naval stores industries harvested pine resin to produce tar, pitch, and turpentine—commodities in high demand during colonial times. Pine woodlands were logged for lumber and converted to agricultural fields. Impacts to easily accessible areas began with the arrival of Europeans, but technological developments of the 1800s, such as the copper still, steam power, and especially railroads,

dramatically increased the rate and area of loss (Frost 1993, pp. 23–24). In the late 1800s, logging operations moved to the previously inaccessible interior forests of longleaf, shortleaf (*P. echinata*), and loblolly pines. This especially intense period of logging from 1870 to 1930 resulted in the loss of nearly all of the old-growth pine forest in the Southeast.

Historically, natural disturbances created and maintained open pine conditions throughout the southeast and fire was the predominant disturbance in the longleaf ecosystem. Although there is uncertainty in burn regimes between various habitat types and along environmental gradients, fire frequencies in longleaf pine ecosystem have been estimated at one to three years (Frost 1998, p. 76). Frequent burning has been shown to maintain species richness of the ground cover layer (Glitzenstein et al. 2003, p. 23). Season of fire has important effects on vegetation composition. Growing season burns, generally defined as the period of time when most plants are actively growing, coincides with the bulk of wildfires caused by lightning strikes, and has been shown to better meet management objectives for longleaf pine forests (Knapp et al. 2009, p. 3).

Historically, lightning was the primary ignition source shaping the evolution of these fire-maintained ecosystems, but Native Americans played a role in maintaining them and it is difficult to separate the anthropogenic effects verses the effects of lightning on the vegetative community (Frost 1993, p. 34; Van Lear et al. 2005, p. 151). After European settlement and prior to the mid-1800s, farmers burned the woodlands regularly to improve forage for free-ranging livestock, but by the mid to late 1800s fencing of livestock caused a decrease in burning (Frost 1993, p. 34). Although many people continued to use fire in agricultural fields well into the 1900s, the rise of mechanical and chemical agriculture replaced fire-based agricultural methods.

Active fire exclusion/suppression began to be institutionalized in the southeastern United States between 1910 and 1930 (Frost 1993, p. 35). Some foresters and landowners denounced fire as detrimental rather than an integral or useful component of the natural system. Fire suppression increased with the rise of pine plantations, a land use that began in the 1930s and continues today (Frost 1993, p. 36). Due to the suppression of fire and the natural disturbance fires provide, longleaf pine communities have converted to fire-intolerant trees and shrubs that eventually shade out the ground cover and render the forest unsuitable for much of the fire-adapted biota (Van Lear et al. 2005, p. 155). Closed-canopied habitats which result from fire suppression are unsuitable for the Florida pinesnake (Enge et al. 2016, p. 26).

Timber harvesting is one of the most prominent disturbances that shapes plant and animal communities in forests, and the southeastern United States is the leading timber-producing region in the country (Prestemon & Abt 2002, p. 299). Many open pine forests in the Southeast have been replaced by pine plantations or mixed pine hardwood stands, often harvested on short rotation intervals (15-30 year). Short-rotation planted pine forests differ from the natural stands in a variety of ways, including higher stand densities, closed canopies, deep litter beds, and sparse understories, and harvesting of the short-rotation planted pine is done primarily through clear cutting (Means 2005b, pp. 142–143; Todd & Andrews 2008, p. 754). Many reptile species

have been shown to decline in abundance over time following the clearing of primary forest or conversion to plantation forest (Glor et al. 2001, p. 719; Kanowski et al. 2006, pp. 13–14).

Dense closed canopy forests with little to no groundcover do not provide adequate habitat for the Florida pinesnake but thinning dense stands of forest can benefit species that occupy longleaf pine forests. Timber harvesting, including thinning of dense stands can help mimic the open historical, natural condition of longleaf stands. Many forests are so densely stocked that thinning is required before a prescribed burning regime can be effective. A wide range of silvicultural practices influence canopy conditions. Even-aged regeneration harvests often used in pine management provide abundant sunlight to stimulate groundcover establishment and growth. However, benefits are ephemeral as reforested areas grow and develop closed canopy conditions that shade groundcover (Greene et al. 2019, p. 205). Subsequent treatments including precommercial and commercial thinning can re-establish open pine conditions.

Heavy equipment is routinely used in the management of public and private lands. Heavy equipment is utilized in activities such as site preparation, reforestation, restoration, prescribed fire, herbicide applications, and harvest operations. The impacts of heavy equipment on Florida pinesnake at the subspecies level is not known. Mechanical site preparation techniques can directly kill or injure fossorial herpetofauna by damaging the soil structures in which those species occur and making the habitat less suitable (Bailey et al. 2006, p. 45). Mechanical site-preparation can lead to a potential loss of underground refugia that might otherwise be left following logging in pine plantations (Means 2005a, p. 78).

With the loss of the longleaf pine ecosystem has come declines in southeastern pocket gopher populations as well and the southeastern pocket gopher is now considered a species of conservation concern throughout its range (Parsons 2019, p. 3). The loss of underground refugia associated with the decline in southeastern pocket gophers and gopher tortoise populations, has likely had an impact on Florida pinesnake populations (Enge et al. 2016, p. 26).

Urban Development

Urbanization plays both direct and indirect roles in the decline of many species (McKinney 2002, p. 883) and impacts many wildlife species from direct loss of habitat, fragmentation of habitat, increased road mortality, increased human persecution, and the increase in domestic predators, such as cats (*Felis catus*) and/or dogs (*Canis lupus familiaris*). Urbanization fragments and replaces natural habitats with artificial structures, impervious concrete and asphalt surfaces, manicured lawns, gardens full of exotic plant species, and increases levels of air, water, noise, and light pollution, putting the survival of many wildlife species in jeopardy (Sutherland 2009, p. 35). Snakes seem to be particularly sensitive to effects of urbanization, and this intolerance has played a key role in the general declines reported for snake species around the world (Gibbons et al., 2000, entire; Andrews & Gibbons, 2005, entire; Row et al., 2007, entire; Sutherland, 2009, p. 52).

Prescribed fire has generally been accepted as a primary habitat management tool; however, increased urbanization limits its use in many locations. Urban sprawl can fragment habitat and, in many areas, complicates the logistics of performing adequate and seasonally appropriate burns, further straining staff and budget resources. A combination of urban sprawl and migration of humans to rural areas has created an extensive wildland-urban interface (WUI), the area where houses and wildland vegetation meet or intermingle. Wildland-urban interfaces limit the ability to conduct prescribed fires due to issues associated with smoke management and fear of fires escaping and having catastrophic effects. Human health and safety issues increasingly complicate fire management as well.

Active fire exclusion/suppression and a lack of a controlled burning program in WUIs results in increased fuel loads and a subsequent increase in the likelihood of future destructive fires (Winter et al. 2002, p. 15). Furthermore, because of constraints on implementing prescribed fire in WUIs, many areas of suitable or restorable sites remain unburned each year and without other habitat management there is an increased risk of habitat degrading to the point that it is unsuitable for Florida pinesnakes due to woody species encroachment.

In the southeastern United States, the urban footprint is projected to greatly increase over the next 50 years, with median projections showing that the amount of land in urban areas will increase by 139% by 2060 (Terando et al. 2014, p. 4). Urbanization is not predicted to be uniform across the region. The largest urban expansions are projected in the Blue Ridge, Ridge and Valley, Southern Coastal Plain, and Piedmont ecoregions; and new urban centers are projected in the Appalachian Mountains and central Florida regions (Terando et al. 2014, p. 6).

Many "hotspots" of projected urban development are predicted to occur within, or near known occurrence records for Florida pinesnakes, or predicted suitable habitat. Although we do not know the exact response of the Florida pinesnake to various levels of urbanization, we do know that urbanization will likely result in the loss, degradation, and fragmentation of habitat, increased amount of WUIs, increased persecution by humans, increased road mortality, and increases in mesopredators, such as raccoons (*Procyon lotor*) and domestic predators. Thus, we consider urbanization to be a significant threat to the Florida pinesnake in areas where urbanization might occur.

Fragmentation

Human-induced disturbances, particularly from land use changes discussed in the previous sections, not only have the potential to result in loss or degradation of habitat, but also fragmentation of habitat. Habitat fragmentation is the breaking apart of continuous habitat into multiple patches (Fahrig 2003, p. 509). Development increases the prevalence of roads and associated infrastructure, which increase the fragmentation of the habitat and additionally result in the potential for increased mortality from vehicular traffic. Fragmentation can have a variety of negative impacts on wildlife, including greater mortality rates associated with landscape

modifications (e.g., roads), more frequent encounters with humans, reduced resources in smaller patches, reduced reproduction, restricted gene flow, and increases in predation and competition (Wiens 1994, p. S97; Kjoss & Litvaitis 2001, p. 285). Reduction of larger habitat patches into smaller patches can lead to population declines due to limited resource availability and can also negatively affect day-to-day movement (Barbour & Litvaitis 1993, p. 326). Fragmentation may also negatively affect larger-scale movements such as dispersal and seasonal migration.

Many snake species are likely to be sensitive to habitat fragmentation because they occur in low densities, have limited dispersal abilities, have thermal constraints, and are subject to direct and indirect mortality caused by humans (Kjoss & Litvaitis 2001, p. 286; Webb & Shine 1997, p. 213). Of particular concern is the role that roadways play in fragmenting habitat. Depending on the size and traffic volume, newly constructed roads can effectively become barriers that divide and isolate populations (Roe et al. 2006, p. 162). The increasing encroachment of roads into natural areas may isolate populations, prevent movement between nest sites and hibernacula, restrict gene flow, and limit access to mates (Vanek & Wasko 2017, p. 115). Additionally, roads not only impinge on life history requirements of species, but also facilitate other threats, such as conversion of more habitat, creation of roads for access, and the spread of invasive species (Forman & Alexander 1998, pp. 221–222), and lead to increases in direct mortality, discussed in the next section.

Data are unavailable on the impact of habitat fragmentation on the Florida pinesnake, however, pinesnakes tracked in southwestern Georgia avoided crossing graded and paved roads (Miller et al. 2012, p. 709) and a modeling study determined that fragmentation by roads may be an impediment to maintaining viable populations of pinesnakes (Steen et al. 2012, p. 1092). Habitat fragmentation and the associated road mortality is considered a leading threat to Florida pinesnake populations.

3.2 Road Mortality

As discussed above, roads cause habitat fragmentation and pose a barrier to movement that can isolate populations, and roads increase direct mortality for many snake species (Andrews & Gibbons 2005, p. 772). Snakes can be more severely affected by road mortality than some other animal groups because they might use roads for thermoregulation, are relatively slow-moving, some will remain immobile on roads in response to oncoming vehicles, and are often intentionally hit by drivers (Andrews & Gibbons 2005, p. 778; Bonnet et al. 1999, p. 40; Rosen & Lowe 1994, p. 143). It has been observed that reptiles are struck by vehicles at a greater rate than would be expected by chance, suggesting that drivers intentionally target reptiles on roads (Ashley et al. 2007, p. 137). It has been estimated that vehicular traffic has killed tens or hundreds of millions of snakes in the United States since the start of motorized transportation (Rosen & Lowe 1994, p. 147).

An increase in the number of mortalities from vehicles may result in reduced gene flow among populations, decreased potential for dispersal into fragmented habitats, and altered demographics in the form of lower survival and immigration rates – all of which can lead to declines or extirpation of populations of snakes. One study of black ratsnakes (*Pantherophis obsoletus*) found that, when including an estimate of road mortality within a population viability analysis, the estimated extinction probability increased from 7.3% to 99% over 500 years (Row et al. 2007, p. 117). An increased mortality rate for different reproductive classes can have profound consequences for a population; for example, increased mortality of females can reduce a population's growth rate more than mortality of males (Row et al. 2007, p. 118).

The highest mortality on roads among snakes usually occurs among adult males during the mating season, neonates or hatchlings immediately after birth or hatching, and adult females during egg laying migrations (Bonnet et al. 1999, p. 47). Roads that bisect high quality habitat have higher levels of mortality than those that bisect lower quality habitat (Shepard et al. 2008, p. 357). Snake populations could experience especially high levels of road mortality during periods where high traffic volumes and species' seasonal movements coincide (Ashley et al. 2007, p. 141).

Varying evidence exists as to whether Florida pinesnakes show a behavioral aversion to crossing paved or graded roads. Two studies provide moderate to weak evidence that Florida pinesnakes exhibit a behavioral aversion to crossing roads. During a telemetry study in southwestern Georgia, Florida pinesnakes did not cross paved roads that abutted their home ranges and infrequently crossed graded roads; however, they were frequently observed crossing harrowed dirt roads (Miller et al. 2012, p. 711). Similarly, in a study in northern Florida, Florida pinesnakes crossed dirt roads less often than would be expected; however, in the case of this study, the dirt roads at the study sites often divided more suitable habitat from less suitable habitat and that may have explained the apparent aversion to crossing roads (Hayman 2019, p. 9). Additionally, that study observed radio-tracked male and female Florida pinesnakes did cross a moderately high-traffic road when suitable sandhill habitat was on either side, and we also know that Florida pinesnakes cross roads because many of the occurrence records are from encounters on roads (Figure 3-1). Through 2016, Florida Fish and Wildlife Conservation Commission alone documented 108 Florida pinesnake records on roads, of which 41% were animals that had been killed (Enge et al. 2016, p. 27). We do not know the full impact that road mortality has on this subspecies, but the high number of observed Dead on Roads (DORs) provides evidence that road mortality is occurring. Pinesnakes may be vulnerable to road mortality due to their large size, extensive movements, diurnal activity, and relatively slow speed (Gibbons & Dorcas 2005, p. 112; Rudolph et al. 1999, p. 129).

Figure 3-1. Florida pinesnake crossing a dirt road in Florida (upper left) and a DOR Florida pinesnake on paved road (lower right). Photos K. Enge (upper left) and P. Hill (lower right), Florida Fish and Wildlife Conservation Commission.

3.3 Invasive Species

3.3.1 Red Imported Fire Ants

Negative impacts on wildlife associated with documented introductions of non-native species are increasing, but the long-term consequences of many introductions are still poorly known (Langkilde 2009, p. 208). The red imported fire ant (RIFA; *Solenopsis invicta*), originating from South America, was first introduced as early as 1918 to the United States at the port of Mobile, Alabama, and subsequently spread from there (Wilson 1951, p. 68). The current range and predicted expansion of the red imported fire ant across the United States is shown in Figure 3-2 (U.S. Department of Agriculture 2017, unpaginated). Most importantly the red imported fire ant occurs throughout the entirety of the Florida pinesnake's range.

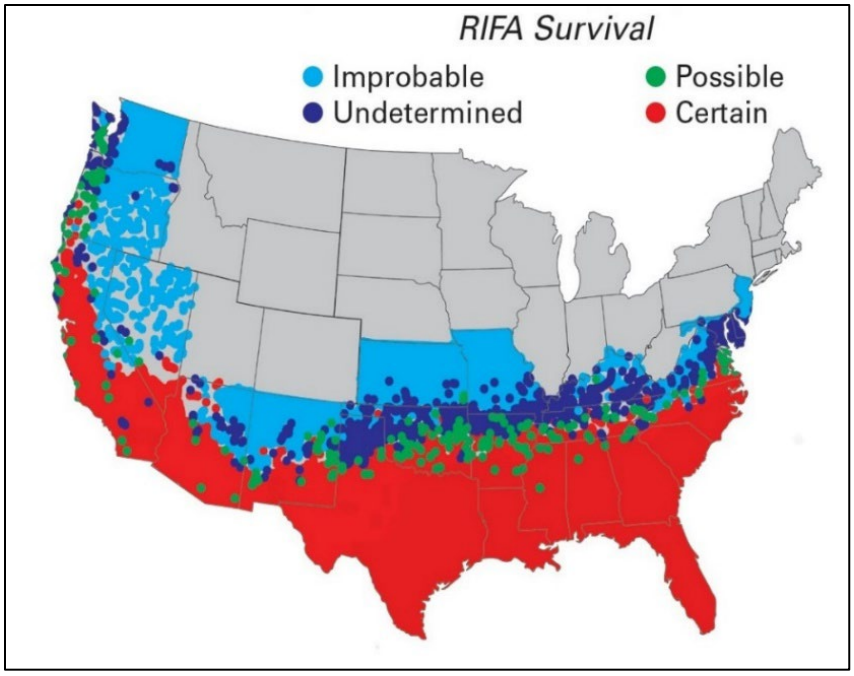

Figure 3-2. Current range and predicted range expansion of the red imported fire ant (RIFA; *Solenopsis invicta*) across the contiguous United States. Survival of RIFA is certain in the red area, possible in the green, undetermined in the dark blue, and improbable in the light blue area. Source: U.S. Department of Agriculture, 2017, unpaginated.

Red imported fire ant abundance can quickly increase, and infiltrate disturbed and early-successional habitats (Todd et al. 2008, p. 540). As a result, red imported fire ants can become the dominant ant in mature longleaf pine forests (Stuble et al. 2011, p. 620). Reptiles are particularly susceptible to red imported fire ants. Many species of reptiles are oviparous (egg-laying), and their eggs and young snakes can be depredated by red imported fire ants, both pre- and post-hatching (Diffie et al. 2010, entire). In addition, many reptile species inhabit disturbed areas or create disturbed soils with nest excavation, that red imported fire ants prefer (Darracq et al. 2017, p. 2). Species that nest under or on the ground and in or near open habitat are regularly preyed upon by red imported fire ants (Todd et al. 2008, p. 544). Red imported fire ants are aggressive, and their stings (which are actually venomous bites) can result in direct mortality, and their presence can also reduce survival indirectly by reducing food availability and altering behaviors including foraging patterns (Wilcox & Giuliano 2014, pp. 3–4).

In a study of gopher tortoise burrows in southeast Florida, red imported fire ants were present at most burrows (Wetterer & Moore 2005, p. 352) and would attack tortoise hatchlings in the nest prior to emergence (Dziadzio et al. 2016, p. 531). Red imported fire ants have also been linked to population declines of several other native species, including the Houston toad, (*Bufo houstonensis*; (Brown et al. 2012, entire), bobwhite quail (*Colinus virginianus*; (Allen et al. 2000, entire), Texas horned lizard (*Phrynosoma cornutum*; (Price 1990, p. 469.4), and have been proposed to be contributing to the decline of the southern hognose snake (*Heterodon simus*;

(Tuberville et al. 2000, pp. 33–34). The imported fire ant occurs throughout the entirety of the Florida pinesnake's range, however, the impact that red imported fire ants have on Florida pinesnake populations has not been directly assessed.

3.3.2 Feral Hogs

Feral hogs (*Sus scrofa*) negatively affect almost all aspects of ecosystem structure and function (Jolley et al. 2010, p. 519) and are known to have significant impacts on native animal and plant communities through direct consumption and indirectly through rooting and soil disturbance (Barrios-Garcia & Ballari 2012, pp. 2284–2293). Reptile species are particularly susceptible to impacts from feral hogs (Taylor & Hellgren 1997, p. 38). In addition to causing direct mortality to reptiles, feral hogs also have indirect effects on populations through rooting and habitat alteration (Jolley et al. 2010, p. 520). A study at Fort Benning, Georgia estimated that the population of feral hogs (i.e., estimated to be 3,196 individuals) could consume 3.16 million reptiles and amphibians per year (Jolley et al. 2010, p. 521), though we caution that those projections were based a single observed specimen for several taxa and did not include any observations of larger bodied reptiles. Currently, the highest relative abundance of feral hogs occurs in central California and the southeastern United States and broadly overlaps the current range of the Florida pinesnake (Figure 3-3; (McClure et al. 2015, pp. 11, 17). The most likely impacts of feral hogs on Florida pinesnakes would be through hog disturbance to soils, and destruction of nests, but at present there have been no assessments of the direct or indirect effects of feral hog on Florida pinesnakes.

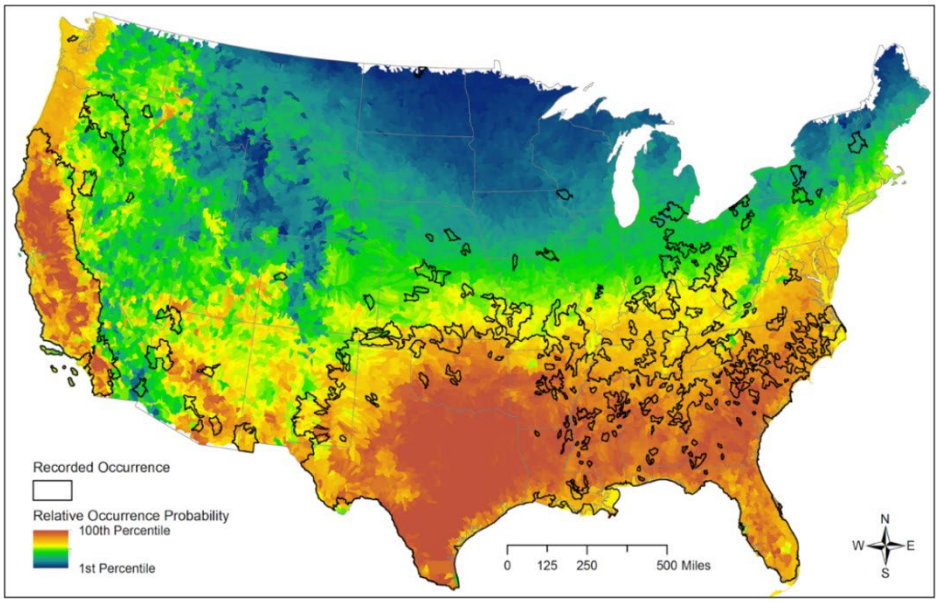

Figure 3-3. Current and predicted range expansion of feral hogs across the contiguous United States. Current range (recorded occurrence) of feral hogs from 1982 to 2012 is outlined in black. Predicted feral hog occurrence is displayed in color based on probability. Source: McClure, et al., 2015, p. 11.

3.3.3 Cogongrass

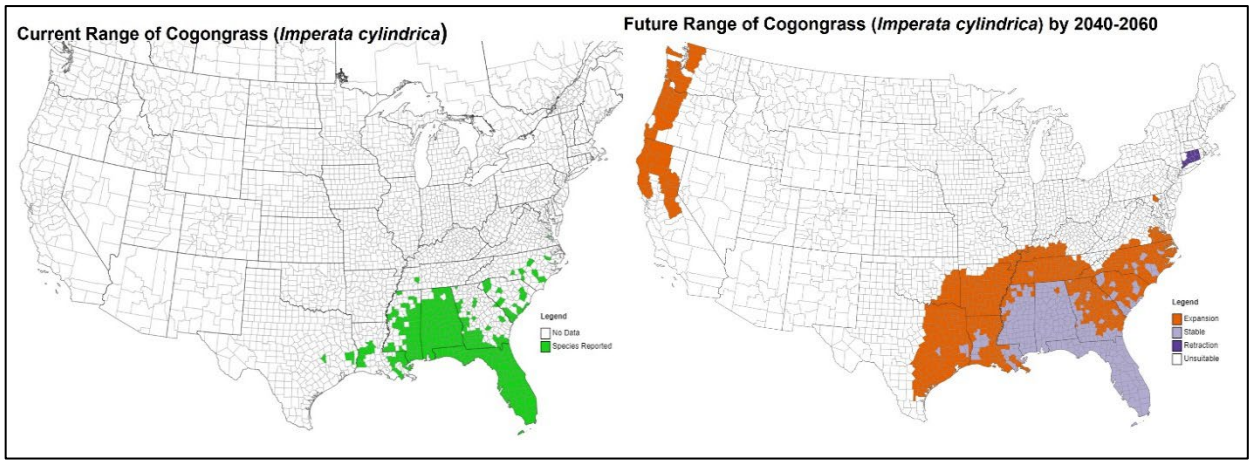

Figure 3-4. Current and future range of cogongrass (*Imperata cylindrica*) within the United States. Current range is in green in the left map and future range is on the right map with purple indicating stable and orange showing expansion. Source: (EDDMapS 2021, unpaginated)

Cogongrass (*Imperata cylindrica*), was introduced into the U.S. as a forage crop and soil stabilizer in the early part of the 20th Century, is well adapted to frequent burning and full sunlight, and now considered one of the worst invasive weeds in the world (Holzmueller & Jose 2011, p. 436). Cogongrass is known to impact longleaf pine ecosystems in the southeast, already occurs throughout much of the Florida pinesnake range, and is predicted to continue to expand (Figure 3-4). Cogongrass can rapidly spread in disturbed areas and those undergoing habitat management and restoration. Unlike other undesirable species in xeric upland communities, cogongrass is well adapted to fire, and may rapidly spread following a disturbance in an ecosystem, such as a prescribed fire (Holzmueller & Jose 2011, pp. 436–437). Once established it is very difficult to eradicate and requires aggressive control with herbicides (Yager et al. 2010, pp. 229–230). Cogongrass displaces native grasses and forms thick dense stands, that decrease native species biodiversity (Holzmueller & Jose 2011, p. 436).

When cogongrass invades an area, it can quickly result in habitat loss for many of the longleaf pine ecosystem associated species, such as gopher tortoise that will not use invaded areas or consume cogongrass (Basiotis 2007, p. 21). The invasive weed can also eliminate much of the food resources and foraging areas for rodents that are the primary prey species of Florida pinesnakes. Because it is fire adapted, cogongrass has additional impacts on the use of prescribed fire by altering fire regimes via increased fuel loads. This effect of cogongrass on fire behavior has the potential to increase direct Florida pinesnake mortality, reduce plant diversity and forage for keystone species and some ecosystem engineers (e.g., Gopher tortoise), and facilitate other invasive plant species (Basiotis 2007, p. 24; Lippincott 1997, pp. 48–65). While the effects of cogongrass on Florida pinesnakes or other snake communities have not been assessed, cogongrass is currently the most likely invasive plant that could negatively affect Florida pinesnake habitat suitability and populations.

3.4 Climate Change

In the southeastern United States, climate change is expected to result in more frequent droughts, more extreme heat (resulting in increases in air and water temperatures), increased heavy precipitation events (e.g., flooding), more intense storms (e.g., frequency of major hurricanes increases), and rising sea level and accompanying storm surges (Intergovernmental Panel Climate Change [IPCC] 2013, entire). Warming in the Southeast is expected to be greatest in the summer, which is predicted to increase drought frequency. Mean annual precipitation is expected to increase slightly, but the precipitation patterns are expected to become more variable with a shift to more extreme periods of high rainfall, which could lead to increased flooding events, and more frequent and intense droughts (IPCC 2013, entire). Changes in climate may affect ecosystem processes and communities by altering the abiotic conditions experienced by biotic assemblages resulting in potential effects on community composition and individual species interactions (DeWan et al. 2010, p. 7). There is uncertainty about how the ecosystems and species in this region will respond to the shifting climate, and effects on species of conservation concern may result from yet undetermined synergistic effects. Climate change may act as a risk multiplier by increasing the risk and severity of more imminent threats such as urbanization or altered fire regimes.

Terrestrial ectotherms, such as the Florida pinesnake, may be at direct risk from climate change because they are less effective at physiologically buffering body temperature changes against changes in ambient temperature. Instead, pinesnakes rely on ambient thermal heterogeneity to regulate their temperature behaviorally. The ability to optimally regulate body temperatures by moving among diverse microhabitats affects their growth, locomotion, and reproduction (Aubret & Shine 2010, p. 246; Deutsch et al. 2008, p. 6668; Kearney et al. 2009, entire). Florida pinesnake reproduction is tied to seasons with suitable temperature and moisture regimes. Altered weather conditions during these seasons may result in periodic reproductive failure, though we do not know of any evidence demonstrating this. Among other reptiles, it has been shown that high temperatures that restrict foraging activity by reptiles can lead to energy shortfalls, and ultimately reduced population growth (Gibbons et al. 2000, p. 660; Huey et al. 2010, p. 833; Sinervo et al. 2010, entire).

We believe the greatest imminent potential risk from climate change will be through its effects on suitable habitat and its management through prescribed fire. Current and continued projected warming will increase the risk of wildfire, insect, wind, and disease damage to southeastern forests. Importantly, forecasts of future climate predict a reduction in the number of suitable days to implement prescribed fire, which could limit restoration and management of the pine savanna conditions upon which Florida pinesnakes depend. When combined with issues associated with WUIs discussed earlier such as expanded urbanization, the ability to manage habitat with prescribed burning could be further constrained and there will be greater potential for the degradation of remaining pinesnake habitat.

Climate change also poses risks to coastal populations of the Florida pinesnake due to sea level rise (SLR). Global mean sea level has risen about 7-8 inches (16-21 cm) since 1900, with about half of that rise occurring since 1993 (Hayhoe et al. 2018, p. 85). In areas of the Southeast, tide gauge analysis reveals as much as 1 to 3 feet (0.30 to 0.91 meters) of local relative SLR in the past 100 years (Carter et al. 2018, p. 757). The future estimated amount that sea level will rise depends on the response of the climate system to warming, as well as on the future scenarios of human-caused emissions (Hayhoe et al. 2018, p. 85).

Coastal populations of Florida pinesnakes are predicted to be directly impacted by inundation of upland habitat directly along the coast by rising sea levels, resulting in loss of habitat. Although the amount of habitat predicted to be lost within a given population due to SLR varies considerably depending on the location of the population, 55 Florida pinesnake populations are considered to be vulnerable to SLR (i.e., population is anticipated to lose some amount of suitable habitat under all SLR scenarios).

3.5 Persecution, Harassment, and the Pet Trade

Humans have a long history of persecuting snakes; whether a snake is venomous or not, they tend to be viewed as vile and loathsome creatures (Burghardt et al. 2009, p. 262). Fear of snakes, called ophiophobia, has made snake conservation more difficult than other vertebrate groups (Burghardt et al. 2009, p. 262). The negative perception of snakes ranges from low interest, to harassment, to persecution resulting in deliberate killing. Many human-snake encounters result in the death of the snake (Whitaker & Shine 2000, p. 121). Due to defensive behaviors such as opening the mouth and hissing loudly, pinesnakes superficially resemble rattlesnakes (Gibbons & Dorcas 2005, p. 112) and they tend to be viewed as a threat to humans. As a result, pinesnakes may be killed by people who do not know they are harmless. Florida Fish and Wildlife Conservation Commission and South Carolina Department of Natural Resources have both noted that they have received reports of people fearing that Florida pinesnakes are venomous and killing them in their yards (Enge et al. 2016, p. 27; A. Grosse, personal communication July 1, 2021, p. pers. comm.). Though persecution is well documented, estimates of snake mortality from persecution are rare, particularly within North America. A study of timber rattlesnakes (*Crotalus horridus*) in Indiana documented 2 of 47 (~5%) radio-tracked snakes within a state forest were killed by persecution over a five-year period (Olson et al. 2015, p. 274). Directly extrapolating this mortality rate to more developed landscapes is not possible, but it is reasonable to assume it would be greater and that as development increases and encroaches on the remaining Florida pinesnake habitat, human-snake interactions and associated snake mortality will likely increase.

In addition to deliberate persecution, pinesnakes may be vulnerable to harvest and persecution methods targeted at other wildlife. In the recent past rattlesnake roundups were organized throughout the southeast, though most of these roundups have now been converted to wildlife festivals. These events usually offered prizes for the largest and most rattlesnakes collected. To

collect rattlesnakes a technique of blowing fumes of noxious liquids "gassing" down tortoise burrows were primarily used (Means 2009, p. 139). Since the Florida pinesnake uses gopher tortoise burrows, it is vulnerable in areas where the practice of "gassing" these burrows to drive out rattlesnakes is common even if now illegal. Research on some ecological effects of "gassing" tortoise burrows has shown that Florida pinesnakes subjected to the noxious fumes died within 24 days (Speake & Mount 1973, p. 267).

In the past, collection was considered a contributor to Florida pinesnake population declines in Florida (Enge et al. 2016, p. 27; Franz 1992, p. 257). Pinesnakes that displayed typical features and coloration were never as desirable to collectors as the more distinctive-looking northern and black subspecies, but there was a demand for amelanistic, leucistic, and patternless specimens (Enge et al. 2016, p. 27). Currently in Florida, the Florida pinesnake is protected by regulation that prohibits their commercial sale and limits possession to one individual, except for albino and amelanistic specimens (likely captive-bred) that can also be sold (Enge et al. 2016, p. 27). In South Carolina, there are no possession limits for pinesnakes, but they are not permitted to be collected on state, federal, or tribal lands. They can legally be collected from private lands. In Georgia, native non-venomous snakes are illegal to collect or possess without a state permit, as is the sale of native non-venomous snakes. Therefore, a persistent demand for pinesnakes among reptile collectors and with social media networks that make it easier to learn the locations of robust pinesnake populations make collection a moderate threat to the subspecies even with increased regulation against harvest, sale, or possession.

3.6 Disease

In wild populations of reptiles, debilitating diseases are most likely secondary expressions in individuals with impaired resistance caused by one or more primary environmental stressors, such as habitat degradation, invasive species, or pollution (Gibbons et al. 2000, p. 658). These primary environmental stressors can lead to immune suppression, which can further lead to an increase in morbidity and mortality from infectious disease (Allender et al. 2006, p. 107). Over the past several decades, the number of emerging fungal diseases and the number of species extinctions and extirpations caused by those diseases has increased (Lorch et al. 2015, p. 1).

Ophidiomycosis (also known as snake fungal disease) caused by the fungus *Ophidiomyces ophiodiicola*, is a pathogen of North American snakes that poses a potentially serious threat to the health and stability of snake populations (Allender et al. 2020, p. 1). Infected wild snakes have several distinct lesions on various parts of the body, head, or tail, and often the animal will die from complications from the infection rather than from direct fungal damage (Lorch et al. 2016, pp. 4–5). First noted in 2006 as a severe skin infection associated with a precipitous decline in timber rattlesnakes (*Crotalus horridus*) in the northeastern United States, Ophidiomycosis has since been implicated in widespread morbidity and mortality across the eastern United States (Allender et al. 2015, p. 188; Lorch et al. 2016, p. 1). The fungus *O. ophiodiicola* has been detected in at least 23 states and one Canadian province, though there is

some speculation that it may be more widely distributed than the documented cases suggest because the efforts to monitor the health of many snake populations are limited (Thompson et al. 2018, p. 1).

Ophidiomycosis has been most often observed in pit vipers; however, the fungus has been detected on the skin of captive and free-ranging snake species representing 12 genera within two families, including colubrid snakes (Allender et al. 2015, p. 188; Thompson et al. 2018, p. 3). Recently, ophidiomycosis has been documented in Florida pinesnakes (Haynes et al. 2020, p. 4). Among nine Florida pinesnakes sampled in southeast Georgia, three individuals had lesions consistent with ophidiomycosis and one individual had lesions and a positive PCR test for *O. ophiodiicola* (Haynes et al. 2020, p. 4). It is not reported if all these samples came from the same site/population or if the fungus has been found at multiple sites/populations. Although fungal infection from *O. ophiodiicola* is concerning as a potential direct cause of death for individual Florida pinesnakes, there is also potential for indirect effects as well. The prevalence of disease within a snake population can cause differences in the health and body condition of snakes in the population. For example, the inadequate condition of female snakes in a population due to disease can affect reproductive rates and viability of that population (Allender et al. 2006, p. 31). Additionally, behavioral changes from ophidiomycosis may manifest as prolonged periods spent basking, resulting in higher metabolic rate and faster consumption of resources and may increase exposure to predators (Allender et al. 2020, p. 11). Though the impact of snake fungal disease on most snake populations is currently unknown, the known effects of this and other infectious diseases is a concern, especially for species persisting at small population sizes (Allender et al. 2015, p. 194).

Another emerging threat that could eventually impact wild snake populations, including the Florida pinesnake, within the southeast United States is *Raillietiella orientalis,* an invasive pentastomid parasite introduced by nonnative Burmese pythons (*Python bivittatus*; (Farrell et al. 2019, p. 73). The lung parasite has been found to have infected at least 14 native snake species in south/central peninsula Florida and has been documented outside the established range of the Burmese python, indicating that native snakes are competent hosts and that the parasite is rapidly moving north (Miller et al. 2020, p. 1,8). Many pythons found sick or dead are infected with the parasite and there is evidence of increased mortality in the native pygmy rattlesnake (*Sistrurus miliarius*; (Farrell et al. 2019, p. 75). This parasite has not yet been documented in Florida pinesnakes and it is unknown if there are any sublethal effects on native snakes and how the parasite impacts snake fitness and populations (Farrell et al. 2019, p. 75).

3.7 Conservation Measures

3.7.1 Conservation Lands

Suitable habitat for the Florida pinesnake can be found within National Wildlife Refuges, National Forests, State Lands, and other protected areas set aside for conservation across the

subspecies' range (Table 3-1). The subspecies does not have a sensitive species status under U.S. Forest Service policies for National Forest lands, however, the National Forests in Florida manages habitat for a diversity of species, including the Florida pinesnake. Habitat improvements, including ecosystem restoration, enhancement, protection, prescribed burning, and mechanical upland habitat restoration conducted across the subspecies' range have likely benefited Florida pinesnake populations. Most conservation lands owned by Federal and State agencies are expected to remain protected and managed for conservation purposes in the near future, which would eliminate the risk of direct loss of habitat to urbanization in these areas.

It should be noted, that while most conservation lands whether federal, state, county, city, or private in nature, have regulations, rules, and policies that protect most wildlife occurring on such lands, enforcement and prosecution is often ineffective. Therefore, caution is advised regarding populations as "protected" simply because they occur on conservation land.

Many of the conservation lands in which Florida pinesnakes occur are managed for other longleaf-associated species, such as red-cockaded woodpeckers (*Picoides borealis*) and gopher tortoises. This habitat management for these species likely benefits Florida pinesnakes, particularly when management results in an open canopy system with more diverse groundcover.

Table 3-1. Conservation lands within the current Florida pinesnake range that continue to provide suitable habitat and have occurrence records. This is not a comprehensive list of all conservation lands for the subspecies and does not include private lands held in conservation easement.

STATE	MANAGER	PROPERTY
SOUTH CAROLINA	Dept. of Energy	Savannah River Site
	U.S. Forest Service	Francis Marion National Forest
	State	Aiken Gopher Tortoise Heritage Preserve
		Lewis Ocean Bay Heritage Preserve
		Palachucola Wildlife Management Area
		Tillman Sand Ridge Wildlife Management Area
		Webb Wildlife Center
GEORGIA	U.S. Fish and Wildlife	Okefenokee National Wildlife Refuge
	Dept. of Defense	Fort Benning
		Fort Gordon
		Fort Stewart
		Townsend Bombing Range
	State	Big Hammock Wildlife Management Area
		Chattahoochee Fall Line Wildlife Management Area
		Sandhills Wildlife Management Area
		Yuchi Wildlife Management Area
	Private	Jones Center at Ichauway
FLORIDA	U.S. Fish and Wildlife	Lower Suwannee National Wildlife Refuge
		Merritt Island National Wildlife Refuge
		St. Marks National Wildlife Refuge
	U.S. Forest Service	Apalachicola National Forest
		Ocala National Forest
	Dept. of Defense	Avon Park Air Force Range
		Camp Blanding Joint Training Military Reservation
		Eglin Air Force Base
		Naval Air Station Whiting Field

	State	Alfred B. Maclay Gardens State Park Apalachee Wildlife Management Area Bell Ridge Longleaf Wildlife and Environmental Area Belmore State Forest Blackwater River State Forest Camp Blanding Military Reservation Charles H. Bronson State Forest Chassahowitzka Wildlife Management Area Choctawhatchee River Water Management Area Devil's Millhopper Geological State Park Econfina Creek Water Management Area Edward Ball Wakulla Springs State Park Etoniah Creek State Forest Fort White Wildlife and Environmental Area Fred Gannon Rocky Bayou State Park Goethe State Forest Ichetucknee Springs State Park Indian lake State Forest Jennings State Forest Joe Budd Wildlife Management Area Jonathan Dickinson State Park Kissimmee Prairie Preserve State Park Lafayette Forest Wildlife and Environmental Area Lake Louisa State Park Lake Talquine State Forest Lake Wales Ridge Wildlife and Environmental Area Marjorie Harris Carr Cross Florida Greenway State Recreation Area Mike Roess Gold Head Branch State Park Palatka-to-Lake-Butler State Trail Paynes Prairie Preserve State Park Pine Log State Forest Platt Branch Wildlife and Environmental Area Ralph E. Simmons Memorial State Forest Savannas Preserve State Park Seminole State Forest Silver Springs State Park Split Oak Forest Wildlife and Environmental Area St. Sebastian River Preserve State Park Stephen Foster Folk Culture Center State Park Suwannee Ridge Wildlife and Environmental Area Three Lakes Wildlife Management Area Torreya State Park Tosohatchee Wildlife Management Area Twin Rivers State Forest Watermelon Pond Wildlife and Environmental Area Wakulla State Forest Withlacoochee State Forest Withlacoochee State Trail
ALABAMA	U.S. Forest Service Dept. of Defense State	Conecuh National Forest Fort Rucker Blue Spring Wildlife Management Area

Sources: (Enge et al. 2016, pp. 67–69; Georgia Department Natural Resources 2015, pp. D15–D16; Petersen et al. 2017, pp. 3–18; South Carolina Department Natural Resources 2015, p. Supplement Volume: Longleaf Pine Reptile Guild p. 4)

3.7.2 Department of Defense

Throughout the Southeast, eight military installations have records of Florida pinesnakes (Petersen et al. 2017, p. 3-18). Active prescribed burning programs are implemented on most military installations to manage for longleaf pine ecosystems, which can benefit conservation of the Florida pinesnake. As part of implementation of the Sikes Improvement Act (1997), the Secretaries of the military departments are required to prepare and implement Integrated Natural Resource Management Plans (INRMP) for each military installation in the United States. No installations specifically include Florida pinesnake habitat and population management prescriptions and goals within their INRMPs; however, most of the INRMPs do include specific management for other longleaf species such as the red-cockaded woodpecker and gopher tortoise, which would provide some benefit to Florida pinesnakes. The Department of Defense's Readiness and Environmental Protection Integration (DoD REPI) program also offers opportunities to expand land conservation beyond installation boundaries to improve military training flexibility by defending against incompatible development and reducing regulatory restrictions that inhibit military activities. Working through landscape partnerships, the DoD REPI program has helped protect, restore, and maintain longleaf pine habitat across the Southeast.

3.7.3 Longleaf Ecosystem Conservation and Restoration

There are several initiatives and programs in the Southeast whose objectives include the establishment, restoration, and management of the native upland longleaf pine ecosystem. These include the Working Lands for Wildlife programs of the Natural Resources Conservation Service, USFWS Partners for Fish and Wildlife programs, and America's Longleaf Restoration Initiative. Public land partners, including Federal and State Agencies, private landowners, non-governmental organizations are all active partners in America's Longleaf Restoration Initiative. This is a collaborative effort of multiple public and private sector partners that actively supports range-wide efforts to restore and conserve longleaf pine ecosystems with a 15-year goal to increase longleaf from 3.4 to 8.0 million acres. Over the past decade, more than 1.3 million acres of longleaf has been planted and now 4.7 million acres of longleaf forests occur across the historic range (America's Longleaf Restoration Initiative 2019, p. 2).

Candidate Conservation Agreements (CCAA) with Assurances are voluntary commitments made by non-Federal partners to undertake actions that will remove or reduce threats to candidate or other at-risk species. The goal of any CCAA is to provide a net conservation benefit to the covered species and to preclude the need to list species under the ESA. As an incentive to the non-Federal property owner who engages in voluntary conservation actions for a particular species or group of species, landowners are given regulatory assurances if the species is listed under the ESA. Only one such agreement could provide potential benefits to the Florida pinesnake. In 2017, a CCAA was established with the Camp Blanding Joint Training Center providing protections for approximately 17,000 acres (6,879 hectares) of sandhill to be managed

for the benefit of multiple at-risk species, including the Florida pinesnake (USFWS et al. 2017, entire).

3.7.4 State Protections

The Florida pinesnake is listed as State threatened in Florida and Alabama, and not listed in Georgia and South Carolina (Table 3-2). In Florida, the Florida pinesnake is protected by regulation that prohibits their commercial sale and limits possession to one individual, except for albino and amelanistic specimens (likely captive-bred) that can also be sold (Enge et al. 2016, p. 27). In South Carolina pinesnakes are high priority species and though there are no possession limits, under a new regulation it is illegal to transfer possession of native reptiles and amphibians in the State, or export them from the State, making it illegal to buy, sell, trade, or gift the subspecies (South Carolina Department Natural Resources 2020, unpaginated).

Table 3-2. Florida pinesnake listing status and rank by state (north to south) within the subspecies' range.

State	State Listing	State Rank/Priority
North Carolina*	Threatened	Species of Greatest Conservation Need
South Carolina	Not Listed	S2/Highest Priority
Georgia	Not Listed	S3/High Priority
Florida	Threatened	S3/Taxa of Concern
Alabama	Threatened	S2/High Priority

*Information is for northern pinesnake (*Pituophis melanoleucus melanoleucus*) Sources: (Alabama Department Conservation and Natural Resources 2015, p. 410; Florida Fish and Wildlife Conservation Commission 2019, p. 156; Georgia Department Natural Resources 2015, p. D-10; North Carolina Wildlife Resources Commission 2015, p. 126; South Carolina Department Natural Resources 2015, pp. 2–6)

3.8 Summary of Factors Influencing Viability

We reviewed the potential factors that could be affecting the viability of the Florida pinesnake (Figure 3-5). Concerns about the subspecies' status revolved around the following factors: (1) habitat loss, conversion, and fragmentation resulting in the loss of longleaf pine habitat across the range of the Florida pinesnake; (2) based on the number of DOR individuals in our occurrence records, road mortality that could result in altered population structures; (3) invasive species; (4) effects of climate change resulting in increased temperatures, decreased precipitation, increased severe weather such as drought, flooding, or storms, changes in wildfire frequency and intensity, decreased ability to conduct prescribed burns, and sea level rise; (5) persecution and harassment by humans; and (6) infectious diseases such as snake fungal disease that may have impacts on existing populations.

The primary concerns for the Florida pinesnake's status are related to the loss and degradation of habitat. Habitat loss is due to a number of factors including fire suppression, silvicultural practices, SLR, conversion of land to agriculture, and urbanization. The current constraints on the ability to manage longleaf pine habitat through prescribed fire may be exacerbated by urbanization and climate change. It is possible that several of these factors are acting

synergistically to impact the Florida pinesnake. Although there is uncertainty surrounding the stressors we evaluated and their synergistic effects, this does not prevent us from making a credible assessment of the likely direction and magnitude of those impacts, even though it may not be possible to make such predictions of impacts with precision.

Projections of habitat loss due to urban development and climate change are carried forward in our assessment of Florida pinesnake populations and the overall viability of the subspecies. We were not able to assess impacts from invasive species, such as red imported fire ants and feral hogs, persecution, over-collection for the pet trade and increased harassment, and disease because datasets and or other information sources do not exist that capture the extent and degree of impact of these stressors to Florida pinesnake populations across the subspecies' range.

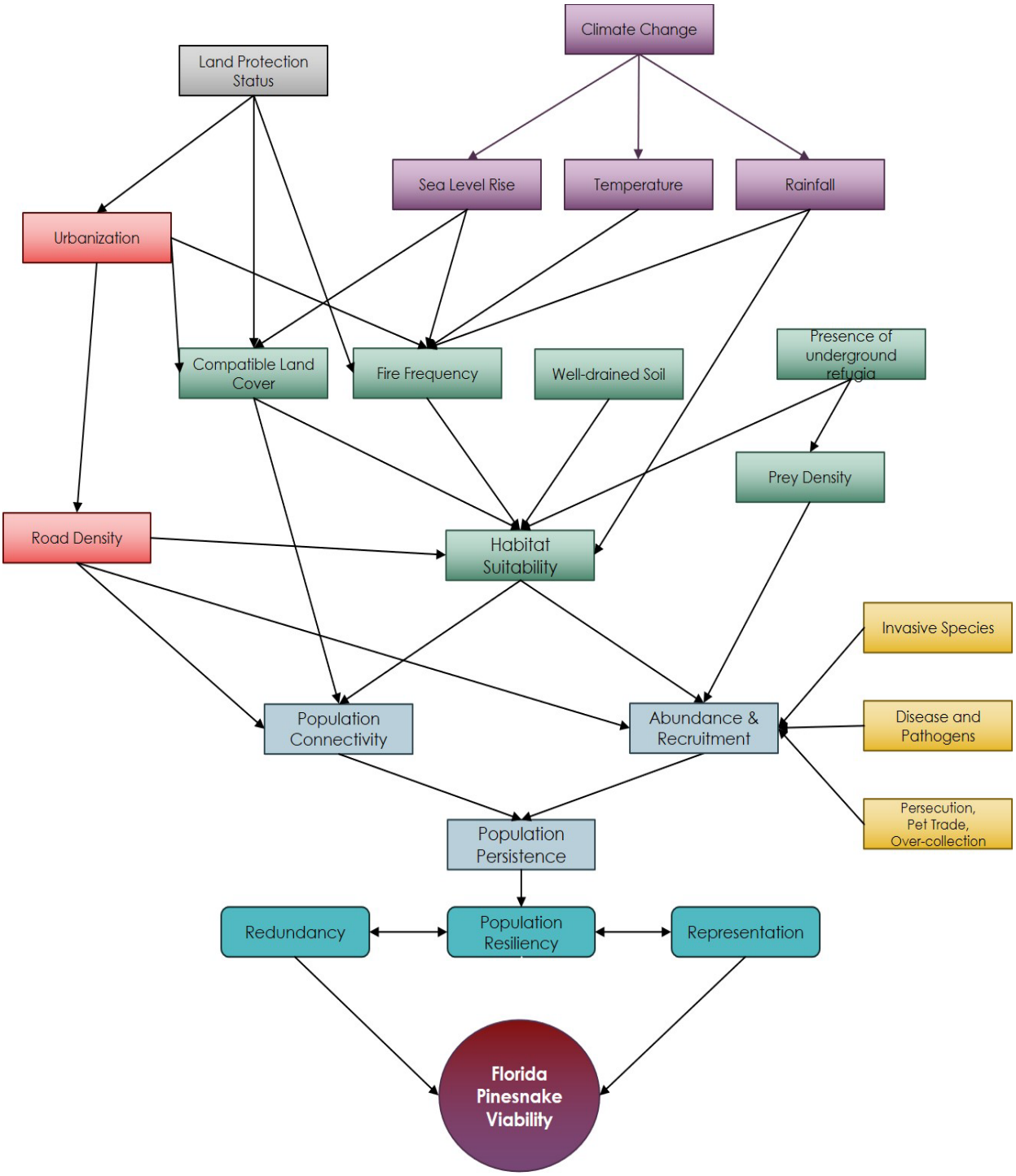

Figure 3-5. Influence diagram for the Florida pinesnake, showing relationships between factors and subspecies' viability. Color meanings: Grey – land ownership and protection status; Light red and yellow – stressors; Purple – climate-based factors; Green – habitat-based factors; Light blue – population metrics; Teal – components of viability (3Rs); Dark red/purple – subspecies viability.

CHAPTER 4 – POPULATION AND SUBSPECIES NEEDS AND CURRENT CONDITION

In this chapter, we consider the historical and current distribution of the Florida pinesnake. We first define populations of the subspecies. Next, we characterize the needs of the subspecies in terms of population resiliency and subspecies' representation and redundancy (the 3Rs). Finally, we estimate the current condition of the Florida pinesnake using population and habitat metrics used to characterize the 3Rs.

4.1 Methods for Estimating Current Condition

For the purpose of this assessment, we defined ***viability*** as the ability of the Florida pinesnake to sustain populations in the wild over time. Using the SSA framework, we describe viability of the Florida pinesnake by first defining our populations, estimating current condition, and then in Chapter 5 predicting the future condition using the metrics of the 3Rs.

To characterize the 3Rs for the Florida pinesnake, we used an existing habitat suitability model developed for the Florida pinesnake to estimate population persistence across a portion of the subspecies' range, which we refer to as the analysis area (Figure 4-1; Crawford et al. 2020a, Supplemental Text S1, entire). The analysis area boundary was created during a multi-species habitat modeling study that began prior to developing the SSA (Crawford et al. 2020a, entire). We then used the patterns found in these analyses to perform a qualitative assessment of conditions for areas outside of the analysis area. Due to very recent changes in what we are presenting as the subspecies range, the analysis area does not extend to the northern portion of the range beyond the southeastern portion of South Carolina.

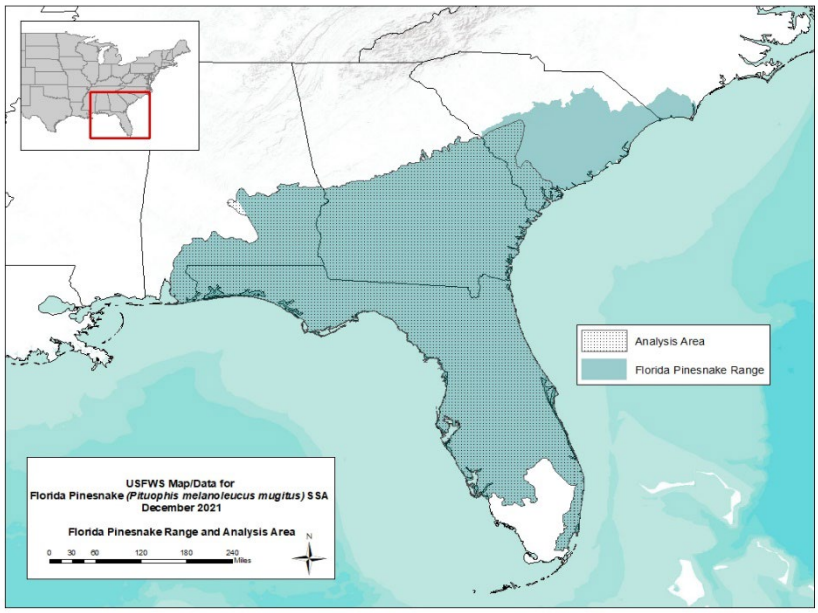

Figure 4-1. Comparison of the Florida pinesnake range (teal) and analysis area used in the habitat and persistence modeling in the SSA (dotted area).

4.1.1 Defining Populations

For this assessment, we defined populations for the subspecies as contiguous areas surrounding known Florida pinesnake occurrences with habitat conducive to survival, movement, and inter-breeding among individuals within the area. We compiled all subspecies records in Geographic Information System (GIS). The compiled dataset included records maintained by Natural Heritage Programs, USFWS, U.S. Forest Service, DoD, State agencies, academic researchers, and HerpMapper, which include opportunistic sightings and observations during research and monitoring. To delineate populations, we used records with available latitude and longitude information, and we buffered the subspecies occurrence records by 6 km (3.73 mi). The choice of buffer distance was similar to protocols used by NatureServe (NatureServe 2021, unpaginated) but further informed by reviewing telemetry studies for the subspecies and expert input. NatureServe suggests a separation distance for colubrid snakes of 10 km (6.2 miles) for suitable habitat and 1 km (0.62 miles) for unsuitable habitat. This recommendation was based on a limited number of studies of movement and range in colubrid snakes (NatureServe 2021, unpaginated). Two telemetry studies of Florida pinesnakes indicated that a 10 km buffer may be too large for purposes of delineating populations. Those studies estimated that Florida pinesnake home ranges were smaller, between 57-59.2 ha (141-146 ac), relative to other similar-sized colubrid snakes (Franz 2005, p. 121; Miller et al. 2012, p. 709). The maximum annual movement or dispersal distance reported for a Florida pinesnake is 4.26 km (2.65 mi; Franz 2005, p. 124). Though it is reasonable to believe that dispersing pinesnakes could and would make larger movements, experts we consulted agreed that 6 km was a reasonable distance for a buffer around any Florida pinesnake record. This distance represents the potential maximum radial extent an individual would likely occupy and is likely a generous estimate of the area actually used by an individual.

Once the 6 km buffers were created, any buffers that overlapped each other were merged, which created contiguous areas (populations) where any points within 6 km of each other were included as part of the same population. We divided one population that spanned the Savannah River, but we did not divide populations spanning smaller rivers or interstates (primary roads) as we felt that the size of this subspecies allowed for their movement across these barriers. Additionally, we clipped final population boundaries by the Savannah and Chattahoochee Rivers and the U.S. Coastline.

4.1.2 Population Resiliency

To characterize resiliency, we used a modified version of the population persistence model developed for the southern hognose snake SSA and subsequently published in the peer-reviewed literature (USFWS 2019, entire; Crawford et al. 2020b, entire). We used this model, an adapted Cormack-Jolly-Seber (CJS) model (Cormack 1964, entire; Jolly 1965, entire; Seber 1965, entire) designed to estimate survival of individual animals based on mark-recapture data, to analogously estimate persistence (“survival”) of populations based on their observation histories.

Accordingly, in this adapted model, parameters estimated included re-detection probability of the species within the defined population area (analogous to "re-capture probability") and the probability of population persistence (analogous to "apparent survival probability"). Specifically, "current persistence probability" is defined as the probability that there is at least one Florida pinesnake in 2021 in the defined area for that population. The complement (1 – current persistence) can be interpreted as the probability a population is extirpated as of 2021. Inference in this model is thus at the site occupancy level, similar to an occupancy model; however, like the CJS model, site persistence is conditioned on the initial observation of a Florida pinesnake at the site. This type of model does not account for the potential colonization of sites or addition of populations, only the probability that populations persist once established (and initially observed).

Additionally, "current persistence probability" is not a direct measure of population size or growth rates, which are population-level metrics we could not estimate with available data. We assume large populations with sufficiently high survival and recruitment rates will have a higher probability of persisting over time. We also assume that more resilient populations are those that tend to occupy higher quality habitat, are more likely on protected lands, have some level of connectivity to other populations, and therefore, will be more likely to continue to persist on the landscape over time if current conditions did not change.

The population persistence model took information about Florida pinesnake detections, search effort and detection covariates, and site conditions to estimate the probability each population persisted from one year to the next through 2021 (i.e., "annual persistence probability"). The model uses several types of information to estimate persistence for each population. First, it uses a "detection history" of when Florida pinesnakes were or were not seen in a population area. We expect populations where Florida pinesnakes have been detected often and recently to have a higher probability of currently persisting. Second, detection histories account for search effort in the designated population area as determined by records of Florida pinesnakes and 12 other commonly co-occurring snake species reported by stakeholders or publications and citizen science records available through HerpMapper (HerpMapper 2021, unpaginated). Third, because a high percentage of Florida pinesnake records were encounters on roads, we assumed a bias for roads increasing the likelihood of detection and we included a measure of road area when we modeled detection. Specifically, we quantified the total area within each population boundary that was within 2 km (1.24 mi) of a major road. Fourth, the model uses three attributes of a population's conditions that may influence persistence probability: (1) mean habitat suitability inside a population boundary, (2) the proportion of the population boundary on protected land, and (3) the number of nearby populations within 10 km (6.2 mi) (Table 4-1). For protected lands, we used areas that are currently publicly owned and managed, as well as private lands that are registered in State or Federal programs where natural resource conservation is the goal.

We constructed the model in a Bayesian framework and fit the model with Markov chain Monte Carlo (MCMC) methods in Jags called from R via the R2jags package (Su and Yajima 2012, unpaginated). We calculated each population's current persistence probability and the associated uncertainty from the proportion of model iterations where the population was still persisting in 2021. Additionally, we summed the number of populations estimated to persist in 2021 within each representative unit and range-wide in each model iteration, and we used these to calculate the mean estimated number of populations persisting at these spatial extents.
We assigned diffuse prior distributions for all parameters, and we generated three MCMC chains using 350,000 iterations where we retained every third iteration from the last 300,000 iterations, yielding a final set of 300,000 samples from posterior distributions of the parameters. We assessed convergence based on the Brooks-Gelman-Rubin statistic (< 1.1 for all parameters) and by visually inspecting chain mixing in MCMC trace plots and posterior distribution plots for evidence of unimodality. To assess goodness-of-fit, we conducted posterior predictive checks (Gelman et al. 2000, entire) where we simulated datasets using parameters estimated in the model, calculated the mean number of populations with simulated detections of Florida pinesnakes in four time periods (1920-1929, 1950-1959, 1980-1989, and 2012-2021), and compared mean observations in these periods from the real dataset with values from simulated datasets. We based parameter inferences on posterior means and 95% Bayesian credible intervals (BCIs; 2.5th – 97.5th percentile of the distribution).

Estimates of Florida pinesnake persistence probabilities range between 0 (0%) and 1 (100%). The categories we chose were unlikely or extirpated (< 50%), more likely than not on landscape (50 – 79%), very likely on landscape (80 – 89%), highly likely on landscape (90 – 95%), and extremely likely on landscape or extant (95 – 100%). For a population to be classified in the more than likely, very likely, or extremely likely categories, the lower 95% credible interval of the estimated current persistence probability had to be equal to or greater than 50%, 80%, or 95% respectively. This criterion allowed us to say that we were 97.5% certain that a population was in that persistence category. A population was classified as unlikely or low likelihood of persisting if the lower limit of the 95% credible interval for current persistence probability was below a probability of 0.5 even if the mean estimate for that population exceeded 0.5. This means that our classification of persistence was conservative and unlikely to over-represent the current resilience of a population or range-wide status of the species.

Table 4-1. List of predictors used to model Florida pinesnake population persistence, the hypothesized relationships to population resilience, and the average and range of predictors across 133 populations within the analysis area.

Site condition predictor	Biological justification	Mean (range)
Habitat Suitability Index (HSI)	Higher habitat suitability represents areas of higher quality (well-drained, sandy soils, compatible forest/grassland landcover, frequent fires) that should increase Florida pinesnake survival, recruitment, and persistence. The average HSI within a population boundary is highly correlated to the amount of suitable habitat.	20.6% (1.1 – 55.3%)
Proportion on protected land	Higher proportions of a population that occur on protected land should increase habitat quality through regular management practices and may limit direct threats such as road mortality and collection.	21.2% (0 – 100%)
No. of populations within 10 km	Populations close to other populations may have a higher chance of long-term persistence. Nearby populations may provide opportunities for "rescue" where recolonization can occur after a catastrophe and provide for some genetic exchange; alternatively, nearby populations could provide a signal that there are localized conditions (e.g., geological, climatic) that promote population persistence that have not been otherwise captured in our analyses.	1.8 (0 – 15)

4.1.3 Species Representation and Redundancy

Representation reflects the ability of a species to adapt to changing environmental conditions and can be measured by the breadth of genetic or environmental diversity within and among populations. For the Florida pinesnake, we do not have information related to genetic diversity within and among populations. In the absence of subspecies-specific genetic and ecological diversity information, representation can be assessed based on the extent and variability of habitat characteristics across the geographical range (Wolf et al. 2015, p. 204). Ecoregions are a system of classification based on physiography, where areas with similar characteristics of land formation, dominant soil and vegetation types, climate, air and sea currents, and distribution of flora and fauna are grouped into a single ecoregion (Bailey 1983, entire; Bailey et al. 1994, entire). Ecoregions have been used to reflect broad areas within which local adaptations and genetic coadaptation have likely occurred. Therefore, we used ecoregions to act as an appropriate proxy for factors likely to influence the adaptive capacity of Florida pinesnakes across the landscape. We broke the Florida pinesnake range into seven representative units based on grouping Environmental Protection Agency (EPA) Level IV ecoregions by similar ecological characteristics (e.g., soil, geology) and dividing them by the Savannah and Chattahoochee rivers where appropriate (Figure 4-2).

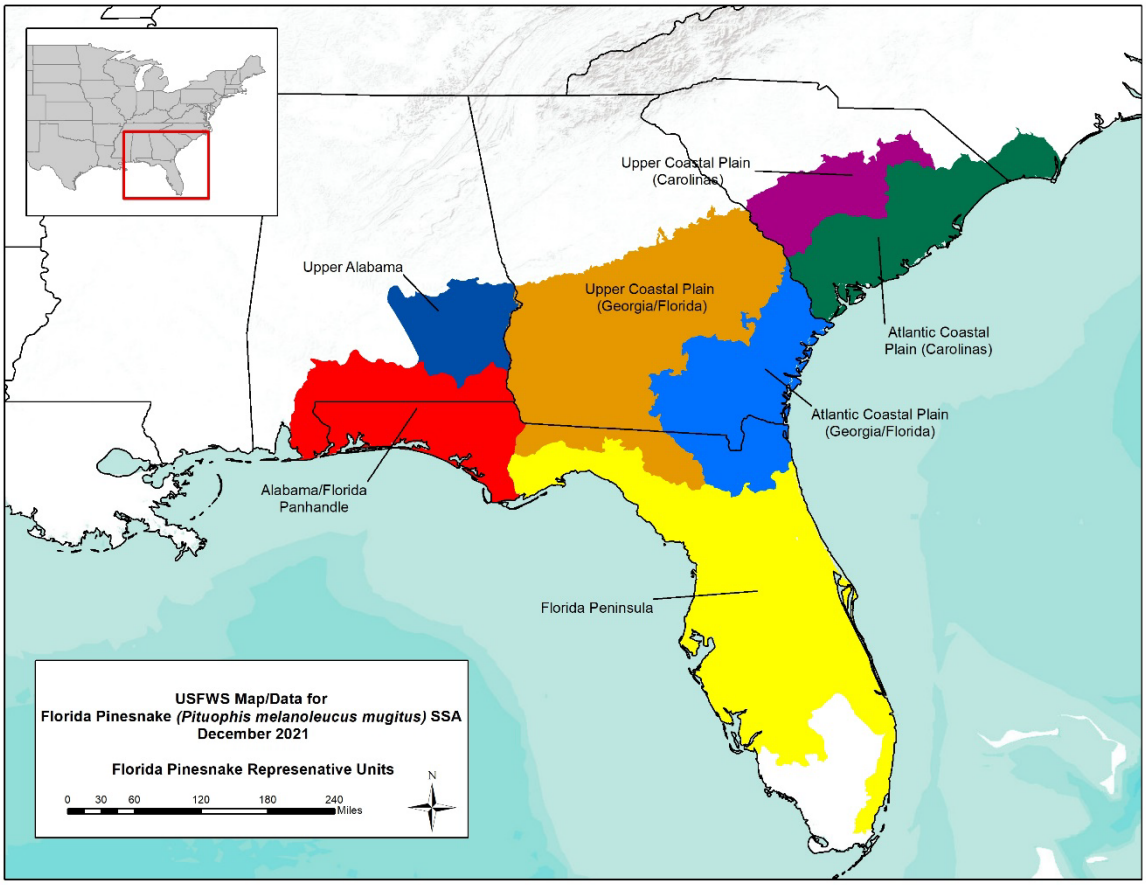

Figure 4-2. Representative units for the Florida pinesnake. Seven units were selected based on ecoregion groups.

In addition, the Florida pinesnake needs to exhibit some degree of spatial redundancy to maintain viability. Species-level redundancy reflects the ability of a species to withstand catastrophic events and remain extant, and is best achieved by having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events. Species that are well-distributed across their historical range are considered less susceptible to extinction and more likely to be viable than species confined to a small portion of their range (Carroll et al. 2010, entire; Redford et al. 2011, entire). In addition to guarding against a single or series of catastrophic events, redundancy is important to protect against losing irreplaceable sources of adaptive diversity.

When investigating redundancy for Florida pinesnakes, we consider a catastrophe to be any destructive event or episode that involves population-level impacts with the potential to negatively influence population resiliency outside of normal environmental and demographic stochasticity. Natural catastrophes that have the potential to occur in the Florida pinesnakes' range include large-scale drought, wildfires, hurricanes, and disease outbreaks.

To characterize representation and redundancy, we captured predictions of the number of resilient populations within a representative unit and range-wide in two ways. First, we summed the number of populations within a unit classified by their current estimated persistence

probabilities as described above (Section 4.1.2). Second, we recorded the number of populations predicted to persist in 2021 within each unit and range-wide using direct outputs from the model. Each model iteration recorded the number of populations persisting in 2021 in each representative unit and range-wide, and we used all model iterations to calculate the mean (the most likely prediction) and 95% credible intervals for the predicted number of persisting populations in 2021.

We measured representation using the number and distribution of resilient populations (i.e., those above a certain persistence probability threshold) across representative units in the species' range, as well as assessing the spatial distribution (latitudinal and longitudinal variability) of resilient populations. To have high representation, Florida pinesnakes would have at least one highly resilient population in each of the representative units and range-wide would have highly resilient populations that span the latitudinal and longitudinal extent of historical populations. We measured redundancy using the current number and distribution of resilient populations within representative units and across the range of the subspecies. To have high redundancy, the Florida pinesnake would need to have multiple highly resilient populations within a representative unit and throughout its range. For more detailed information beyond what is summarized in this section, see Appendix A.

4.2 Current Condition Results

4.2.1 Populations

For our population analysis, we obtained 1,989 Florida pinesnake records, from years 1881-2021. Many of the early occurrence records were for the county only without available coordinates. The occurrence records were spread throughout the subspecies' range, with the Florida Peninsula representative unit having the most records (Table 4-2). From these records, we identified a total of 153 populations of Florida pinesnakes with a mean area of 42,976 ha (range = 6,713 – 1,324,018 ha; Figure 4-3). If a population spanned more than one representative unit, the population was placed in the unit where most of the population boundary occurred for that population.

Table 4-2. Summary of number of occurrence records for the Florida pinesnake, with number of populations for representative units and range-wide.

Representative unit	No. of records	No. of populations
Upper Coastal Plain (South Carolina)	202	8
Upper Coastal Plain (GA/FL)	544	31
Atl. Coastal Plain (Carolinas)	89	17
Atl. Coastal Plain (GA/FL)	178	28
Florida Peninsula	761	47
AL/FL Panhandle	206	18
Upper Alabama	9	4
Range-wide	1989	153

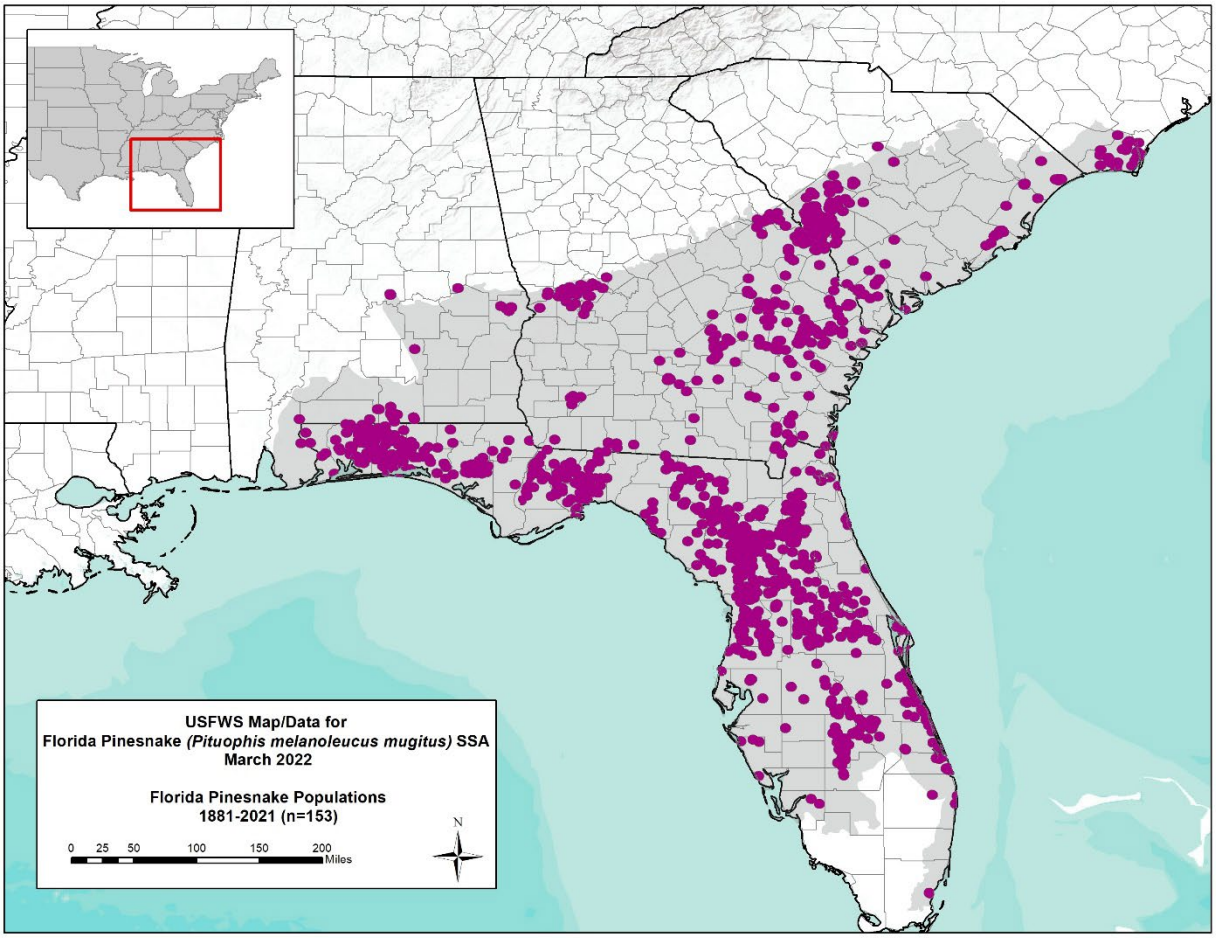

Figure 4-3. Distribution of 153 Florida pinesnake populations across the subspecies' range (shaded background). Populations represented were delineated using 6 km buffers around Florida pinesnake records and merging overlapping buffers.

4.2.2 Current Population Resiliency

To assess the current condition of the Florida pinesnake, we modeled persistence using 1,922 of the 1,989 occurrence records grouped into 133 populations that occurred within the analysis area. We were unable to estimate current persistence probability for 2 populations in the analysis area that represented two records that had no date associated with them, nor an additional 18 populations representing 65 records that fell outside the analysis area. Again, because the persistence model depended on a habitat suitability model already produced for the subspecies prior to recent reclassification of the subspecies range, we could not include those populations within the newly extended range in our model estimates.

Out of the 141 years we modeled (1881-2021), the mean number of years that at least one Florida pinesnakes was found in a given population was 5.17(range = 1 to 81). The number of observer-days per year per population (including only years with > 0 survey effort for each site) ranged from 1 to 94 (mean = 3.67), and records of non-target species were present in 84 of 133

(63.2%) populations. The persistence model showed adequate convergence and the posterior predictive check indicated that the model fit the data adequately in the first and last time periods assessed but had poorer fit in the two intermediate time periods assessed (see Appendix A for details). The mean detection rate, the probability of detecting at least one Florida pinesnake in a population in a given year, was relatively high (mean = 0.63, 95% CI = 0.57 – 0.69).

We present the number of populations (out of 133) in each persistence category, as well as the cumulative number of populations at or above each persistence category, in Table 4-3; Figure 4-4. Range-wide, we estimated that we have likely lost 30.8% (41/133) of Florida pinesnake populations (i.e., populations with a less than 50% probability of current persistence). Conversely, 69.2% of total populations had a greater than 50% probability of current persistence as of 2021. Moreover, we estimated that 49.6% of the total populations were very likely or extremely likely to persist as of 2021 (moderate and high resiliency), and 21.8% of total populations were extremely likely to persist as of 2021 (high resiliency). When looking at only currently extant populations (n=92), 71.7% (66/92) are very likely or extremely likely to persist (≥80% probability of persistence), and 31.5% (29/92) are extremely likely (>95% probability of persistence) to persist as of 2021. We report mean persistence estimates and 95% credible intervals for all populations analyzed in Appendix A. Estimated current persistence probability decreased moderately with years since the last observation (Figure 4-5) and as expected, increased with habitat suitability, proportion of protected land, and number of populations within 10 km. One population stands out with a higher-than-average current persistence probability given that it was last observed in 1950. This population had relatively high habitat suitability and high proportion of area on protected land, but no known search effort in years following the last observation. Because of these conditions, the model predicted this population had a higher chance of persistence compared to other populations not observed since the early 1950s. Conducting additional, follow-up surveys at sites where Florida pinesnakes have not been observed in recent years might have yielded additional observations that would affect current persistence estimates.

Table 4-3. Number and percentage of Florida pinesnake populations in each persistence category, cumulative number of populations at or above each threshold for all populations (n=133) and for currently extant populations (n=92) in 2021, and area occupied.

Population Persistence	Number of populations in each category	% of total	Cumulative number of populations at or above each threshold	% of total (% of extant populations)	Area occupied (acres/ km2)	% of total area occupied
Extremely Likely on Landscape (Extant) 95-100%	29	21.8%	29	21.8% (31.5%)	10662434.5/ 43167.8	69.6%
Very Likely on Landscape 80-94.9%	37	27.8%	66	49.6% (71.7%)	2232475.2/ 9038.4	14.6%

More Likely than Not 50-79.9%	26	19.5%	92	69.2% (100%)	941039.6/ 3809.9	6.2%
Unlikely < 50% (Extirpated)	41	30.8%	-	-	941039.6/ 5971.4	9.6%
Total	133	100%			15310880.8/ 61987.4	100%

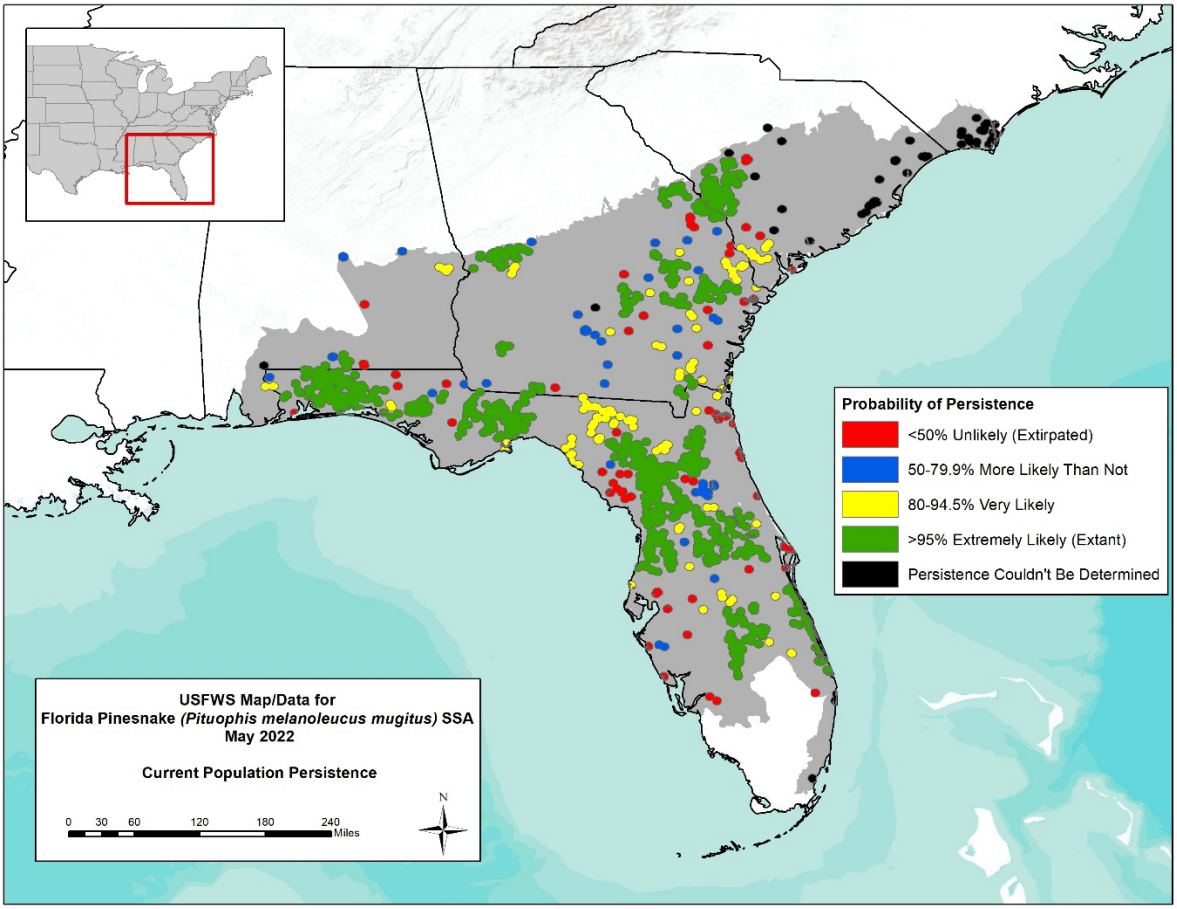

Figure 4-4. Populations of Florida pinesnakes by category of current persistence probability in 2021.

The relationship between persistence and HSI agreed with expert judgment about the habitat conditions that should favor Florida pinesnake persistence. The model also supported the belief that populations on protected lands should have a greater likelihood of persistence, but this effect was not as strong as the other predictors. The relationship between number of populations within 10 km and persistence supported the importance of connectivity among populations. This may indicate the demographic importance of "rescue effects," where recolonization can occur after a catastrophe, or be an artifact of large core areas of high-quality habitat that support multiple populations or subpopulations over large geographic areas. Representative unit did not appear to have a large effect on persistence probability (see Appendix A for additional details).

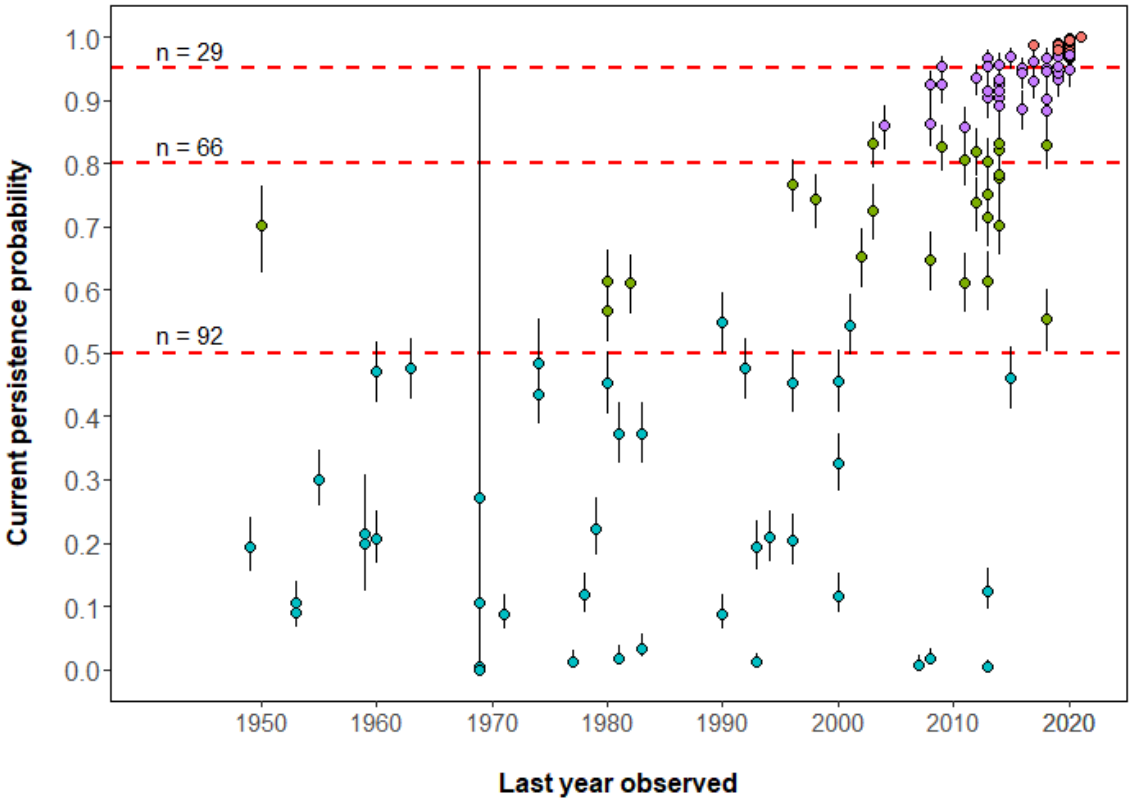

Figure 4-5. Probability of persistence in the current year (2021) for Florida pinesnake populations (n=133) related to the last year an individual was observed in the population. Blue points represent populations unlikely to be persisting on the landscape (i.e., do not have a 95% or greater probability of exceeding the 50% persistence probability threshold), and green, purple, and red points represent populations that have at least a 95% probability of exceeding the 50, 80, or 95% persistence probability thresholds, respectively. The 50, 80, and 95% thresholds are indicated with horizontal red dashed lines and n values indicate the total number of populations with a 95% probability of being above each threshold.

4.2.3 Current Species Representation and Redundancy

We summarized estimated current [as of 2021] representation and redundancy (Table 4-4; Figure 4-6). Overall, we estimated that across all populations over time, 69.2% of populations are more likely than not to currently exist on the landscape as of 2021. We estimated there has likely been some range-wide loss of redundancy. Out of 133 populations within the analysis area, we estimated that 92 are more likely than not (>50% probability) persisting as of 2021. We considered these to be extant populations. Populations with higher estimated persistence probabilities were spread evenly across the subspecies' range. We estimated that six of the seven representative units have more than 50% of the known populations that are more likely than not (>50% persistence probability) to persist as of 2021, however only one representative unit (i.e., Upper Coastal Plain (South Carolina), was estimated to have more than 50% of their known populations still extremely likely to persist (≥95%) as of 2021 and based on age of the records for the populations that were not in the analysis unit, the percentage of populations in this category maybe overestimated. We estimated there is a minimum of at least one Florida

pinesnake population within each of the representative units that are very likely to persist on the landscape today (≥80%, moderate to high resiliency). The Florida pinesnake has experienced a decrease in latitudinal variability associated with range contraction, particularly in southwest Florida. Though we could not include two populations from this region in our analysis because they fell outside of the HSI analysis area the records that make up these populations are sparse (4 records) and are considered historic (1919-1980), indicating a low likelihood that these populations still exist.

Table 4-4. Florida pinesnake summary statistics of number of occurrence records, number of populations (pops), mean persistence probability, and cumulative number of populations at each current persistence probability thresholds within representative units.

Representative unit	No. of records	No. of pops total (in analysis area)	Mean (±95% CIs) no. of pops predicted to persist	No. of pops at or above persistence threshold (as percentage of no. pops/unit)			
				Unlikely < 50% (Extirpated)	More Likely than Not ≥ 50%	Very Likely on Landscape ≥ 80%	Extremely Likely on Landscape (Extant) ≥ 95%
Upper Coastal Plain (South Carolina)	202	8 (4)	3 (3, 4)	1 (25%)	3 (75%)	3 (75%)	3 (75%)
Upper Coastal Plain (GA/FL)	544	31 (30)	23 (18, 28)	5 (16.7%)	25 (83.3%)	16 (53.3%)	9 (30.0%)
Atl. Coastal Plain (Carolinas)	89	17 (5)	2 (1,4)	3 (60%)	2 (40%)	2 (40%)	0 (0%)
Atl. Coastal Plain (GA/FL)	178	28 (28)	21 (15, 26)	8 (28.6%)	20 (71.4%)	15 (53.6%)	3 (10.7%)
FL Peninsula	761	47 (45)	28 (24, 33)	17 (37.8%)	28 (62.2%)	22 (48.9%)	9 (36%)
AL/FL Panhandle	206	18 (17)	11 (7, 16)	6 (35.3%)	11 (64.7%)	7 (41.2%)	5 (29.4%)
Upper Alabama	9	4 (4)	3 (1, 4)	1 (25%)	3 (75%)	1 (25%)	0 (0%)
Range-wide	1989	153 (133)	92 (81, 102)	41 (30.8%)	92 (69.2%)	66 (49.6%)	29 (21.8%)

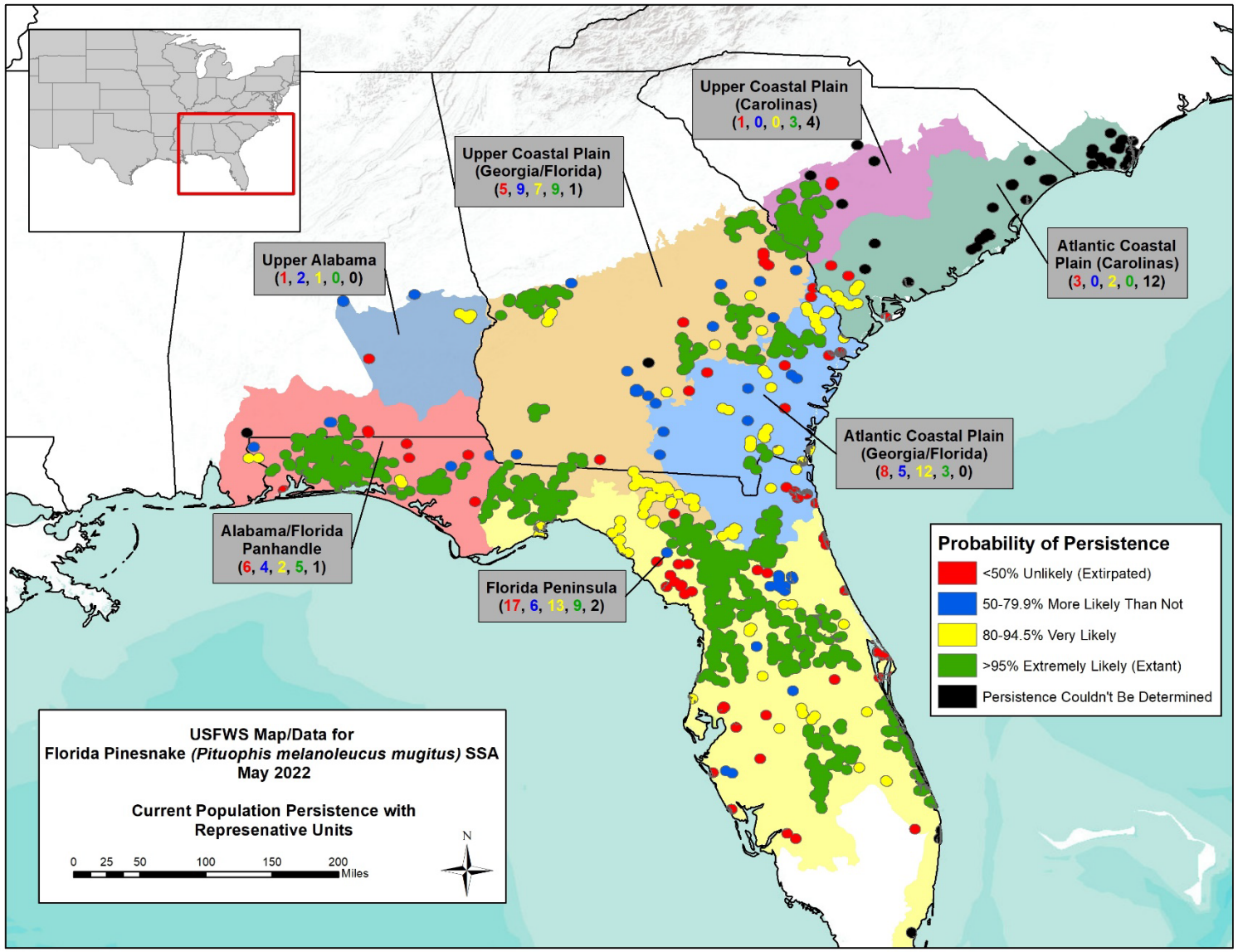

Figure 4-6. Florida pinesnake populations across representative units based on probability of persistence. Green populations are extremely likely to currently occur on the landscape or be extant; yellow populations are very likely; blue more likely than not; red populations are unlikely to currently exist (i.e., considered extirpated), and black are populations where persistence could not be estimated.

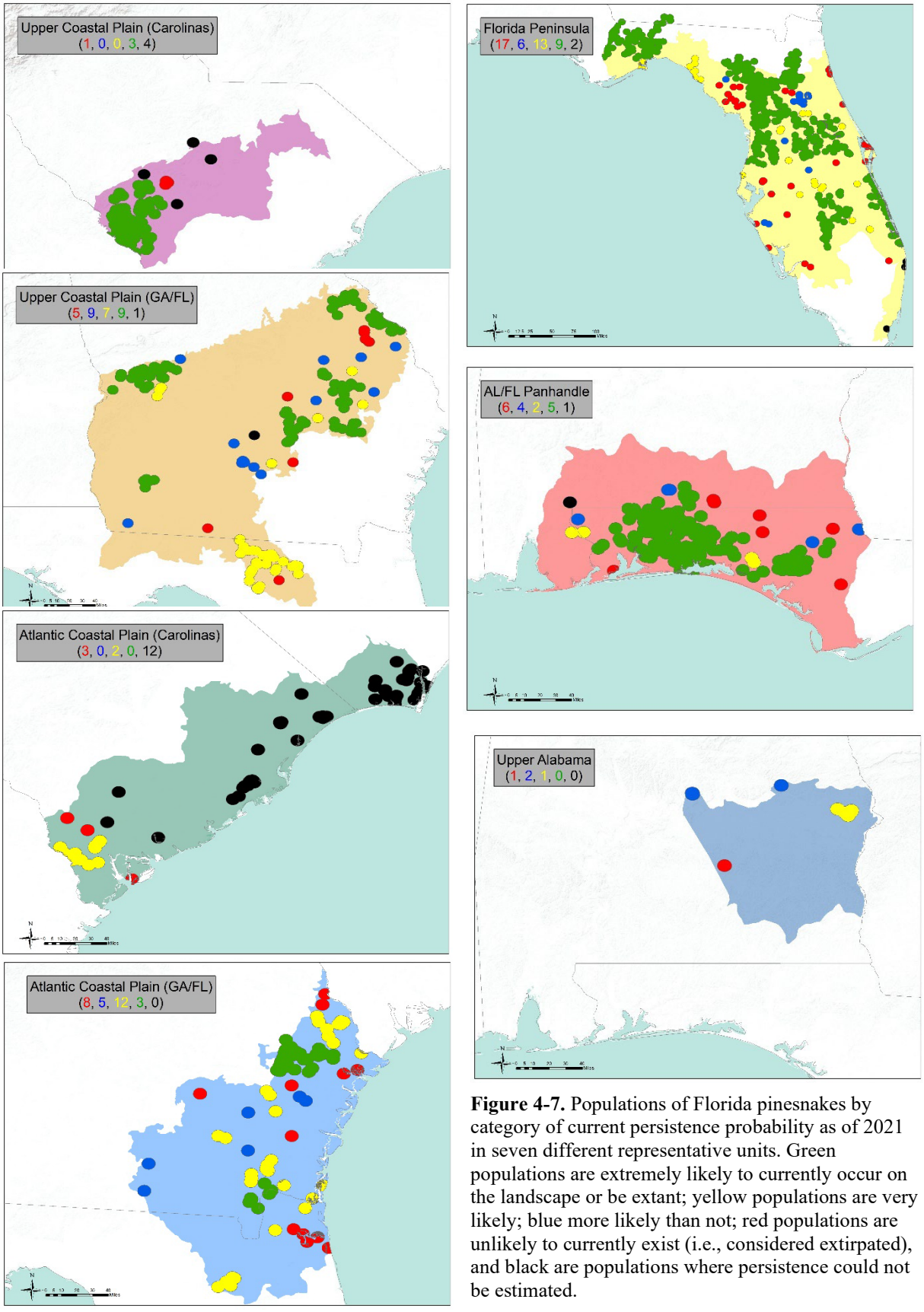

Figure 4-7. Populations of Florida pinesnakes by category of current persistence probability as of 2021 in seven different representative units. Green populations are extremely likely to currently occur on the landscape or be extant; yellow populations are very likely; blue more likely than not; red populations are unlikely to currently exist (i.e., considered extirpated), and black are populations where persistence could not be estimated.

Upper Coastal Plain (South Carolina)

We estimate this unit has 3 populations (75% of total populations in unit) that are extremely likely (≥95%, high resiliency) to persist on the landscape in 2021; 1 population that is likely extirpated; and 4 populations for which current persistence could not be determined. For these populations that current persistence could not be determined there is only 1 occurrence record associated with each of the populations and the records range from 1932-2006. The three populations that are extremely likely on the landscape appear to be highly resilient, as snakes continue to be regularly observed.

Upper Coastal Plain (GA/FL)

The Upper Coastal Plain (GA/FL) unit has the second greatest number of delineated populations (31) and we estimated that 30.0% of those populations are extremely likely to be persisting (≥95%, high resiliency) in 2021. Resilient populations are spread evenly across the representative unit. We estimate that this representative unit has likely lost at least five (16.7%) populations.

Atlantic Coastal Plain (Carolinas)

We estimate that 3 of the 5 populations for which we could estimate persistence probability in this unit are unlikely (<50%) to persist (extirpated) in 2021 and two populations are very likely to be persisting (≥80%, moderate to high resiliency) in 2021. There are 12 populations for which current persistence could not be determined; however, many of the occurrence records for these populations are fairly recent.

Atlantic Coastal Plain (GA/FL)

We estimate that the Atlantic Coastal Plain (GA/FL) unit has 20(71.4%) populations that are more likely that not (≥50% to persist in 2021, and of those, 3 populations (10.7%) are extremely likely to be persisting (>95%, high resiliency) in 2021. This representative unit has likely lost at least 8 (28.6%) of its historic, designated 28 populations.

Florida Peninsula

The Florida Peninsula unit had the most delineated populations (47), and the greatest number of occurrence records. We estimate this unit has 9 populations that are extremely likely to be persisting (≥95%, high resiliency); however, we estimated it also has the highest number of populations (17) that are likely extirpated (37.8%). The subspecies has experienced some range contraction due to the loss of populations in the southern extent of this unit. This unit has the single largest population in terms of number of occurrence records and designated area.

Alabama/Florida Panhandle

We estimate that this unit has 5 populations that are extremely likely to be persisting (≥95%, high resiliency) including one fairly large population that occupies a large area of suitable habitat. However, we estimate that this unit also has one of the highest percentage of populations (35.3%; 6 of 17) to be likely extirpated and the resilient populations are mainly located along the coast and not distributed evenly across the unit.

Upper Alabama Unit

We estimate that this unit has 3 populations that are more likely than not (≥50%) persisting in 2021 but no populations estimated to be extremely likely to be persisting (≥95%). Two of the populations are more likely than not on the landscape (50-79.9% probability of persistence; low resiliency), and 1 is very likely on the landscape (80-94.9% probability of persistence; moderate resiliency). We estimate this unit has likely lost at least 1 population and, overall, records for this subspecies within this unit are limited.

4.3 Summary of Current Conditions

Current resiliency, as measured by current persistence probabilities, for the Florida pinesnake has decreased from historical conditions. We considered a population to be highly resilient if it had a current persistence probability of ≥95%, defined as the probability that there is at least one Florida pinesnake in the population in 2021. We estimated that 30.8% of the delineated Florida pinesnake populations are likely extirpated and we estimate that 21.8% of delineated populations exhibit high resiliency (>95% probability of persistence) as of 2021. For currently extant populations, 71.7% (66/92) are very likely or extremely likely to persist (≥80% probability of persistence), and 31.5% (29/92) are extremely likely (>95% probability of persistence) to persist as of 2021. We estimate that the remaining 48.1% of total populations (69.5% of extant populations), span a wide range of estimated persistence probabilities related to habitat suitability, the number of other geographically proximate populations, the proportion of the population boundary on protected land, and the time since the last record for the subspecies within the area delineated for the population.

Current representation, as measured by the number and distribution of resilient populations (i.e., those above a certain persistence probability threshold) across representative units in the subspecies' range has decreased from historical conditions. To have high representation the subspecies must have highly resilient populations located in each of the representative units, and those occupied units should span the latitudinal and longitudinal extent of historical populations. Our estimated persistence probabilities for delineated Florida pinesnake populations does not fully meet those criteria as two representative units do not have populations in the highest persistence category and those units are on the northern and western portions of the subspecies range.

Current redundancy, as measured by using the current number and distribution of resilient populations within representative units and across the range of the subspecies has also declined from historical conditions. To have high redundancy the species needs to have multiple resilient populations within representative units and throughout its range. Range-wide, the number of populations more likely than not to currently persist has decreased by 30.8%, relative to the historical number of populations, and we estimate that all seven representative units have likely lost at least one historic, delineated population. Nonetheless, all representative units have multiple resilient populations, which meets the criteria for redundancy.

CHAPTER 5 – FUTURE CONDITIONS AND VIABILITY

In the previous chapters, we considered the Florida pinesnake's ecological needs, factors influencing its viability, and the estimated current condition of the subspecies. We now consider what the subspecies' future condition is likely to be. We apply our future scenarios to the concepts of resiliency, representation, and redundancy to describe the future viability of the Florida pinesnake by considering plausible scenarios and the subspecies' response to those conditions.

5.1 Methods for Estimating Future Condition

In evaluating future conditions for the Florida pinesnake, we considered stressors that may influence future viability of the subspecies and developed several plausible scenarios representing the potential effects of these stressors. We performed spatial analyses to predict changes in land cover and fire frequency under various levels of urbanization, sea level rise (SLR), and climate change induced changes to the burn windows that dictate opportunities for prescribed fire. Then, using the model framework developed for the current condition analysis described in Chapter 4, we created a stochastic simulation model that projected population persistence into the future as influenced by changes in habitat suitability (Figure 5-1). We summarized predicted patterns of population persistence to the years 2040, 2060, and 2080.

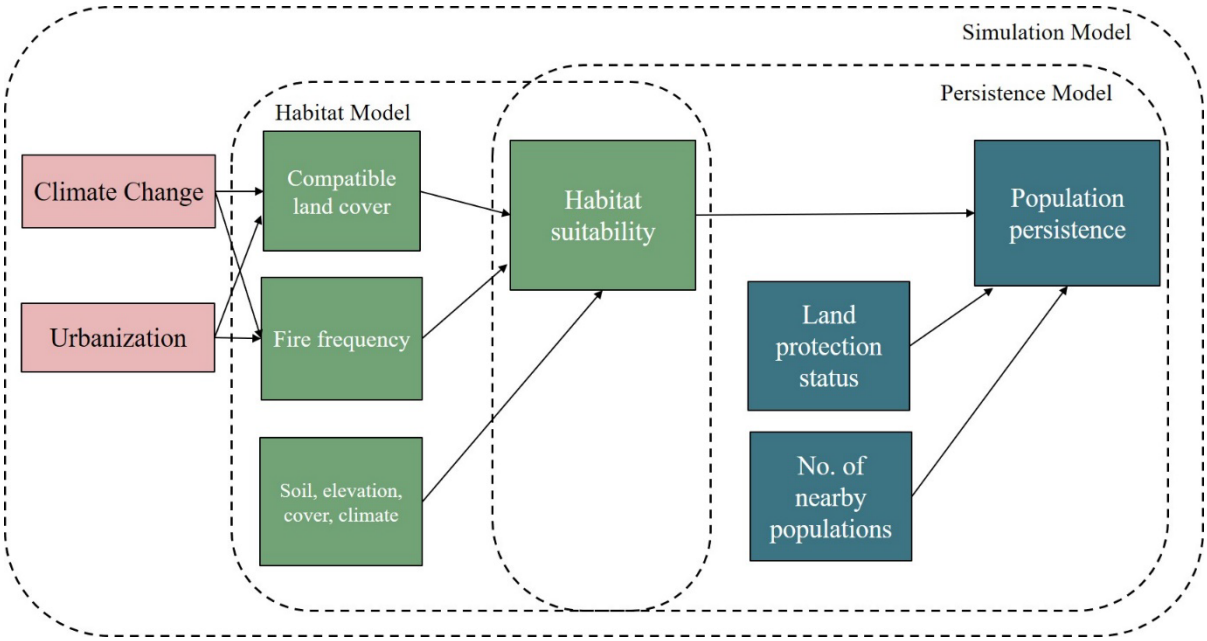

Figure 5-1. Conceptual model linking components included in the habitat suitability model (green), current persistence model (blue), and impacts of stressors (pink) in the simulation model for future conditions of the Florida pinesnake.

5.1.1 Stressors and Effects on Habitat Conditions

To predict future conditions, we selected stressors that were identified as factors influencing viability (Chapter 3; Figure 3-5), we anticipate will have an impact on Florida pinesnake populations in the future, and for which we could reasonably access needed covariate projections that could be integrated into the persistence model. We selected urbanization, SLR, and predicted changes to fire frequency due to climate change as the stressors we could model through predicted effects on land cover and the associated habitat suitability model (Crawford et al. 2020a, entire). Changes in habitat suitability ultimately influenced population persistence, as predicted by the stochastic simulation model (Section 5.1.3; Figure 5-1; Appendix A). We note, there are other stressors potentially affecting Florida pinesnake populations, such as invasive species, persecution, and disease, that will continue into the future, but the availability of data and our understanding of these stressors' effects precluded their integration into the future persistence model. Below we describe the data sources we used for capturing stressors and spatial analysis methods used in ArcGIS to model the effects of stressors on habitat conditions.

We captured the extent and rate of urbanization using the Slope, Land cover, Exclusion, Urbanization, Transportation, and Hillshade (SLEUTH) model, which simulates patterns of urban expansion across the southeastern United States based on observations of past urban growth and transportation networks, including the sprawling, fragmented, "leapfrog" development, which has been the dominant form of development in the Southeast (Terando et al. 2014, entire). The SLEUTH model predicts the probability of urbanization ranging from 0-100%, with higher probabilities indicating areas more likely to be developed. The model specifies a 0% probability of urbanization for protected areas (e.g., National Forests, State-owned conservation lands); however, new conservation lands have been created by government agencies and other partners since the model was developed. Therefore, we set the probability of urbanization to 0 for any cell that overlapped our raster of currently protected areas. We used SLEUTH projections from 2040, 2060, and 2080, and linearly interpolated the annual extent of urbanization between those periods. We modeled the effect of habitat loss from urbanization by a given year by removing any area currently classified as compatible land cover that overlapped areas likely to be urbanized above a certain probability threshold (see scenario descriptions in Section 5.1.2). We then recalculated the proportion of compatible land cover for each population that subsequently affected overall habitat suitability.

The climate change projections used in this SSA report are based on Representative Concentration Pathway (RCP) scenarios. The RCPs are the current set of scenarios used for generating projections of climate change. We used climate change projections based on RCP 4.5 and RCP 8.5, the "medium-low" and "highest" scenarios, respectively. We captured potential habitat loss due to inundation from SLR. We obtained projections of SLR associated with RCPs from the Sixth IPCC Assessment Report (IPCC 2021, pp. 21–23) and used SLR inundation maps from the National Oceanic and Atmospheric Administration (NOAA 2021, unpaginated). For

each projected height of SLR associated with 2040, 2060, and 2080, we created a raster of inundated areas. Similar to modeling urbanization, we linearly interpolated the annual extent of inundation due to SLR between periods. We reduced the amount of compatible land cover for populations if currently compatible land cover was predicted to be inundated by SLR, given a certain inundation level and year.

We also wanted to capture the effects of urbanization on fire frequency to account for fire exclusion/suppression that often occurs in areas adjacent to developed lands and the effects of climate change on windows of time suitable for prescribed burns. Studies have found evidence of fire exclusion/suppression in habitats within 600 m to 5 km (0.4 to 3.1 miles) of urban areas (Theobald and Romme 2007, entire; Pickens et al. 2017, p. 105). Therefore, we chose an intermediate value of 3.2 km (2 miles) to capture the interaction between urbanization and fire frequency. Using the predicted urbanized areas at each time period and urbanization probability level, we identified areas within 3.2 km of present and projected urban boundaries. We then projected future reductions in fire frequency by applying a distance-weighted reduction to any cell within 3.2 km of urban areas. Using this approach, fire frequency for any cell overlapping future urban areas was reduced by 100% (equaled 0) and any cell more than 3.2 km away from urban areas was reduced by 0% (unchanged). Any cell between 0 and 3.2 km away from urban areas was reduced by a percentage proportional to its distance; for example, a cell 1.6 km (1 mile) from an urban area was reduced by 50%. We then recalculated the mean fire frequency for each population. To account for changes in fire frequency from climate change, we used U.S. Geological Survey historical and future prescribed burn windows for the Southeast United States (Kupfer et al. 2021, unpaginated). This dataset included observed mean proportion of days with suitable weather conditions for burning by month for the periods of 2000-2009 and 2010-2017 and future projections of decadal mean proportions of days suitable for burning by season. To correspond with the fire frequency predictor used in our model of habitat suitability, which used annual fire detections from 2001-2016, for each cell we calculated an annual weighted average of the historical 2000-2009 and 2010-2017 data and weighted monthly data according to number of days per month. For future burn windows, we used data for each decade leading up to 2040, 2060, and 2080, and for each cell we calculated the decadal averages across seasons weighted by number of days per season. We calculated population means for each burn window data set and used the population averages to calculate proportional changes in fire windows for 2040, 2060, and 2080. We assumed that these would result in the same proportional changes in fire frequency and reduced each population's mean fire frequency accordingly, which subsequently affected overall habitat suitability. Two populations did not have data for observed fire window, so we used values from the nearest populations.

5.1.2 Scenario development

We developed six different plausible scenarios to simulate future conditions (Table 5-1). In each of the scenarios we varied the levels of urbanization and used two different RCPs for climate

change. For all the scenarios we assumed management for the subspecies would remain at the current status quo. For status quo management we assumed: (1) the amount of protected area for each population will remain constant through time (e.g., protected land will not be sold or urbanized); (2) compatible land cover will not increase; and (3) fire frequency will only change with urbanization and climate change as described in section 5.1.1 (e.g., managers will not increase prescribed fire).

Table 5-1. List of scenarios used to predict future conditions for the Florida pinesnake, showing levels of urbanization, and climate change RCP.

Scenario
Low Urbanization/RCP 4.5 (best case)
Low Urbanization/RCP 8.5
Medium Urbanization/RCP 4.5
Medium Urbanization/RCP 8.5
High Urbanization/RCP 4.5
High Urbanization/RCP 8.5 (worst case)

For the scenarios, when analyzing a low rate of urbanization, we selected areas with a greater than 90% probability of being urbanized (only areas with high certainty of development), for a medium rate of urbanization we selected areas with a greater than 50% probability of being urbanized, and for a high rate of urbanization we selected areas with a greater than 10% probability of being urbanized (includes areas with a low probability of development). Where we used RCP 4.5, we used a SLR of 1ft. at years 2040 and 2060 and a SLR of 2 ft. at year 2080. For RCP 8.5, we used a SLR of 1ft. at years 2040 and 2060 and a SLR of 3 ft. at year 2080. Burn window reductions were also assessed for both an RCP 4.5 and RCP of 8.5.

5.1.3 Stochastic simulation model

To predict the probability of persistence for each population in the future, we developed a stochastic simulation model that was based on the current probability of persistence and future predicted changes in habitat suitability.

We built a multi-loop simulation model (McGowan et al. 2014, entire) that allowed us to simulate thousands of replicates of each population under the six different scenarios. This approach accounted for random year-to-year stochasticity as well as uncertainty around rates (i.e., annual persistence probability) estimated from the current persistence model. The model looped through 10,000 iterations for each of the scenarios. In each iteration, it looped through each of the 133 Florida pinesnake populations within the analysis area and simulated persistence from the present (2021) to 2080.

The core of the simulation model was the current conditions persistence model (see 4.1.2). For each population, the model used current persistence probability to simulate whether it persisted or became extirpated in the next year (2022). From 2023 to 2080, the probability a population persisting from one year to the next was a function of its habitat suitability, proportion of protected land, and number of nearby populations, given the conditions of each scenario. For each year in a model iteration, it selected the annual values of land cover and fire frequency that were influenced by rates of urbanization and climate change. It then used these conditions, along with all other predictors used in the habitat suitability model (e.g., soil drainage, local elevation) that did not change in any scenario, to calculate mean habitat suitability for each population and year. Finally, it used the new values of a population's habitat suitability, along with the proportion of protected land and number of nearby populations, to estimate the probability a population persists each year in the future through 2080.

For this model, the primary output was the probability a population persists at a given year in the future through 2080, which we call "future persistence probability." We calculated the future persistence probability for each population as the proportion of model iterations where the population was persisting at a given year. The complement of future persistence probability can be interpreted as the probability a population has become extirpated by a given year. These future persistence probabilities for populations are directly comparable to the current persistence probabilities estimated in Chapter 4. We characterized future conditions similarly to current conditions by summarizing the number of populations at or above certain persistence probability thresholds (50, 80, and 95%) in each representative unit and range-wide, given each of the six scenarios. Like in the current conditions analysis, in addition to a specific population's persistence probability, each model iteration recorded the number of populations persisting at each time step in each representative unit and range-wide. We used all model iterations to calculate the mean (the most likely prediction) and 95% credible interval for the number of persisting populations, in each representative unit and range-wide, each year. Similar to results for current conditions, the mean number of persisting populations approximately equals the number of populations with a 50% or greater persistence probability. We summarized these outputs for three future time horizons: 2040, 2060, and 2080. For more details on methods used in the future conditions analysis, see Appendix A. Importantly, a feature of this model is that it does not have any processes for colonization or recolonization. Therefore, persistence probabilities are invariably going to be <1 for each population, this means that it is inevitable that the model will predict some loss of populations over time. To address this inherent feature of the model, we also ran the model forward to 2040, 2060, and 2080 assuming no changes in urbanization or climate and SLR. This creates a null expectation for model projected population losses against which we could compare the models with projected changes in urbanization, climate, and SLR. In our results we refer to this as the null model.

5.2 Future Condition Results

5.2.1 Summary of analysis

Changes in habitat conditions used to predict future persistence varied slightly between scenarios (Table 5-2). Projected high urbanization and RCP 8.5 SLR across the Southeast for the year 2080 is shown in Figure 5-2. Under current conditions, the average population had compatible land cover of 43% (range = 2 – 83%), which decreased between 9% (best case scenario) and 12% (worst case scenario) on average across all scenarios (Table 5-2). In terms of area, the average population had 233 km^2 (range = 2 – 6323 km^2) of compatible land cover under current conditions, which was reduced by 57 – 72 km^2 on average across future scenarios (Table 5-2). While some populations experienced no loss of land cover, others were predicted to lose as much as 43.5% of compatible land cover due to urbanization and climate change.

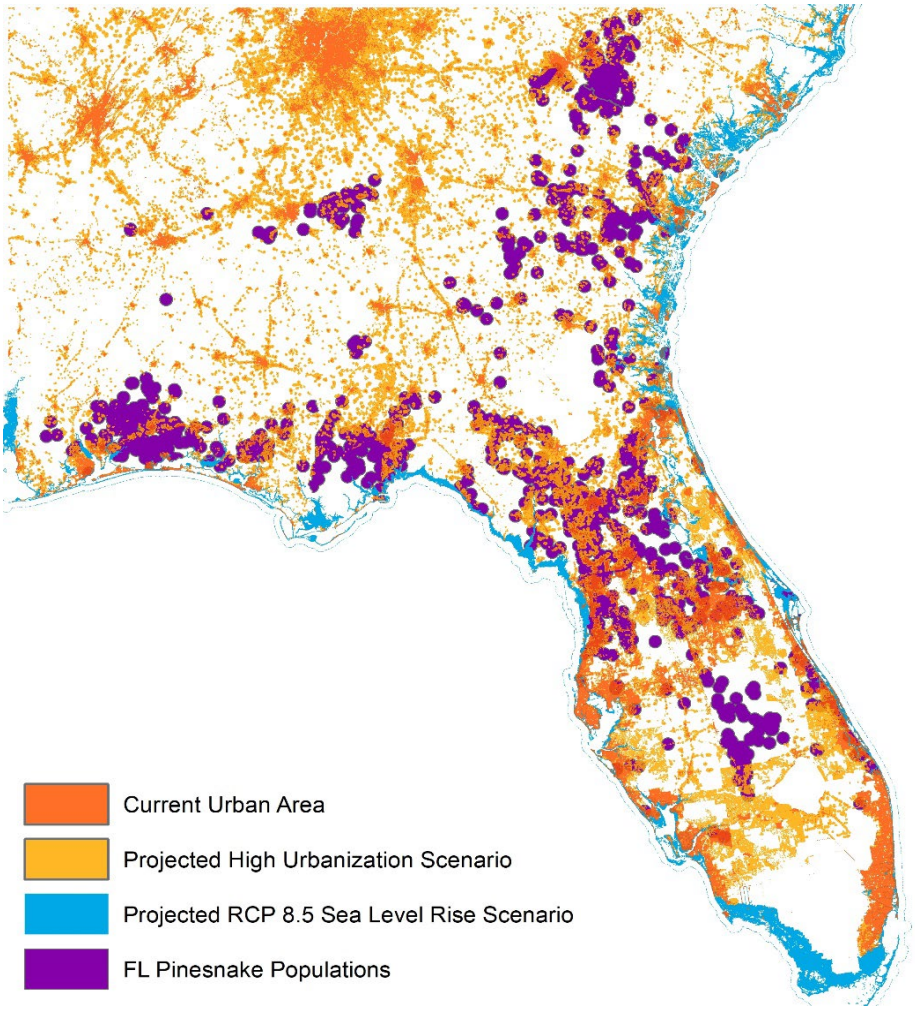

Figure 5-2. Projected high urbanization and SLR under RCP 8.5 for the year 2080 in the Southeast, United States. Certain Florida pinesnake populations (purple), within the analysis area, were predicted to experience substantial habitat loss due to urbanization and SLR.

Burn windows, which averaged 293 days/year (range = 242 – 329 days/yr) under current conditions, were reduced by 41 – 71 days on average across all scenarios (Table 5-2). These shorter burn windows, in combination with urbanization, resulted in fire frequencies (mean = 3.5%; range = 0-16.2% under current conditions) being reduced by 2.0 – 2.3% on average across scenarios (Table 5-2). Using the reduced fire frequencies, the total number of burns between 2021 – 2080 for an average population is expected to be reduced by 1 with a range of approximately 0 to 3 fewer burns across all populations (Table 5-2), compared to if burn frequency was not reduced from current levels (mean = 2; range = 0 – 10 burns from 2021-2080 under current levels). Finally, under current conditions, the average population had a mean HSI of 20.6% (Table 4-1), which decreased between 4.6% (best case scenario) and 5.7% (worst case scenario) on average across all scenarios (Table 5-2).

Table 5-2. Per-population predicted changes in habitat variables (compatible land cover, fire frequency, burn windows, burn days, and Habitat Suitability Index [HSI]) between the present and 2080. Means (top value in each cell) represent the average change across populations, and minimums and maximums (bottom values in each cell) represent the lowest and highest changes, respectively, predicted for a single population.

Habitat variable	Scenario					
	Low Urbanization /RCP 4.5	Med Urbanization /RCP 4.5	High Urbanization /RCP 4.5	Low Urbanization /RCP 8.5	Med Urbanization /RCP 8.5	High Urbanization /RCP 8.5
Compatible land cover (%)	-8.9 (-38.2, 0.0)	-10.3 (-40.9, 0.0)	-12.0 (-43.5, 0.0)	-9.0 (-38.2, 0.0)	-10.4 (-40.9, 0.0)	-12.1 (-43.5, 0.0)
Compatible land cover (km^2)	-57 (-2490, 0)	-64 (-2743, 0)	-72 (-3037, 0)	-57 (-2490, 0)	-64 (-2743, 0)	-72 (-3038, 0)
Fire frequency (% of years burned)	-2.0 (-6.0, 0.0)	-2.0 (-6.2, 0.0)	-2.1 (-6.9, 0.0)	-2.1 (-6.4, 0.0)	-2.2 (-6.7, 0.0)	-2.3 (-7.4, 0.0)
Burn window days per year	-41 (-67, -24)	-41 (-67, -24)	-41 (-67, -24)	-71 (-102, -48)	-71 (-102, -48)	-71 (-102, -48)
Total burns (2021-2080)	-0.87 (-2.74, 0.01)	-0.90 (-2.79, 0.01)	-0.94 (-3.14, 0.01)	-0.90 (-2.80, 0.01)	-0.93 (-2.85, 0.01)	-0.97 (-3.18, 0.01)
HSI (%)	-4.6 (-35.4, 0.0)	-5.0 (-38.1, 0.0)	-5.5 (-39.9, 0.0)	-4.8 (-35.4, 0.0)	-5.2 (-38.1, 0.0)	-5.7 (-39.9, 0.0)

5.2.2 Future Population Resiliency

Using the simulation model, we estimated the future persistence probability for each of the current 133 total populations in the analysis area through the year 2080. We also present our results in terms of the 92 currently extant populations, since 41 populations were identified as extirpated in the current condition analysis. As with current persistence we could not analyze 2 populations within the analysis area nor the 18 populations outside the analysis area. All six scenarios yielded nearly identical predictions of the number and percentage of resilient populations. These patterns were seen across all future time horizons (2040, 2060, and 2080). To simplify the reporting of our results since there was so little difference between the scenarios, we only show the results for the Low Urbanization/RCP 4.5 (best case) and High Urbanization/RCP

8.5 (worst case) in our tables. We followed similar steps as in the current condition analysis to summarize population resiliency by reporting the number of populations at each persistence threshold under these two future scenarios in years 2040, 2060, and 2080 (Table 5-3; Figure 5-4). In Figure 5-3, we display the distribution of populations by category of persistence probability across the subspecies' range under current conditions, the Low Urbanization/RCP 4.5 (best case) scenario in 2080, and the High Urbanization/RCP 8.5 (worst case) scenario in 2080.

For all scenarios, we predicted that future population persistence decreased from current conditions and most populations that fell within the extremely likely on landscape (≥95%, high resiliency) threshold under current conditions were predicted to have lower persistence probabilities in the future. The number and percentage of populations likely to be extirpated (<50%) increased for all scenarios and future time horizons, relative to current conditions. For example, the percentage of total populations likely to be extirpated (<50%) in 2021 is 30.8% of the total populations, leaving 92 currently extant populations. Under the Low Urbanization/RCP 4.5 scenario by 2040, that total percentage increased to 42.9%, which represents 16 additional populations predicted to be extirpated. For the currently extant populations (n=92), this represents 17.4% of populations, covering 3.9% of the total currently occupied range area (Table A-3). By 2060, the percentage increased to 58.6% total populations over time (an additional 21 populations predicted to be extirpated), and by 2080 we predicted the loss of an additional 18 populations for a total of 96 populations over time (72.2% of total populations over time) predicted to become extirpated. For the currently extant populations, this represents 40.2% of populations covering 11% of the total currently occupied range area (Table A-3). These numbers are very similar for the High Urbanization/RCP 8.5 (worst case) scenario. For the High Urbanization/RCP 8.5 (worst case) the model predicted 5 populations in the extremely likely on landscape (≥95%, high resiliency) category of persistence in 2040, covering 45.3% of the total currently occupied range area, and only 1 in 2080, covering 24.6% of the total currently occupied range area (Table A-3).

We also predicted future population persistence under the null model, where SLR, urbanization, and burn windows remained at current levels (i.e., HSI remained at the values used in the current conditions model). Population persistence was projected to be very similar under the null model as compared to the six scenarios of climate change/urbanization. Specifically, across all populations over time (n=133), the percentage of populations likely to be extirpated was very similar to the Low Urbanization/RCP 4.5 scenario in 2040 (43.6% vs 42.9%) and 2060 (57.1% vs 58.6%), and slightly lower than the Low Urbanization/RCP 4.5 scenario in 2080 (66.9% vs 73.7%). Additionally, the null model predicts 2 populations in the highest category of persistence, compared to 1 population predicted under the worst-case and best-case scenarios in 2080.

The very small difference in resiliency between the scenarios was consistent with the minimal differences between the scenarios in the amount of land cover loss and change in habitat

suitability for populations (Table 5-2). The similarity in results between the stressor scenarios and the null model suggest that future population declines predicted to occur are primarily a consequence of population persistence probabilities being <1 for each population and no processes included in our model for colonization or recolonization, rather than a consequence of decreased habitat suitability due to climate change and urbanization.

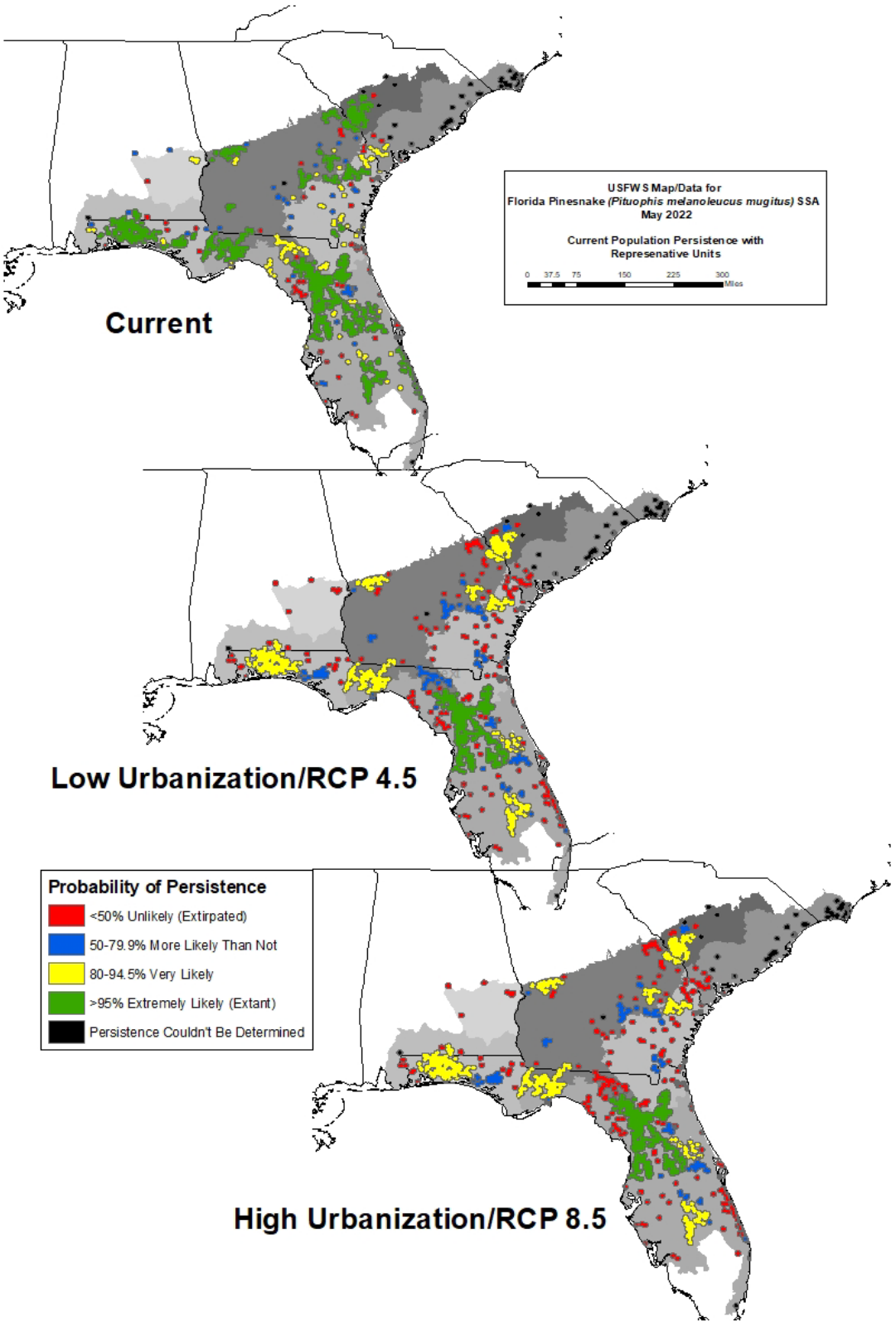

Figure 5-3. Spatial distribution of all Florida pinesnake populations over time (n=133) by category of persistence probability across the subspecies' range under current conditions, the Low Urbanization/RCP 4.5 (best case) scenario in 2080, and the High Urbanization/RCP 8.5 (worst case) scenario in 2080. Representative units are differentiated in gray.

Table 5-3. Distribution of Florida pinesnake populations (n =133) in each category of population persistence in 2040, 2060, and 2080, for two scenarios. Numbers are reported as the number of populations in each category and as the cumulative number of populations at or above each threshold with the percentage of total in parentheses. Summary for the currently extant populations (n=92) is provided in Table A-3 in the appendix.

Population persistence category	**Scenario**					
	Current Condition		Low Urbanization/RCP 4.5 (best case)		High Urbanization/RCP 8.5 (worst case)	
Year - 2040	Each category (% of total)	Cumulative (% of total)	Each category (% of total)	Cumulative (% of total)	Each category (% of total)	Cumulative (% of total)
*Extremely Likely on Landscape (Extant) 95-100%	29 (21.8%)	29 (21.8%)	6 (4.5%)	6 (4.5%)	5 (3.8%)	5 (3.8%)
*Very Likely on Landscape 80-94.9%	37 (27.8%)	66 (49.6%)	24 (18.0%)	30 (22.3%)	25 (18. %)	30 (22.5%)
*More Likely than Not 50-79.9%	26 (19.5%)	92 (69.2%)	46 (34.6%)	76 (57.1%)	45 (33.8%)	75 (56.3%)
Unlikely (Extirpated) < 50%	41 (30.8%)	-	57 (42.9%)	-	58 (43.6%)	-
Year - 2060	Current in Each Category	Current Cumulative	Each Category	Cumulative	Each Category	Cumulative
*Extremely Likely on Landscape (Extant) 95-100%	29 (21.8%)	29 (21.8%)	3 (2.3%)	3 (2.3%)	3 (2.3%)	3 (2.3%)
*Very Likely on Landscape 80-94.9%	37 (27.8%)	66 (49.6%)	12 (9.0%)	15 (11.3%)	12 (9.0%)	15 (11.3%)
*More Likely than Not 50-79.9%	26 (19.5%)	92 (69.2%)	40 (30.1%)	55 (41.4%)	38 (28.6%)	53 (39.8%)
Unlikely (Extirpated) < 50%	41 (30.8%)	-	78 (58.6%)	-	80 (60.2%)	-
Year - 2080	Current in Each Category	Current Cumulative	Each Category	Cumulative	Each Category	Cumulative
*Extremely Likely on Landscape (Extant) 95-100%	29 (21.8%)	29 (21.8%)	1 (0.8%)	1 (0.8%)	1 (0.8%)	1 (0.8%)
*Very Likely on Landscape 80-94.9%	37 (27.8%)	66 (49.6%)	10 (7.5%)	11 (8.3%)	10 (7.5%)	11 (8.3%)
*More Likely than Not 50-79.9%	26 (19.5%)	92 (69.2%)	26 (19.5%)	37 (27.8%)	24 (18.0%)	35 (26.3%)
Unlikely (Extirpated) < 50%	41 (30.8%)	-	96 (72.2%)	-	98 (73.7%)	

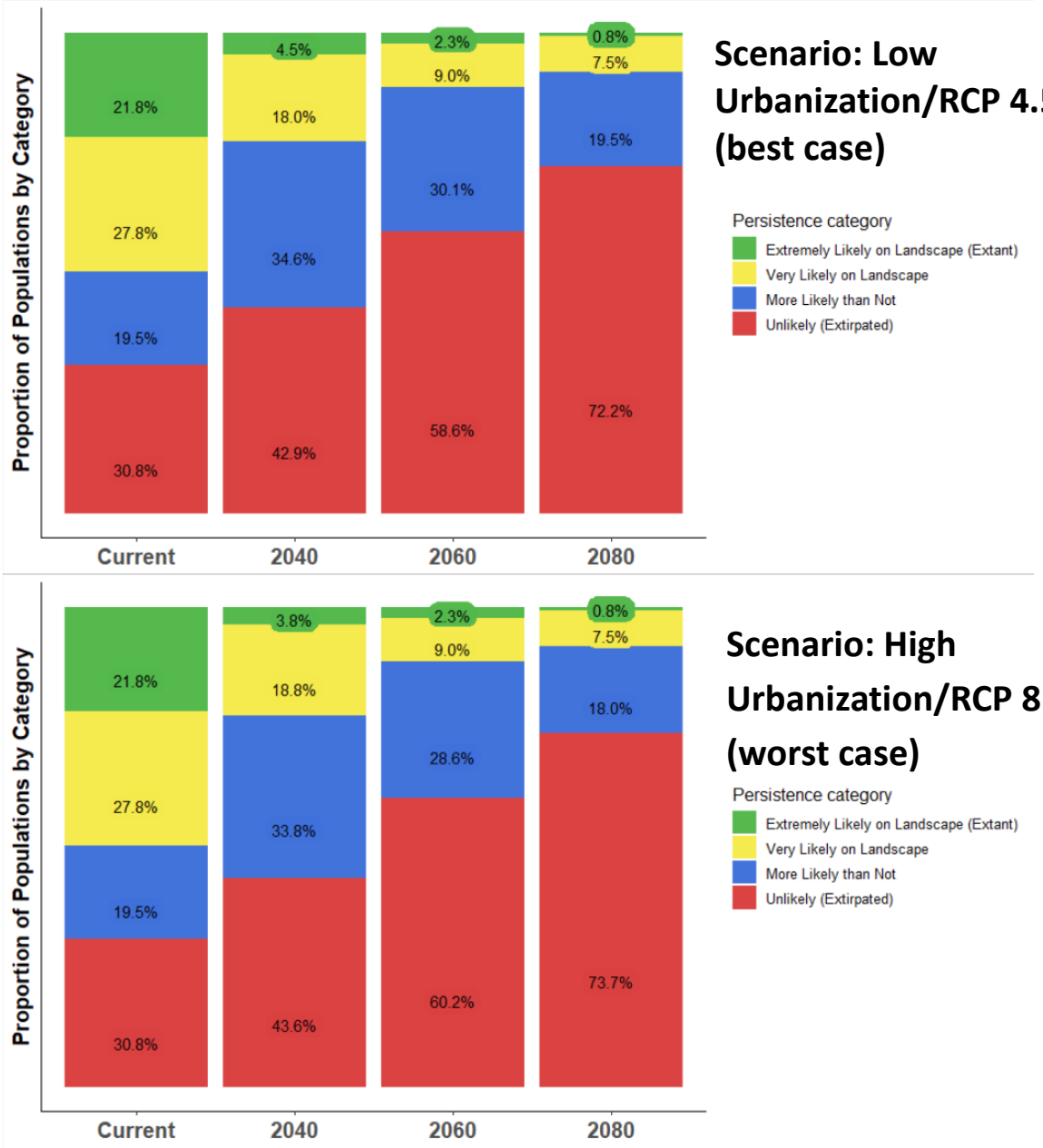

Figure 5-4. Percent of total Florida pinesnake populations over time in each category of population persistence in 2040, 2060, and 2080, for two scenarios, showing how the percent of populations in each of the categories is predicted to change over time. Populations presented are the total populations over time, including all populations considered extirpated in the current condition analysis. Populations in the highest persistence category in green are considered to have the highest resilience, followed by populations represented by yellow and then blue and then red. Proportion of currently extant populations (n=92) and current range area occupied within each resiliency category are presented in Figure A-4.

5.2.3 Future Species Representation and Redundancy

We summarized trends in the future condition of the Florida pinesnake for representation and redundancy using the number of populations likely to persist and their spatial distribution.

Specifically, we report both the persistence probability thresholds, in each of the representative units and across the subspecies' range, in 2040, 2060, and 2080 for all populations over time (n=133) (Table 5-4; Figure 5-5), as well as the mean number of populations predicted to persist with 95% credible intervals (n=133) (Tables 5-5). We report the persistence probability thresholds, in each of the representative units and across the subspecies' range, in 2040, 2060, and 2080 for extant populations only (n=92) in the appendix (Figure A-4). The mean numbers of populations persisting, as predicted directly from the simulation model, are relatively close (± 2 populations) to the number of populations above a 50% persistence probability for all of the representative units.

As the number of populations predicted to persist within a unit decreases, we can assume that the entire representative unit is at risk. Therefore, we interpret a high risk of loss of representation when a representative unit has zero populations above a certain persistence threshold, moderate risk when a unit has one or two populations above a certain threshold, and lower risk when a unit has three or more populations above a certain threshold. We discuss the risk for particular units in relative terms to other units.

Overall, we predicted future representation and redundancy to decrease from current conditions. The number of resilient populations and the number of representative units with resilient populations decreased under all the scenarios. In our current condition analysis, we determined that 5 of our 7 representative units have at least 3 populations extremely likely to be on the landscape (persistence probability ≥ 95%, high resiliency), and the other two units (Atlantic Coastal Plain (Carolinas) and Upper Alabama) had at least 1 population very likely to be on the landscape (persistence probability ≥ 80%, moderate and high resiliency). By 2080, we estimate that only 1 of the 7 representative units will still contain one population at the ≥ 95% threshold, for the "worst case" High Urbanization/RCP 8.5 scenario. Two representative units are predicted to have zero populations at or above the more likely than not to persist threshold (≥ 50%) by 2080 (Table 5-4; Figure 5-5). Thus, the Atlantic Coastal Plain (Carolinas) and the Upper Alabama Unit are predicted to be at the highest risk of losing populations, and the greatest potential for losing representation of the subspecies.

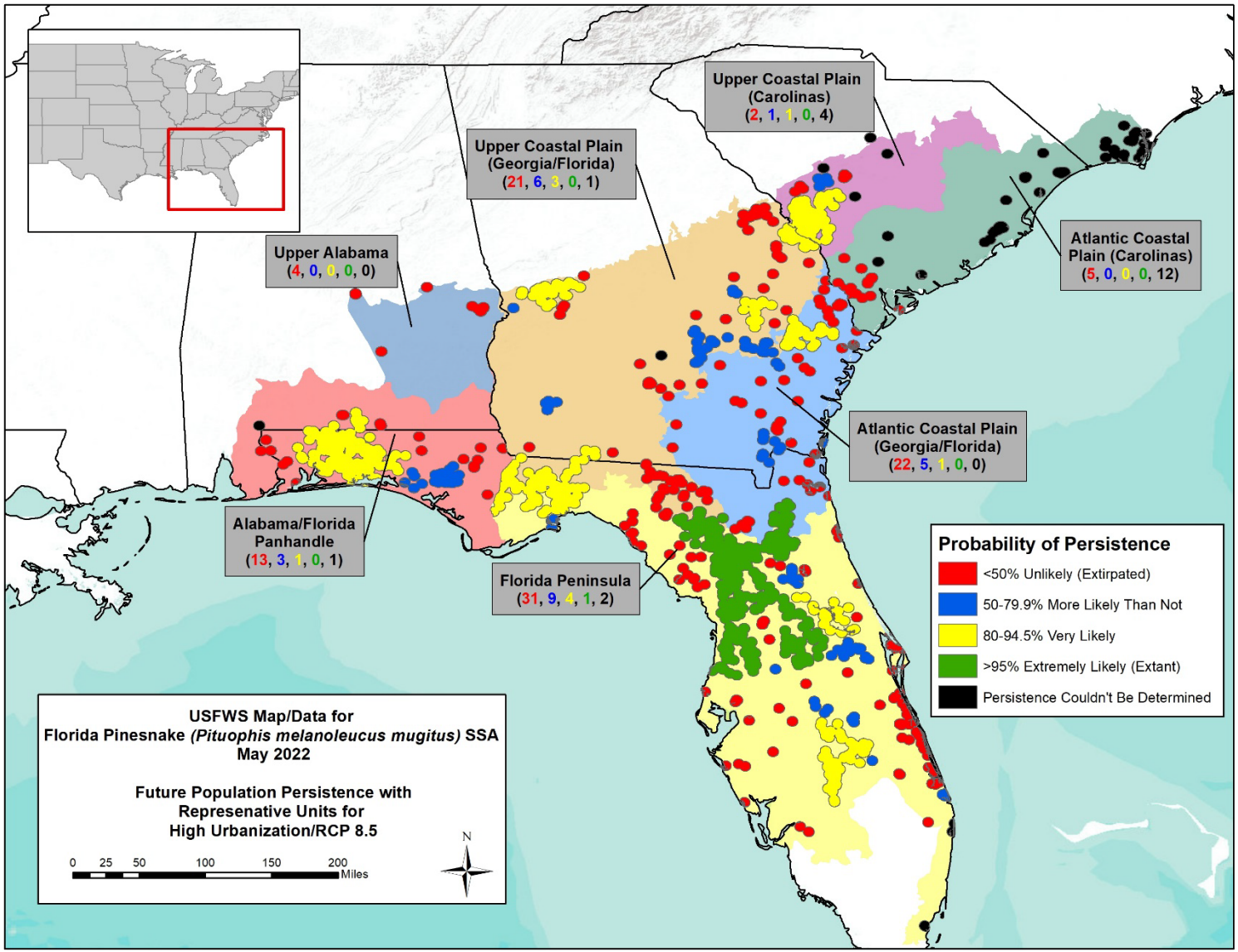

Figure 5-5. Redundancy, measured by the number of populations above the 50% persistence probability threshold, within representative units for the Florida pinesnake, in the year 2080, under the High Urbanization/RCP 8.5 scenario. Numbers reported include all 133 populations over time.

In addition to summarizing future conditions using the number of populations meeting a certain persistence probability threshold, we present the mean number of populations still persisting (and 95% credible intervals), as predicted directly from the simulation model (Table 5-5). We used cell shadings in Table 5-5 to highlight relative risks to losing representation within a unit. As discussed above, we again see that the Atlantic Coastal Plain (Carolinas) and Upper Alabama were the representative units with the highest risk of losing representation, relative to other units. The Atlantic Coastal Plain (Carolinas) had a moderate risk of losing representation as the mean number of populations was 3 in 2040 and decreased overtime. We note that this unit includes populations that we could not include in our analysis. By 2080 the AL/FL Panhandle unit was at moderate risk of losing representation as well.

Into the future, the Florida pinesnake is predicted to experience a decrease in latitudinal and longitudinal variability (i.e., a range contraction) relative to its historic range. Future predictions show that populations at or above the 50% and 80% persistence threshold will likely be distributed throughout the core of the subspecies' range (Figure 5-5) and cover >50% of the occupied range of the species under both scenarios (Figure A-4). However, these more resilient populations will continue to exhibit a degree of clustering, which will likely leave large areas lacking resilient populations (e.g., upper Alabama and coastal South Carolina; Figure 5-5).

Redundancy, as measured by the number of populations above the 50, 80, and 95% persistence thresholds (Table 5-4) and by the mean number of populations persisting (Table 5-5; Figure 5-6), decreased in all scenarios, time periods, and representative units, relative to current conditions. All scenarios yielded nearly identical predictions of the number of persisting populations. This is not surprising since there were only minimal differences between the scenarios in the amount of land cover loss and HSI for populations.

Looking at redundancy within the representative units, we see that all representative units except Atlantic Coastal Plain (Carolinas) and Upper Alabama were predicted to have at least 1 population above the 80% persistence probability threshold and at least 2 populations above the 50% threshold by 2080 under the High Urbanization/RCP 8.5 scenario (Table 5-4). The Florida Peninsula currently has the highest number of populations and was predicted to retain the highest level of redundancy in the future. Populations that occur on protected areas with the highest HSI in the subspecies' range were predicted to persist over time due to the favorable conditions.

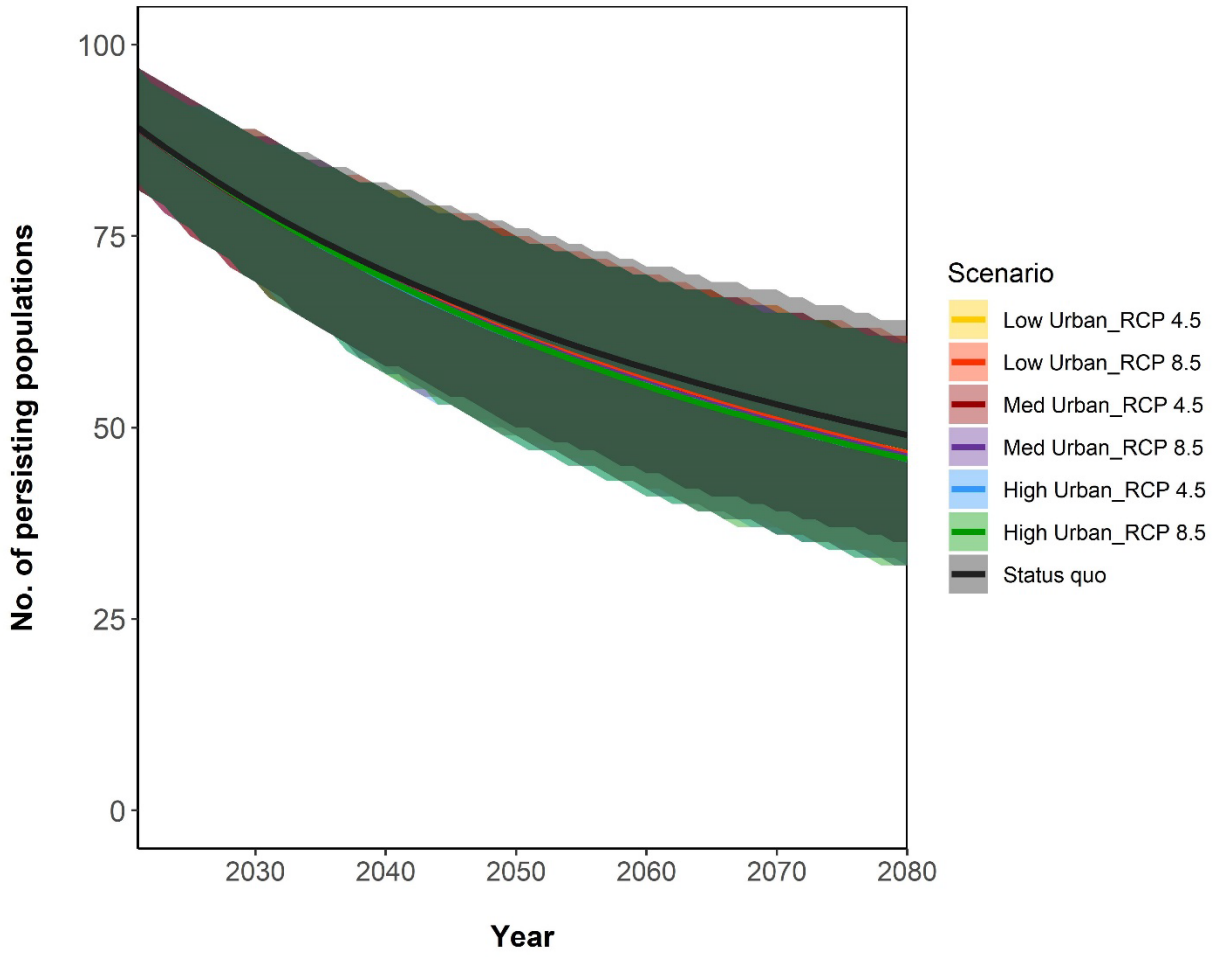

Figure 5-6. Predicted mean (± 95% credible intervals) number of persisting Florida pinesnake populations between the present year (2021) and 2080, given two future climate change scenarios and three scenarios of urbanization.

Table 5-4. Number of Florida pinesnake populations (n=133) at or above the 50, 80, and 95% threshold of persistence probability for current conditions and two scenarios at years 2040, 2060, and 2080, within each representative unit and range-wide for all populations over time. Cell shadings: Red – 0 populations meeting the persistence probability threshold; Orange – 1 or 2 populations within the persistence probability threshold. Summary information for currently extant populations is presented in Table A-4 in appendix.

Representative Unit	Scenario											
	Current Conditions				Low Urbanization/RCP 4.5 (best case)				High Urbanization/RCP 8.5 (worst case)			
Year - 2040	<50	≥50	≥80	≥95	<50	≥50	≥80	≥95	<50	≥50	≥80	≥95
Upper Coastal Plain (South Carolina)	1	3	3	3	1	3	2	1	1	3	2	1
Upper Coastal Plain (GA/FL)	5	25	16	9	9	21	7	1	9	21	7	0
Atl. Coastal Plain (Carolinas)	3	2	2	0	3	2	0	0	3	2	0	0
Atl. Coastal Plain (GA/FL)	8	20	15	3	15	13	5	1	15	13	5	1
FL Peninsula	17	28	22	9	20	25	12	2	20	25	12	2
AL/FL Panhandle	6	11	7	5	8	9	4	1	9	8	4	1
Upper Alabama	1	3	1	0	1	3	0	0	1	3	0	0
Range-wide	41	92	66	29	57	76	30	6	58	75	30	5
Year - 2060												
Upper Coastal Plain (South Carolina)	1	3	3	3	2	2	1	0	2	2	1	0
Upper Coastal Plain (GA/FL)	5	25	16	9	15	5	4	0	15	15	4	0
Atl. Coastal Plain (Carolinas)	3	2	2	0	5	0	0	0	5	0	0	0
Atl. Coastal Plain (GA/FL)	8	20	15	3	19	9	2	1	19	9	2	1
FL Peninsula	17	28	22	9	24	21	7	1	25	20	7	1
AL/FL Panhandle	6	11	7	5	10	7	1	1	10	7	1	1
Upper Alabama	1	3	1	0	3	1	0	0	4	0	0	0
Range-wide	41	92	66	29	78	55	15	3	80	53	15	3
Year - 2080												
Upper Coastal Plain (South Carolina)	1	3	3	3	2	2	1	0	2	2	1	0
Upper Coastal Plain (GA/FL)	5	25	16	9	20	10	3	0	21	9	3	0
Atl. Coastal Plain (Carolinas)	3	2	2	0	5	0	0	0	5	0	0	0
Atl. Coastal Plain (GA/FL)	8	20	15	3	22	6	1	0	22	6	1	0
FL Peninsula	17	28	22	9	30	15	5	1	31	14	5	1
AL/FL Panhandle	6	11	7	5	13	4	1	0	13	4	1	0
Upper Alabama	1	3	1	0	4	0	0	0	4	0	0	0
Range-wide	41	92	66	29	96	37	11	1	98	35	11	1

Table 5-5. Mean number of Florida pinesnake populations (n=133) (Lower, Upper 95% credible intervals) predicted to persist in 2040, 2060, and 2080, for two scenarios, across each representative unit and range-wide.

Representative Unit	**Scenario**		
Year - 2040	Current Condition	Low Urbanization/RCP 4.5 (best case)	High Urbanization/RCP 8.5 (worst case)
Upper Coastal Plain (South Carolina)	3 (3, 4)	3 (1, 4)	3 (1, 4)
Upper Coastal Plain (GA/FL)	23 (18, 28)	19 (12, 24)	19 (12, 24)
Atl. Coastal Plain (Carolinas)	2 (1,4)	1 (0, 3)	1 (0, 3)
Atl. Coastal Plain (GA/FL)	21 (15, 26)	15 (8, 21)	15 (8, 21)
FL Peninsula	28 (24, 33)	23 (17, 28)	23 (17, 28)
AL/FL Panhandle	11 (7, 16)	9 (5, 13)	9 (5, 13)
Upper Alabama	3 (1, 4)	2 (0, 4)	2 (0, 4)
Range-wide	92 (81, 102)	71 (59, 82)	71 (59, 82)
Year - 2060			
Upper Coastal Plain (South Carolina)	3 (3, 4)	2 (0,4)	2 (0,4)
Upper Coastal Plain (GA/FL)	23 (18, 28)	15 (8, 23)	15 (8, 22)
Atl. Coastal Plain (Carolinas)	2 (1,4)	1 (0, 2)	1 (0, 2)
Atl. Coastal Plain (GA/FL)	21 (15, 26)	11 (4, 20)	11 (4, 20)
FL Peninsula	28 (24, 33)	19 (13, 25)	19 (13, 25)
AL/FL Panhandle	11 (7, 16)	7 (3, 12)	7 (3, 12)
Upper Alabama	3 (1, 4)	1 (0, 4)	1 (0, 4)
Range-wide	92 (81, 102)	57 (44, 71)	57 (43, 70)
Year - 2080			
Upper Coastal Plain (South Carolina)	3 (3, 4)	2 (0, 4)	2 (0, 4)
Upper Coastal Plain (GA/FL)	23 (18, 28)	13 (5, 21)	12 (5, 21)
Atl. Coastal Plain (Carolinas)	2 (1,4)	0 (0, 2)	0 (0, 2)
Atl. Coastal Plain (GA/FL)	21 (15, 26)	9 (3, 19)	9 (3, 18)
FL Peninsula	28 (24, 33)	16 (10, 23)	16 (10, 22)
AL/FL Panhandle	11 (7, 16)	6 (2, 12)	6 (2, 11)
Upper Alabama	3 (1, 4)	1 (0, 4)	1 (0, 4)
Range Wide	92 (81, 102)	48 (34, 63)	47 (33, 62)

Cell shadings: Red – lower 95% credible interval includes 0 populations persisting in a unit; Orange – lower 95% credible interval includes 1 or 2 populations persisting in a unit.

5.2.4 Limitations

In any SSA, the process of modeling current conditions and projecting those into the future requires making strategic simplifications of reality using the best available data. We must account for multiple uncertainties and make informed assumptions when necessary. The level of uncertainty is especially high for species that are difficult to detect, and data limited such as the Florida pinesnake. Still, this assessment addressed some of the key uncertainties and yielded useful predictions for characterizing the subspecies' status. Quantitative models are essential tools for capturing the dynamics of complex, ecological systems, predicting species' outcomes, and informing conservation decisions for at-risk and listed species (Morris et al. 2002, entire; McGowan et al. 2017, entire). Through the use of predictive models and multiple scenarios, we captured a range of possible conditions in the future. We highlight and explain some of the key limitations and assumptions of the analyses below.

We used/developed three types of models in our analyses, (1) a habitat suitability model (Crawford et al. 2020a, entire), (2) a current persistence model, and (3) a future persistence model. Each one, like all models, required simplifications and assumptions about the underlying ecological systems. The models generated specific quantitative results that should be recognized as our most reasonable estimates. For example, the results of the habitat suitability model represented areas of relatively high and low suitability based on a set of predictors; however, a specific site may have additional factors, such as the abundance of a key resource, presence of an invasive species, history of over-collection, or area of high road mortality, which we were not able to incorporate into the model.

We were not able to include all the factors that may be influencing viability for the Florida pinesnake (Chapter 3) into the model because spatial data for these stressors were not available across the subspecies' range or at all (e.g., impacts from disease and red imported fire ants). These limitations likely did not hinder the reliability of results from the habitat suitability model since several of the most influential factors identified in the literature and by expert judgment as predictors – specifically, soil drainage, compatible land cover, and fire frequency – were included. Simplifications were also made in the persistence model where we only included three predictors even though an infinite number of factors could influence population persistence locally. However, we developed all models in this assessment using the best available data for the Florida pinesnake and its ecological systems, and despite simplifications, the results of this assessment tended to agree with expert judgment and previous studies characterizing Florida pinesnake habitat use and population persistence. We also projected population outcomes under multiple scenarios using the same model structure, and although there was uncertainty around model predictions, one can consider and compare the range of plausible future conditions to effectively evaluate relative risks to the subspecies.

We delineated 153 populations with 133 of them that served as the primary units for analyses but creating these population boundaries required a set of assumptions that are important to consider

when interpreting the results. Most importantly, we assumed that our full database of subspecies records, which came from datasets maintained by State and Federal agencies and other researchers, adequately represented the distribution of Florida pinesnake populations in the Southeast. While the full dataset likely captured most of the areas where Florida pinesnakes occurred, it likely did not capture all locations where they occur due to the difficulty of detecting the subspecies and the lower search effort and data availability from private lands. This results in some uncertainty around the actual number of populations. We employed a transparent process to delineate populations and develop models, and while assumptions are unavoidable, they were based on best available scientific information.

Models used in current and future conditions hinged on a novel approach to estimate persistence of populations that has only been used in analysis of the southern hognose snake before (USFWS 2019, entire). We made several assumptions about how persistence probability was related to population resiliency. First, we assumed if the population is currently on the landscape, then it must display some level of resiliency for it to have persisted over time. Secondly, most likely larger populations will have a higher current persistence probability than smaller populations. Third, we assumed the more resilient populations are more likely to occur in highly suitable habitat, on protected lands, and in proximity to other populations. Since we were able to account for these factors in our model, populations that meet these criteria most likely have higher current persistence probabilities and greater resiliency.

While persistence probability is a reasonable indicator of resilience, we did not explicitly account for other components of population dynamics within the population (e.g., increases in recruitment, abundance) or between populations (e.g., colonization rates between neighboring populations) that could influence viability. More complex models (e.g., population viability analysis) exist that directly estimate these biological processes; however, they require basic life history and demographic information that have not been estimated for Florida pinesnakes. It is possible that our models are under- or overestimating persistence for certain populations by not explicitly estimating other processes like recruitment or colonization. However, biological processes like recruitment and colonization are represented, at least to some degree, in the set of predictors of annual persistence. We assume that the predictor of HSI, which was positively related to the annual persistence probability, is also positively related to recruitment and survival rates for a population as the habitat it occupies contains more compatible land cover and fewer anthropogenic threats. Similarly, we did not explicitly estimate colonization between populations, but this process was represented by including the number of nearby populations as a predictor of a population's annual persistence probability. We also acknowledge that many – or most – population boundaries include some proportion of unsuitable habitat that, in some cases, could be causing a single population defined in our study to be functioning as several isolated subpopulations. It is likely that smaller, isolated subpopulations would each have lower growth rates and persistence probabilities than a single large population. Directly accounting for fragmentation would require fine-scale information about Florida pinesnake movement

capabilities through different habitat types that is not currently available. However, it was found in preliminary analysis for the habitat suitability models that mean HSI was related to the amount and connectivity of suitable habitat in populations; thus, we assume that including HSI as a predictor of persistence also partially accounted for the influence of fragmentation within a boundary. Even without explicitly estimating recruitment or colonization rates, the model structure allows for populations to be stable (i.e., persist and not become extirpated) over time, which is useful for assessing the degree and distribution of risk of extinction for populations across the subspecies' range. Still, improving the quality and quantity of data for the Florida pinesnake in the future will greatly improve predictions of population outcomes using additional types of models.

For the future persistence model, we used the SLEUTH which simulates patterns of urban expansion across the southeastern United States. Recent models developed for the state of Florida also try to predict the impact SLR and urbanization will have on natural resources (Romañach et al. 2020, entire). For use in this SSA, this recent model has limitations because their study area does not overlap with the full range of the Florida pinesnake making it not feasible to use in our model. However, to help better understand our model and ensure that we are not under-predicting the amount of habitat loss, we pulled data from the Romañach et al. (2020) model and ran it along with the SLEUTH for Florida under a high urbanization scenario. There was a clear correlation between the two data sets, but the SLEUTH projections were generally predicted to have more habitat loss. It means that our model predicted more of a "worst case" scenario than the Romañach et al. (2020) model.

Finally, as previously stated, the habitat suitability model was based on a set of predictors, but there are likely other important climate-related factors that were not included in our model that affect habitat suitability and future population persistence. Although we did include the projected impacts of SLR and unsuitable weather conditions for prescribed fire, there are other probable climate change-related impacts such as, projected increases in mean temperature. An increase in mean temperature will result in increases in soil temperature, which has the potential to negatively impact burrowing species such as the Florida pinesnake. Additionally, there is a great amount of uncertainty in how the longleaf pine ecosystem will respond to climate change (e.g., range contraction vs. shifting range), and any changes in the total acreage or distribution of longleaf pine will likely impact the Florida pinesnake.

5.3 Summary of Future Conditions and Viability based on Resiliency, Representation, and Redundancy

For the Florida pinesnake to maintain ***viability***, it needs to have resilient populations that are able to withstand stochastic events and maintain ecological and genetic diversity, which will help preserve the breadth of adaptive capacity, and hence, the evolutionary flexibility of the subspecies in response to environmental changes. In addition, the populations need to be spread

across the subspecies' range in a way that reduces the chance that a catastrophic event is likely to lead to extinction.

This assessment shows that there have been range-wide declines for this subspecies from its historical number of populations of 133 to current conditions with 92 extant population. Our analysis indicates that future resiliency, representation, and redundancy, as measured by future population persistence, for the Florida pinesnake is predicted to further decline from current conditions under all scenarios considered.

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APPENDIX A: EXTENDED METHODS AND RESULTS FOR CURRENT AND FUTURE POPULATION PERSISTENCE

Population persistence analysis for

Florida pinesnake (*Pituophis melanoleucus mugitus*)

Species Status Assessment (SSA)

Photo credit: P. Hill

1. Methods

We sequentially went through the following steps when conducting this analysis and describe each step in more detail in the sections below. First, we gathered a comprehensive dataset of Florida pinesnake locality records and delineated population boundaries based on clusters of these records (Section 1.1). Second, we gathered locality records for other snake species commonly observed in Florida pinesnake habitats that were found within population boundaries to represent search effort (Section 1.2). Third, we extracted metrics of site-level conditions at the present for each population to use as predictors of persistence (Section 1.3). These predictors included mean habitat suitability index that was estimated as part of the habitat suitability model (Crawford et al. 2020a, entire). Fourth, we constructed a persistence model to estimate current population persistence (in 2021) that leveraged information from the Florida pinesnake and other snake species locality datasets and accounted for imperfect detection (Section 1.4). Fifth, we created six scenarios of stressors to evaluate a range of plausible future conditions for the subspecies; we then performed spatial analyses to capture predicted changes in land cover and fire frequency under each scenario, and we extracted these conditions to populations (Section 1.5). Sixth, we built a simulation model that used habitat conditions and current persistence to forecast population persistence through 2080 under each scenario (Section 1.6). Lastly, we summarized results by representative unit and range-wide currently (in 2021), in 2040, 2060, and 2080 (see Section 2. Results).

The core of this analysis – the persistence model – used a novel structure to estimate current population persistence, given data limitations of the Florida pinesnake (Crawford et al. 2020b, entire). Specifically, we adapted the Cormack- Jolly-Seber model (Cormack 1964, entire; Jolly 1965, entire; Seber 1965, entire), designed to estimate survival of individual animals based on mark-recapture data, to analogously estimate persistence ("survival") of populations based on detection-non-detection data within each population boundary. Aggregating and analyzing pinesnake records by populations can overcome challenges associated with low probability of recapture and allows for estimating processes occurring at the population-level. We performed spatial analyses in ArcGIS version 10.7.1 (ESRI, Redlands, CA) and statistical analyses in R version 4.1.0 (R Core Team 2016, unpaginated).

1.1 Collecting Florida pinesnake records & delineating populations

To develop population persistence models, we first compiled a geospatial database of occurrence records for Florida pinesnakes and created population boundaries around clustered records. We compiled species records from datasets maintained by Natural Heritage Programs, USFWS, USFS, DoD, state agencies, academic researchers, and HerpMapper – an online platform where species records are reported by the public and validated by professional herpetologists (HerpMapper 2021, unpaginated). Records included opportunistic sightings, as well as observations from systematic research and monitoring studies. From the full location dataset, we removed two populations within the analysis area that contained a single record not associated with a date and removed records that fell outside of the analysis area. The analysis area boundary was created during a multi-species habitat modeling study that began prior to developing the SSA (Crawford et al. 2020a, entire), and thus contains the extent of the existing habitat suitability model for the Florida pinesnake. We performed analyses on the remaining 133 populations.

We performed the following steps to delineate populations around all subspecies locations:

Buffered subspecies locations by 6 km and joined buffers that overlapped. The choice of buffer distance was similar to protocols used by NatureServe (NatureServe 2021, unpaginated) but further informed by reviewing telemetry studies for the subspecies and expert input. NatureServe suggests a separation distance for colubrid snakes of 10 km (6.2 miles) for suitable habitat and 1 km (0.62 miles) for unsuitable habitat. This recommendation was based on a limited number of studies of movement and range in colubrid snakes (NatureServe 2021, unpaginated). Two telemetry studies of Florida pinesnakes

indicated that a 10 km buffer may be too large for purposes of delineating populations. Those studies estimated that Florida pinesnake home ranges were smaller, between 57-59.2 ha (141-146 ac), relative to other similar-sized colubrid snakes (Franz 2005, p. 121; Miller et al. 2012, p. 709). The maximum annual movement or dispersal distance reported for a Florida pinesnake is 4.26 km (2.65 mi; Franz 2005, p. 124). Though it is reasonable to believe that dispersing pinesnakes could and would make larger movements, experts we consulted agreed that 6 km was a reasonable distance for a buffer around any pinesnake record. This distance represents the potential maximum radial extent an individual would likely occupy and is likely a generous estimate of the area actually used by an individual.

Split the polygons from step 1 with Savannah and Chattahoochee Rivers.

Removed any polygons from step 2 that did not have any subspecies records within them and clipped final population boundaries by the U.S. Coastline.

After we delineated populations, we associated the population ID number for each Florida pinesnake record in ArcGIS. Using the Florida pinesnake record dataset, we created observation histories (denoted as $y_{i,t}$) describing if at least one individual was observed in a population (i) in a given year (t) ($y_{i,t} = 1$, 0 otherwise). These observation histories were used to model persistence (see Section 1.4).

Collecting non-target snake species records

Models similar to our approach rely on absence information – i.e., when searches occurred but the target species was not observed – to estimate survival, abundance, and persistence (Kéry et al. 2009, entire; Kéry and Schaub 2012, p. 171-239; MacKenzie et al. 2002, entire). However, robust search effort and absence data do not exist for the Florida pinesnake, therefore, we developed a search effort index from occurrence records of Florida pinesnakes and other snake species commonly observed in Florida pinesnake habitats (hereafter search effort dataset) obtained from HerpMapper.org and other partners. We used the search effort dataset to account for imperfect detection of the Florida pinesnake and improve the precision of persistence estimates. Records of other (non-target) snake species that fall within a given population boundary indicate that the area was searched by an individual in a given year and inform the likelihood a population is still persisting. For example, imagine two populations: A and B. Florida pinesnakes have not been observed in both Populations A and B since 1990. Population A has been frequently searched since then, and other species of common snakes continue to be reported in the area. Population B has only been searched once since then. Assuming everything else is equal, there is more evidence that Population A has become extirpated of Florida pinesnakes while Population B is more uncertain. In using non-target species data, we made the following assumptions: 1) non-target records indicate an event when an area known to have Florida pinesnakes at some time was searched, 2) the search was performed in a way that Florida pinesnakes could have been observed (e.g., road surveys), and 3) when a person submitted non-target records but not Florida pinesnake records to HerpMapper after a search, this indicated the area was searched but no Florida pinesnake was found (i.e., we assumed Florida pinesnakes would be reported if found).

We developed a list of 12 non-target species most commonly observed while surveying for Florida pinesnakes: *Agkistrodon contortrix subspp.*, *Cemophora coccinea subspp.*, *Coluber constrictor subspp.*, *Crotalus adamanteus*, *Crotalus horridus*, *Heterodon simus*, *Lampropeltis getula subspp.*, *Masticophis flagellum subspp.*, *Opheodrys aestivus subspp.*, *Pantherophis guttatus*, *Pantherophis obsoletus subspp*, and *Thamnophis sirtalis subspp.* (multiple experts, pers. comm.). We queried HerpMapper and accessed 1,783 records of these snake species that fell within the Florida pinesnake population boundaries. These records included data on observer name and date of observation. We added 778 Southern hognose snake records to the non-target dataset that were collected from databases maintained by Natural Heritage Programs, States, and other Federal and research partners for a companion status

assessment on that species. Finally, we added our full dataset of Florida pinesnake records since these indicate search events as well. While HerpMapper launched in 2013, there were records included in this and the other snake datasets used to create a search effort index that were collected between 1880 and 2021.

We associated the population ID number for each record in the search effort dataset in ArcGIS. We noticed occasions in the HerpMapper data where the same observer would submit multiple records from the same date. In order to better capture the number of search events and create an index of search effort, we filtered the search effort dataset to remove duplicate records coming from the same observer on the same date. Thus, the dataset now represented unique searches by individuals, which we refer to as observer-days, and included 2,126 observer-days in total. We then followed the same process used to create observation histories of Florida pinesnakes to create search effort histories for each population. Search effort histories ($s_{i,t}$) described the number of observer-days for population (i) in year (t). Lastly, we scaled all search effort values, so they were centered on 0 for use in the model.

Extracting current site-level conditions for populations

Using the population boundaries, we summarized population-specific metrics for several spatial variables that likely influence persistence and used these as predictors in the persistence model. We included three metrics representing site-level conditions in the analysis: 1) average Habitat Suitability Index (HSI), estimated in the habitat analysis (Crawford et al. 2020a, entire), 2) proportion of the population area protected, and 3) number of additional populations within 10 km.

We used the Zonal Statistics tool in ArcGIS to calculate mean HSI within each population boundary. HSI can be thought of as a measure of habitat quality between 0 (most unsuitable, lowest quality) to 100% (most suitable, highest quality). Predicted HSI for the Florida pinesnake was positively related to the amount of well-drained, sandy soils, compatible landcover (e.g., mixed/evergreen forest, scrub/shrub, grasslands, flatwoods), fire frequency, mean annual summer precipitation, and Topographic Position Index (Crawford et al. 2020a, entire).

We used the Zonal Statistics tool to calculate proportion of a population boundary currently in protected lands. We created a protected lands raster where areas included in the USGS Protected Areas Database (http://www.protectedlands.net/data/), FNAI Conservation Lands Database (https://www.fnai.org/publications/gis-data) , GADNR Conservation Lands 2019 Database (https://data.georgiaspatial.org), NCNHP Natural Heritage Areas and Managed Areas Databases (https://ncnhde.natureserve.org/content/data-download), TNC Conserved Lands of South Carolina Database (https://tnc.maps.arcgis.com/apps/dashboards/e2214b443c65483091cb744b13fed253), and ALDCNR Wildlife Management Areas, State Lands, State Parks, and Forever Wild Tracts Databases (https://conservationgis.alabama.gov/adcnrweb/rest/services) were given a value of 1. These areas include publicly owned and managed lands as well as private lands registered in state or federal programs where natural resource conservation is one of the management goals. All other area was given a value of 0. Therefore, calculating the mean of the protected lands raster within each population boundary yielded the proportion of protected land.

Lastly, we used the buffer and spatial join tools in ArcGIS to calculate the number of populations found nearby (within 10 km). Although experts believed movement between two populations was unlikely if they were greater than 6 km apart, we recorded populations within 10 km because they may provide opportunities for “rescue” where recolonization can occur after a catastrophe; alternatively, nearby populations could provide a signal that there are localized conditions (e.g., geological, climatic) that promote population persistence that have not been otherwise captured in our analyses.

1.4 Current persistence model

We used a persistence model (Crawford et al. 2020b, entire) to estimate trends in Florida pinesnake populations and derive probabilities that each population in the subspecies' range persists currently in 2021. We used this model, an adapted Cormack-Jolly-Seber (CJS) model (Cormack 1964, entire; Jolly 1965, entire; Seber 1965, entire) designed to estimate survival of individual animals based on mark-recapture data, to analogously estimate persistence ("survival") of populations based on their observation histories. Under this framework, persistence can also be thought of as (1 – the probability of extinction) for each population.

The persistence model was developed using a state-space formulation fitted in a Bayesian framework (Kéry and Schaub 2012, p. 171-239). State-space models explicitly model how the state of a system changes over time (e.g., population persistence and extirpation) as well as the process of observing an individual, given the population has not become extirpated (Figure A-1). Thus, they are helpful in separating real biological signals from error in observation data.

We modeled the state history ($z_{i,t}$) of a population over the period from 1881 to 2021. We modeled state histories of populations as a function of an annual population-specific persistence probability $\varphi_{i,t}$, where a population could either persist ($z_{i,t} = 1$) or become extirpated ($z_{i,t} = 0$), given that it still persisted the year before ($z_{i,t-1} = 1$). Using this approach with our Bayesian formulation, we provided the model with information about the known state of populations, which helped it estimate parameters. For example, a population where Florida pinesnakes were first detected in 1980 and last detected in 2000 indicates that it persisted at least between 1980 and 2000, and we would supply known states (z = 1) for that population between these years (but the model would predict z states each year after 2000). In all years, the prediction of a z state in any year is a random outcome of the model (0 or 1) when no snake was observed in the population that year and is a deterministic outcome (1) for years when snakes were observed. In order to characterize current population resilience, we derived the probability that each population still persisted in 2021 by calculating the proportion of model iterations where a $z_{i,2021} = 1$. We interpret this value as the probability a population has persisted through the entire time period modeled and still exists (which we refer to as current persistence), and we note that this value differs from $\varphi_{i,t}$ estimated in the model that reflects the probability a population will persist into the next year (which we refer to as annual persistence). We modeled observation histories ($y_{i,t}$) for a population as a function of the probability of detecting at least one individual, $p_{i,t}$, given that the population had not become extirpated.

We included additional effects on persistence and detection parameters. For persistence, we included a fixed group effect of representative unit (μ_{rep}). This allowed for a different baseline persistence probability for each unit. We also included three fixed effects for each of the predictors indicating conditions measured at each population: mean Habitat Suitability Index (β_{HSI}), proportion of a population protected ($\beta_{protect}$), and number of nearby populations ($\beta_{connect}$). This allowed for estimating the relationships between each predictor and persistence probabilities across the species' range. For detection, we included an intercept (μ_{mean}). We also included a fixed effect of search effort (β_{effort}), using the search effort index data, and a fixed effect for the total area within each population boundary that was within 2 km of a major road (β_{road}). This was used to account for the effect of roads on re-detection probability, given that many observations of Florida pinesnakes occurred on roads.

We used standard practices for fitting Bayesian models following Kéry and Schaub (2012, p. 171-239). We fit the persistence model with Markov chain Monte Carlo (MCMC) methods in Jags called from R via the R2jags package (Su and Yajima 2012, unpaginated). We assigned diffuse prior distributions for all parameters. These included uniform priors within the interval [0,1] specified on the probability scale for mean re-detection probability and mean persistence probability and normally distributed priors

(mean = 0, precision = 0.37) specified on the logit scale for all beta coefficients. We generated three MCMC chains using 350,000 iterations where we retained every third iteration from the last 300,000 iterations, yielding a final set of 300,000 samples from posterior distributions of the parameters. We assessed convergence based on the Brooks-Gelman-Rubin statistic (< 1.1 for all parameters) and by visually inspecting chain mixing in MCMC trace plots and posterior distribution plots for evidence of unimodality. To assess goodness-of-fit, we conducted posterior predictive checks (Gelman et al. 2000, entire) where we simulated datasets using parameters estimated in the model, calculated the mean number of populations with simulated detections of Florida pinesnakes in four time periods (1920-1929, 1950-1959, 1980-1989, and 2012-2021), and compared mean observations in these periods from the real dataset with values from simulated datasets. We based parameter inferences on posterior means and 95% Bayesian credible intervals (BCIs; 2.5th – 97.5th percentile of the distribution).

We measured Florida pinesnake population resiliency by using current persistence probabilities (between 0 and 100%), and we summarized results by grouping populations into categories representing ranges of persistence probabilities. The categories we chose were unlikely or extirpated (< 50%), more likely than not on landscape (50 – 79%), very likely on landscape (80 – 89%), highly likely on landscape (90 – 95%), and extremely likely on landscape or extant (95 – 100%). For a population to be included in a category, the lower limit of its 95% BCI had to exceed the lower bound of that category. For a population to be highly resilient, it must have a relatively high current persistence probability.

To characterize representation and redundancy, we captured predictions of the number of resilient populations within a representative unit and range-wide in two ways. First, we summed the number of populations within a unit and range-wide with current persistence probabilities at or above each category (threshold) as described for characterizing population resiliency. Second, we recorded the number of populations predicted to persist in 2021 within each unit and range-wide using direct outputs from the model. Each model iteration recorded the number of populations persisting in 2021 in each representative unit and range-wide, and we used all model iterations to calculate the mean (the most likely prediction) and 95% credible intervals for the predicted number of persisting populations in 2021.

One can think of the difference between a specific population's probability of persistence and the mean number of populations persisting within a representative unit by considering a set of four fair coins. Each has a 50% probability of getting a heads – this is a population's persistence probability. If we flip all four coins many times, the most likely outcome, on average, is getting two heads and two tails – this is the mean number of persisting populations predicted in a model iteration. The specific coins that yield a heads may change each trial, but we still expect two out of four heads most commonly. Therefore, when assessing representation and redundancy at the scale of a representative unit and range-wide, it may be helpful to consider the two types of results alongside each other. One can interpret the mean number of persisting populations as the most likely outcome then further assess the current resiliency of populations within a representative unit and range-wide using the number of populations above a certain persistence threshold (e.g., 80%). We also note that the laws of probability make it so this mean number of persisting populations approximately equals the number of populations with a 50% or greater persistence probability.

We measured representation using the number and distribution of resilient populations (i.e., those above a certain persistence probability threshold) across representative units in the subspecies' range as well as assessing the spatial distribution (latitudinal and longitudinal variability) of resilient populations. To have high representation the species must have multiple highly resilient populations located in each of the representative units, and those occupied units should span the latitudinal and longitudinal extent of historical populations. We measured redundancy using the current number and distribution of resilient populations within representative units and across the range of the subspecies. To have high redundancy, the Florida pinesnake would need to have multiple resilient populations within a

representative unit and throughout its range.

1.5 Future scenarios and spatial analysis

We considered processes that may influence future habitat suitability and population persistence of the Florida pinesnake and grouped these into six plausible scenarios to simulate future conditions. We used spatial analyses performed in ArcGIS to capture changes in land cover and fire frequency due to future urbanization and the effects of climate change on sea level rise and on opportunities for prescribed fire. Changes in these spatial predictors subsequently influenced future habitat suitability, as predicted by the habitat suitability model, and population persistence, as predicted by the stochastic simulation model. We describe spatial analyses of future stressors below.

We captured the effects of future urbanization and sea level rise on compatible land cover for Florida pinesnake populations. First, we mapped areas predicted to be urbanized in the future using the Slope, Land cover, Exclusion, Urbanization, Transportation, and Hillshade (SLEUTH) model. SLEUTH is a cellular automata model that applies transition rules to the states of a gridded series of cells, and in this case the transition is that from undeveloped to developed land cover, otherwise known as urbanization (Chaudhuri and Clarke 2013, entire) and has been successfully applied worldwide over the last 18 years to simulate land use change. SLEUTH simulates patterns of urban expansion that are consistent with spatial observations of past urban growth and transportation networks, including the sprawling, fragmented, "leapfrog" development that has been the dominant form of development in the southeastern U.S. (Terando et al. 2014, entire). The SLEUTH model predicts the probability of urbanization ranging from 0-100% within 60-m square cells, which we resampled to 30-m cells to coincide with other spatial data layers, and urbanization is modeled for each decade from 2010 (baseline) to 2100. The SLEUTH model algorithm specifies a 0% probability of urbanization for protected areas (e.g., National Forests, State-owned conservation lands); however, new conservation lands have been created by government agencies and other partners since the model was developed. Therefore, we set the probability of urbanization to 0 for any cell that overlapped our raster of currently protected areas. We used SLEUTH projections from 2040, 2060, and 2080 and selected all cells with a 90 (most conservative), 50, and 10% (most liberal) probability of urbanization to represent low, medium, and high levels of urbanization, respectively. In addition to urbanization, we captured potential loss of habitat due to inundation from sea level rise. We obtained projections of sea level rise associated with Representative Concentration Pathways (RCPs) from the Sixth IPCC Assessment Report (IPCC, 2021, p. 21-23) and used SLR inundation maps from the National Oceanic and Atmospheric Administration (https://coast.noaa.gov/slrdata/; accessed 26 Oct 2021). Based on the IPCC report, for RCP 4.5 we assumed sea level rise of 1ft at years 2040 and 2060 and 2ft at year 2080 and for RCP 8.5 we assumed sea level rise of 1ft at years 2040 and 2060 and 3 ft at year 2080. For each projected height of sea level rise, we created a raster of inundated areas. Finally, we captured the reduction in compatible land cover from urbanization and sea level rise for each time period and stressor level by removing areas predicted to be urbanized or inundated from the current compatible land cover layer using the Raster Calculator tool and calculated the proportion of compatible land cover in each population boundary using the Zonal Statistics tool.

We also captured the effects of urbanization on fire frequency. This was done to account for fire exclusion/suppression that often occurs in habitat adjacent to urban areas, known as the wildland-urban interface, due to safety and smoke management restrictions (Theobald and Romme 2007, entire). Studies have found evidence of fire exclusion/suppression in habitats within 600 m to 5 km of urban areas (Pickens et al. 2017, entire; Theobald and Romme 2007, entire). Therefore, we chose a moderate value of 3.2 km (2 mi) to capture the interaction between urbanization and fire frequency. Using the predicted urbanized areas at each time period and urbanization probability level, we identified areas within 3.2 km of urban areas using the Euclidian Distance tool. We then projected future reductions in

fire frequency starting with the current fire frequency raster and applying a distance-weighted reduction to any cell within 3.2 km of urban areas. Using this approach, fire frequency for any cell overlapping future urban areas was reduced by 100% (equaled 0) and any cell more than 3.2 km away from urban areas was reduced by 0% (unchanged). Any cell between 0 and 3.2 km away from urban areas was reduced by a percentage proportional to its distance – i.e., a cell 1.6 km from urban areas was reduced by 50%. We then recalculated the mean fire frequency for each population.

To account for changes in fire frequency from climate change, we used U.S. Geological Survey historical and future prescribed burn windows for the Southeast United States (Kupfer et al. 2021, unpaginated). This dataset included observed mean proportion of days suitable for burning by month for the periods of 2000-2009 and 2010-2017 and decadal mean future proportions of days suitable for burning by season. To correspond with the fire frequency predictor used in our model of habitat suitability, which used annual fire detections from 2001-2016, for each cell we calculated an annual weighted average of the historical 2000-2009 and 2010-2017 data and weighted monthly data according to number of days per month. For future burn windows, we used data for RCPs 4.5 and 8.5 and each decade leading up to 2040, 2060, and 2080. For each cell we calculated the decadal averages across seasons weighted by number of days per season. We calculated population means for each burn window data set and used the population averages to calculate proportional changes in fire windows for 2040, 2060, and 2080 for each RCP. We assumed that these would result in the same proportional changes in fire frequency and reduced each population's mean fire frequency accordingly, which subsequently affected overall habitat suitability. Two populations did not have data for observed fire window, so we used values from the nearest populations.

Our six future scenarios included all possible combinations of urbanization (low, medium, high) and climate change (RCP 4.5, RCP 8.5). For each scenario, we used the resulting land cover and fire frequency layers as inputs for predicting future habitat suitability and population persistence. We also compared a seventh scenario that included no changes in urbanization or climate and SLR from the current conditions. This created a null expectation for projected population losses against which we could compare the models with projected changes in urbanization and climate.

1.6 Future simulation persistence model

We used a stochastic simulation model that used a Markovian process to predict the probability of persistence in the future for each population in the analysis area based on the current probability of persistence and future predicted changes in habitat suitability and land protection. The primary output metric for this model was the probability a population persists at a given year in the future (2040, 2060, and 2080), which we call "future persistence probability." We calculated the future persistence probability for each population as the proportion of model iterations where the population was persisting at a given year. The complement of future persistence probability can be interpreted as the probability a population has become extirpated by a given year.

We built a multi-loop simulation model (following McGowan et al. 2014, entire) that allowed us to simulate thousands of replicates of each population under different scenarios. This approach accounted for random year-to- year stochasticity as well as uncertainty around rates (i.e., annual persistence probability) estimated from the current persistence model. The model looped through 10,000 iterations for each of the six scenarios. It then looped through each of the 133 Florida pinesnake populations within the analysis area and simulated persistence from the present (2021) to 2080. We briefly discuss the model framework, inputs, and outputs below.

In short, the model calculated inputs from the spatial analysis (compatible land cover and fire frequency), predicted habitat suitability, selected a set of parameter estimates from the current

persistence model, used inputs and the parameter estimates to predict annual probabilities of persistence in the future, and used the annual persistence probabilities to predict the state of each population (persisting or extirpated) each year through 2080. It accomplished this through a series of steps.

Step 1) Each iteration of the model began by calculating annual values of land cover and fire frequency by interpolating the values of these predictors calculated in the spatial analysis at 2040, 2060, and 2080. These input values were scenario dependent.

Step 2) The model then used inputs of land cover and fire frequency, along with all other predictors used in the habitat suitability model (e.g., soil drainage, local elevation) that did not change in future scenarios, to calculate mean habitat suitability for each population and year, given conditions of each scenario. This was done using the habitat suitability model and the predict.glm function in R.

Step 3) Each iteration of the simulation selected a single set of parameter estimates ("posterior estimates") generated from one iteration of the current persistence model. These parameters included mean current persistence probability for each population and the estimated effects of all predictors of the annual persistence probability (mean representative unit persistence, habitat suitability, proportion of protected land, and number of populations within 10 km). This was done in order to account for the uncertainty around parameters (parametric uncertainty) estimated in the current analysis, as well as year-to-year stochasticity in the simulation model. Therefore, the simulation model results varied between iterations based on which parameter estimates were randomly selected.

Step 4) Once we had a sample of posterior estimates and scenario-, population-, and year-specific inputs, the model calculated annual probabilities of persistence using the same model structure as the persistence analysis. Annual persistence probability was a function of mean representative unit persistence, habitat suitability, proportion of protected area, and number of populations within 10 km. The model simulated the state of each population each year through 2080. The probability of persistence in 2022 was equal to the mean current persistence probability for each population multiplied by the annual persistence probability for that population that was randomly selected from the estimates generated in the current persistence model ($\varphi_{current}*\varphi_t$). The state of the population (persisting or extirpated) in the next year was drawn from a Bernoulli trial using that probability. For years 2023 through 2080, the state of a population was drawn from a Bernoulli trial using the annual persistence probability, given that the population had persisted in the previous year. Thus, as in our current persistence model, extirpation was an absorbing state (i.e., recolonization was not allowed).

Step 5) The model recorded summary outputs useful for characterizing resiliency, redundancy, and representation. For this model, the primary output was the probability a population persists at a given year in the future through 2080, which we call "future persistence probability." We calculated the future persistence probability for each population as the proportion of model iterations where the population was persisting at a given year. The complement of future persistence probability can be interpreted as the probability a population has become extirpated by a given year. These future persistence probabilities for populations are directly comparable to the current persistence probabilities. We characterized future conditions similarly to current conditions by summarizing the number of populations at or above certain persistence probability thresholds (50, 80, and 95%) in each representative unit and range-wide, given each of the six scenarios. Like in the current conditions analysis, in addition to a specific population's persistence probability, each model iteration recorded the number of populations persisting at each time step in each representative unit and range-wide. We used all model iterations to calculate the mean (the most likely prediction) and 95% credible intervals for the number of persisting populations in each representative unit and range-wide each year. Similar to results for current conditions, the mean number of persisting populations approximately equals the number of populations with a 50% or greater persistence probability. We summarized these outputs for

three future time horizons: 2040, 2060, and 2080.

2. Results & Discussion

2.1 Current and historical populations

For our population analysis, we obtained 1,989 Florida pinesnake records, from years 1881-2021. Many of the early occurrence records were for the county only without available coordinates. The occurrence records were spread throughout the subspecies' range, with the Florida Peninsula representative unit having the most records. From these records, we identified a total of 153 populations of Florida pinesnakes with a mean area of 42,976 ha (range = 6,713 – 1,324,018 ha). If a population spanned more than one representative unit, the population was placed in the unit where most of the population boundary occurred for that population. Of the total pool of records and populations, we used 1,922 occurrence records grouped into 133 populations within the analysis area to fit the persistence model and simulate future persistence.

2.2 Predictor effects on current persistence

The persistence model showed adequate convergence, as assessed with traditional practices (MCMC chains were mixing well, Brooks-Gelman-Rubin statistics < 1.1 for all parameters). Posterior predictive checks indicated that the model fit the data adequately in the first and last time periods assessed but had poorer fit in the two intermediate time periods assessed (Figure A-2). This lack of fit in the intermediate time periods likely indicates that the detection model failed to capture some non-linearity between effort and re-detection probability through time, and that detection is likely underpredicted. Consequently, persistence estimates may be biased high. Efforts to correct this issue via log transformation of the effort variable and incorporating a random effect in the detection model did not improve fit.

As estimated in the persistence model, persistence varied by population but was positively and significantly influenced by mean habitat suitability and number of populations within 10 km (Table A-1). Persistence was also positively associated with the proportion of the population boundary in protected areas; however, the beta coefficient for this effect slightly overlapped 0 (Table A-1). Mean annual persistence probabilities were similar across most representative units (Table A-1). The baseline detection rate – the probability of detecting at least one pinesnake in a population in a given year – was relatively high (0.63; Table A-1). The effects of effort and area of the population within 2 km of a major road on detection were also significantly positive (Table A-1).

The relationships between persistence and site predictors agreed with expert judgment about the habitat conditions that should favor Florida pinesnake persistence (Figure A-3). Although not significant, the direction of the effect for proportion on protected areas supported that populations have greater persistence when they are on protected lands (Figure A-3). This is likely because populations on protected lands have a reduced risk of habitat loss, especially if protected lands have been established and managed for conservation over longer periods, which would cause higher population persistence probabilities. The strong relationship between number of populations within 10 km and persistence (Figure A-3) could support that nearby populations may provide opportunities for "rescue" where recolonization can occur after a catastrophe. It may also be an artifact of large core areas of high-quality habitat that support multiple populations or subpopulations over large geographic areas.

2.3 Current conditions

2.3.1 Current resiliency

Current resiliency, as measured by current persistence probabilities, for the Florida pinesnake has decreased from historical conditions. We considered a population to be highly resilient if it had a relatively high current persistence probability, defined as the probability that there is at least one Florida pinesnake in the population in 2021. We estimated that 30.8% of the historic, delineated Florida pinesnake populations are likely extirpated and, we estimate that 21.8% of historic, delineated populations exhibit extremely high resiliency as of 2021. We estimate that the remaining 48.1% of populations span a wide range of estimated persistence probabilities related to habitat suitability, the number of other geographically proximate populations, the proportion of the population boundary on protected land, and the time since the last record for the subspecies within the area delineated for the population. We report mean persistence estimates and 95% credible intervals for each population in Table A-2.

Current representation and redundancy

Current representation, as measured by the number and distribution of resilient populations (i.e., those above a certain persistence probability threshold) across representative units in the subspecies' range has decreased from historical conditions. To have high representation the subspecies must have highly resilient populations located in each of the representative units, and those occupied units should span the latitudinal and longitudinal extent of historical populations. Our estimated persistence probabilities for delineated Florida pinesnake populations does not fully meet those criteria as two representative units do not have populations in the highest persistence category and those units are on the northern and western portions of the subspecies range.

Current redundancy, as measured by using the current number and distribution of resilient populations within representative units and across the range of the subspecies has also declined from historical conditions. To have high redundancy the species needs to have multiple resilient populations within representative units and throughout its range. Range-wide, the number of populations more likely than not to currently persist has decreased by 30.8%, relative to the historical number of populations, and we estimate that all seven representative units have likely lost at least one historic, delineated population. Nonetheless, all representative units have multiple resilient populations, which meets the criteria for redundancy.

2.4 Future conditions

Future habitat conditions

Under current conditions, the average population had compatible land cover of 43% (range = 2 – 83%), which decreased between 9% (best case scenario) and 12% (worst case scenario) on average across all scenarios. In terms of area, the average population had 233 km^2 (range = 2 – 6323 km^2) of compatible land cover under current conditions, which was reduced by 57 – 72 km^2 on average across future scenarios. While some populations experienced no loss of land cover, others were predicted to lose as much as 43.5% of compatible land cover due to urbanization and climate change. Burn windows, which averaged 293 days/year (range = 242 – 329 days/yr) under current conditions, were reduced by 41 – 71 days on average across all scenarios. These shorter burn windows, in combination with urbanization, resulted in fire frequencies (mean = 3.5%; range = 0-16.2% under current conditions) being reduced by 2.0 – 2.3% on average across scenarios. Using the reduced fire frequencies, the total number of burns between 2021 – 2080 for an average population is expected to be reduced by 1 with a range of approximately 0 to 3 fewer burns across all populations, compared to if burn frequency was not reduced from current levels (mean = 2; range = 0 – 10 burns from 2021-2080 under current levels). Finally, under current conditions, the average population had a mean HSI of 20.6%, which decreased between 4.6% (best case scenario) and 5.7% (worst case scenario) on average across all scenarios.

Future resiliency

We used the simulation model to predict the future persistence probability for each of the 133 populations in the analysis area through the year 2080. We followed similar steps to the current conditions analysis to summarize population resiliency across the species' range by reporting the number of populations in each category of persistence probability (50%, 80%, and 95%).

For all scenarios, we predicted that future population persistence decreased from current conditions and most populations that fell within the extremely likely on landscape (≥95%) threshold under current conditions were predicted to have lower persistence probabilities in the future. The number and percentage of populations likely to be extirpated (<50%) increased for all scenarios and future time horizons, relative to current conditions. For example, the percentage of total populations likely to be extirpated (<50%) in 2021 is 30.8% under the Low Urbanization/RCP 4.5 scenario. By 2040, that percentage increased to 42.9%, which represents 16 additional populations predicted to be lost. By 2060, the percentage increased to 58.6% (an additional 21 populations predicted to be lost), and by 2080 we predicted the loss of an additional 18 populations for a total of 96 populations (72.2%) predicted to become extirpated. These numbers are very similar for the High Urbanization/RCP 8.5 (worst case) scenario. For the High Urbanization/RCP 8.5 (worst case) the model predicted 5 populations in the extremely likely on landscape (≥95%) category of persistence in 2040 and only 1 in 2080.

The very small difference in resiliency between the scenarios was consistent with the minimal differences between the scenarios in the amount of land cover loss and change in habitat suitability for populations. The similarity in results between the stressor scenarios and the null model suggest that future population declines predicted to occur are primarily a consequence of population persistence probabilities being <1 for each population and no processes included in our model for colonization or recolonization, rather than a consequence of decreased habitat suitability due to climate change and urbanization.

Future representation and redundancy

Overall, we predicted future representation and redundancy to decrease from current conditions. The number of resilient populations and the number of representative units with resilient populations decreased under all the scenarios. In our current condition analysis, we determined that 5 of our 7 representative units have at least 3 populations extremely likely to be on the landscape (persistence probability ≥ 95%), and the other two units (Atlantic Coastal Plain (Carolinas) and Upper Alabama) had at least 1 population very likely to be on the landscape (persistence probability ≥ 80%). By 2080, we estimate that only 1 of the 7 representative units will still contain one population at the ≥ 95% threshold, for the "worst case" High Urbanization/RCP 8.5 scenario. Two representative units are predicted to have zero populations at or above the more likely than not to persist threshold (≥ 50%) by 2080. Thus, the Atlantic Coastal Plain (Carolinas) and the Upper Alabama Unit are predicted to be at the highest risk of losing populations, and the greatest potential for losing representation of the subspecies.

Into the future, the Florida pinesnake is predicted to experience a decrease in latitudinal and longitudinal variability (i.e., a range contraction) relative to its historic range. Future predictions show that populations at or above the 50% and 80% persistence threshold will likely be distributed throughout the core of the subspecies' range. However, these more resilient populations will continue to exhibit a degree of clustering, which will likely leave large areas lacking resilient populations (e.g., upper Alabama and coastal South Carolina).

Redundancy, as measured by the number of populations above the 50, 80, and 95% persistence thresholds and by the mean number of populations persisting, decreased in all scenarios, time periods, and representative units, relative to current conditions. All scenarios yielded nearly identical predictions of the number of persisting populations. This is not surprising since there were only minimal differences between the scenarios in the amount of land cover loss and HSI for populations.

Looking at redundancy within the representative units, we see that all representative units except Atlantic Coastal Plain (Carolinas) and Upper Alabama were predicted to have at least 1 population above the 80% persistence probability threshold and at least 2 populations above the 50% threshold by 2080 under the High Urbanization/RCP 8.5 scenario. The Florida Peninsula currently has the highest number of populations and was predicted to retain the highest level of redundancy in the future. Populations that occur on protected areas with the highest HSI in the subspecies' range were predicted to persist over time due to the favorable conditions.

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Table A-1. Parameter estimates (means and 95% Bayesian credible intervals) for the persistence model predicting the probability of persistence and detection between 1881 and 2021 for 133 populations of Florida pinesnakes. Posterior parameter estimates that do not overlap zero are interpreted as ecologically important.

Parameter	Mean	Lower 95%	Upper 95%
Persistence			
μ_{rep}			
Upper Coastal Plain (South Carolina)	0.845	0.494	0.996
Upper Coastal Plain (GA/FL)	0.932	0.825	0.987
Atlantic Coastal Plain (Carolinas)	0.708	0.378	0.945
Atlantic Coastal Plain (GA/FL)	0.936	0.836	0.991
FL Peninsula	0.941	0.883	0.977
AL/FL Panhandle	0.929	0.794	0.993
Upper AL	0.956	0.837	0.999
β_{HSI}	3.630	0.868	6.387
$\beta_{protect}$	1.770	-0.239	4.084
$\beta_{connect}$	4.189	1.912	6.549
Detection			
μ_{mean}	0.630	0.569	0.690
β_{effort}	0.134	0.068	0.210
β_{road}	0.176	0.093	0.269

Parameter notations: μ_{rep} – representative unit intercepts for persistence on the probability scale; β– covariate effects on persistence and detection on the logit scale; μ_{mean} –intercept for detection on the probability scale

Table A-2. Florida pinesnake population summary statistics of site-level conditions, survey effort, number of years with ≥ 1 Florida pinesnake observation, last year observed, mean and lower and upper 95% Bayesian credible intervals for current persistence – the probability of a population being extant in 2021.

Population ID	Area		Habitat Suitability Index	Proportion protected	No. of pops within 10 km	Representative unit [1]	Area within 2 km of roads (ha)	Total effort	No. years with FL pinesnake observation(s)	Last year observed	Current Persistence prob. (%)		
	Acres	Km^2									Mean	Upper 95%	Lower 95%
1	1146379.7	4641.2132	26.9	45.7	4	5	218,225	226	43	2021	100.0	100.0	100.0
2	159424.39	645.44288	17.0	3.5	3	4	37,389	25	10	2018	88.4	85.0	91.3
3	40269.107	163.03282	35.0	59.1	1	5	9,138	9	4	2013	96.6	94.2	97.9
4	16581.061	67.1298	16.9	3.0	0	6	6,713	4	1	1969	27.2	23.1	31.8
5	27934.423	113.09483	22.8	2.0	1	6	4,756	2	1	1949	19.3	15.4	24.1
6	80314.678	325.16064	23.7	42.4	3	6	13,592	7	3	2019	99.1	97.5	99.6
7	42261.142	171.09774	36.8	74.2	2	6	8,842	5	1	2008	92.4	89.5	94.7
8	43789.296	177.2846	19.6	20.3	2	6	9,347	6	2	2020	98.9	97.5	99.6
9	53915.607	218.28181	24.9	7.8	2	6	4,926	3	2	2011	85.7	82.0	88.8
10	208040.13	842.26774	27.7	19.9	3	6	39,950	27	14	2020	99.4	98.2	99.9
11	55051.943	222.88236	10.4	5.3	0	5	4,522	4	3	1994	20.8	17.1	25.0
12	20038.623	81.12803	16.5	19.5	0	5	5,556	2	1	1953	8.9	6.6	12.2
13	27934.423	113.09483	10.0	14.9	0	5	3,494	16	1	1969	0.3	0.0	1.4
14	20307.266	82.21565	9.2	3.9	1	5	5,639	2	1	1969	10.6	7.9	13.9
15	48682.752	197.09616	15.5	7.4	1	5	8,990	5	2	2013	61.4	56.6	66.1
16	37067.863	150.07232	32.6	41.3	1	5	8,234	18	8	2019	97.7	95.8	98.8
17	32243.916	130.54217	25.5	22.4	0	5	471	2	1	2019	96.8	94.5	98.1
18	27934.423	113.09483	17.3	0.2	0	5	4,856	1	1	2001	54.4	49.6	59.3
19	27934.423	113.09483	17.3	56.1	1	5	2,529	1	1	2004	85.9	82.3	89.1
20	87378.68	353.75984	14.9	17.5	2	5	27,052	18	6	2020	98.9	97.5	99.6
21	27768.224	112.42196	23.0	22.4	0	5	3,009	3	1	1993	19.2	15.7	23.4
22	27934.423	113.09483	19.2	17.5	1	5	4,831	1	1	2014	90.4	86.9	92.8
23	683856.28	2768.6489	24.5	85.9	4	5	67,425	121	36	2020	99.5	98.2	99.9

24	27934.423	113.09483	16.0	5.7	1	5	7,041	1	1	1955	30.1	25.7	34.7
25	50461.038	204.2957	27.7	100.0	1	5	4,239	4	4	2017	98.6	97.1	99.5
26	37589.702	152.18503	12.6	8.6	0	5	13,717	5	3	1978	11.9	9.0	15.3
27	44091.474	178.50799	21.1	100.0	2	5	3,951	4	2	2012	93.6	90.6	95.5
28	20900.663	84.61807	14.0	5.7	0	5	5,779	2	1	2019	93.7	90.9	95.7
29	46877.885	189.78901	15.1	78.2	2	5	8,637	6	2	2013	90.3	86.9	92.8
30	27934.423	113.09483	18.1	36.1	1	5	-	1	1	2014	92.6	89.8	94.9
31	27934.423	113.09483	13.0	12.9	0	5	8,024	2	1	2013	71.5	66.9	75.7
32	307262.33	1243.977	15.7	19.1	2	5	78,259	29	17	2020	98.9	97.5	99.6
33	27934.423	113.09483	20.0	100.0	1	5	755	5	1	2014	95.7	93.3	97.3
34	27934.423	113.09483	20.2	17.4	1	5	695	3	1	2000	32.5	28.1	37.2
35	23986.335	97.11067	14.3	18.0	0	5	8,730	2	2	1992	47.6	42.7	52.4
36	27934.423	113.09483	9.6	0.8	2	5	7,545	2	1	2012	81.9	77.9	85.5
37	217621.52	881.05877	13.1	20.1	3	5	57,802	33	19	2021	100.0	100.0	100.0
38	42734.905	173.01581	16.7	16.7	2	5	9,350	5	3	2016	95.0	92.4	96.7
39	52551.871	212.76061	25.9	99.8	0	5	1,459	15	6	1981	1.7	0.8	3.8
40	27934.86	113.0966	19.2	30.2	1	5	5,087	2	1	2018	96.6	94.5	98.1
41	337513.36	1366.4508	21.6	42.7	4	5	67,049	29	13	2020	99.7	98.6	100.0
42	51446.477	208.28533	37.6	100.0	3	5	10,976	8	2	2013	95.5	93.0	97.1
43	153191.44	620.20827	13.2	31.8	3	5	17,174	45	7	2013	0.3	0.0	1.4
44	18889.737	76.47667	16.7	23.2	0	5	6,768	7	1	1977	1.1	0.5	2.9
45	27934.423	113.09483	10.6	0.3	2	5	4,751	4	1	1990	8.7	6.4	11.9
46	53895.514	218.20046	37.0	5.9	3	5	14,254	7	2	1971	8.7	6.4	11.9
47	122000.68	493.92988	41.6	89.5	4	5	7,436	42	9	2014	70.3	65.6	74.5
48	32196.294	130.34937	33.1	47.7	1	5	1,473	2	2	1950	70.2	62.7	76.4
49	55167.191	223.34895	32.2	21.6	2	5	3,230	16	2	1983	3.4	1.9	5.5
50	27934.423	113.09483	31.9	4.3	3	5	2,284	3	1	2003	83.2	79.3	86.6
51	43552.631	176.32644	39.9	5.8	2	2	10,367	5	4	2009	92.5	89.5	94.7
52	91875.39	371.96514	23.5	2.5	1	4	21,217	75	5	2020	94.9	92.1	96.5

53	36138.953	146.31155	18.5	11.7	0	5	12,932	41	3	1969	0.0	0.0	94.9
54	27934.423	113.09483	18.7	0.0	2	2	4,892	1	1	1960	47.0	42.2	51.9
55	3270478.9	13240.805	28.6	20.0	15	5	621,801	947	81	2021	100.0	100.0	100.0
56	27934.423	113.09483	7.7	26.1	2	6	7,192	1	1	1963	47.6	42.7	52.4
57	27934.423	113.09483	4.6	9.2	1	6	7,701	2	1	1953	10.5	7.9	13.9
58	27934.423	113.09483	2.5	11.1	1	6	4,589	1	1	1959	21.5	15.6	28.2
59	27934.423	113.09483	21.9	36.3	1	2	3,951	3	1	2012	73.8	69.2	77.8
60	27940.675	113.12014	3.2	3.8	1	2	2,514	22	1	2007	0.7	0.2	2.2
61	35925.747	145.44837	8.4	4.5	1	6	7,541	3	2	1981	37.2	32.7	42.1
62	32520	131.65992	24.3	73.9	1	6	5,498	6	2	2002	65.2	60.5	69.8
63	87065.149	352.49048	36.6	32.1	0	2	23,485	277	19	2021	100.0	100.0	100.0
64	27934.423	113.09483	1.1	0.0	0	7	5,007	1	1	1983	37.2	32.7	42.1
65	27934.423	113.09483	15.3	5.3	0	2	4,710	1	1	2013	75.2	70.8	79.2
66	73062.64	295.80016	23.2	5.4	0	7	11,387	5	5	2014	89.0	85.6	91.7
67	27934.423	113.09483	11.8	97.1	1	2	4,366	1	1	2019	98.4	96.4	99.1
68	31584.596	127.87286	16.8	0.0	0	7	7,219	8	2	2013	80.2	76.1	83.9
69	61560.919	249.23449	38.0	1.4	1	2	15,611	3	2	2013	91.4	88.4	93.9
70	27934.423	113.09483	17.0	0.0	0	7	9,974	1	1	2011	80.5	76.3	84.1
71	371471.63	1503.9337	55.3	22.0	3	2	105,196	67	15	2019	98.6	96.8	99.3
72	27934.423	113.09483	26.8	3.7	1	2	6,510	1	1	2009	82.7	78.7	86.1
73	17834.45	72.20425	2.1	3.6	1	4	6,896	1	1	1959	19.8	12.3	30.8
74	27934.423	113.09483	11.6	33.9	1	4	4,634	5	1	2018	95.2	92.7	96.9
75	87817.483	355.53637	7.3	11.0	2	4	31,076	16	4	2013	12.5	9.6	16.1
76	27934.423	113.09483	16.9	0.0	0	4	5,080	1	1	2008	64.6	59.9	69.3
77	66958.02	271.0851	17.4	44.1	2	4	9,649	9	4	2020	97.9	96.1	99.0
78	50881.224	205.99686	10.8	11.9	1	4	13,034	2	2	2016	88.7	85.3	91.5
79	64880.401	262.67369	17.7	8.6	4	4	17,050	7	5	2020	99.4	98.2	99.9
80	27649.731	111.94223	9.7	6.8	2	4	6,321	1	1	2017	93.0	90.1	95.1
81	17685.252	71.60021	31.9	87.9	1	4	-	1	1	2009	95.3	92.7	96.9

82	27934.423	113.09483	18.8	10.0	0	4	10,474	1	1	2014	82.3	78.2	85.7
83	51166.495	207.1518	9.8	58.4	3	4	6,225	4	2	2015	96.9	94.8	98.3
84	64118.399	259.58866	11.9	4.6	3	4	6,879	13	3	2016	94.4	91.8	96.3
85	27934.423	113.09483	22.4	6.8	0	4	8,150	1	1	2014	83.2	79.3	86.6
86	46891.214	189.84297	19.0	1.0	0	4	7,009	2	2	2020	96.7	94.5	98.1
87	27934.423	113.09483	10.0	2.4	0	4	4,727	1	1	2000	45.5	40.7	50.4
88	27934.423	113.09483	24.0	0.0	1	2	7,624	1	1	2018	94.5	91.8	96.3
89	27934.423	113.09483	21.9	5.4	0	2	9,340	6	1	1993	1.1	0.4	2.5
90	27934.423	113.09483	6.5	0.2	1	4	6,371	1	1	2019	93.4	90.4	95.3
91	27934.423	113.09483	17.9	2.7	1	4	948	1	1	1980	45.3	40.3	50.3
92	47185.998	191.03643	18.0	32.3	1	4	11,659	4	2	2011	61.2	56.4	65.9
93	43237.039	175.04874	18.2	1.7	3	4	8,359	2	2	2017	96.2	93.9	97.7
94	27934.423	113.09483	10.2	0.0	3	4	3,981	4	1	1979	22.2	18.0	27.1
95	165432.32	669.76647	25.9	8.3	4	2	32,848	24	12	2021	100.0	100.0	100.0
96	27934.423	113.09483	27.6	4.6	2	2	6,794	1	1	2014	93.3	90.4	95.3
97	223321.61	904.13608	31.3	12.4	3	2	57,468	45	13	2020	99.5	98.2	99.9
98	25517.046	103.30788	7.3	11.0	2	4	7,174	1	1	1974	48.5	41.9	55.3
99	21070.36	85.3051	7.6	12.1	2	4	2,233	2	1	1960	20.7	16.7	25.1
100	27934.423	113.09483	39.3	0.7	1	2	6,455	1	1	1996	76.7	72.4	80.6
101	27934.423	113.09483	23.8	0.0	1	2	7,500	1	1	1974	43.6	38.7	48.4
102	27934.423	113.09483	12.6	0.0	3	2	8,751	1	1	2014	91.3	88.1	93.6
103	370194.53	1498.7633	22.7	66.3	6	4	70,217	58	20	2021	100.0	100.0	100.0
104	27934.423	113.09483	11.7	0.0	3	2	9,166	1	1	1980	61.3	55.9	66.3
105	229693.51	929.93324	28.1	1.4	6	2	68,316	17	8	2020	99.8	98.6	100.0
106	48007.523	194.36244	30.6	8.0	2	2	14,390	3	3	2020	99.1	97.5	99.6
107	17266.056	69.90306	22.5	49.9	0	3	-	13	1	2008	1.6	0.7	3.2
108	27934.423	113.09483	23.7	6.0	2	2	7,840	1	1	2008	86.2	82.5	89.3
109	27934.423	113.09483	26.8	13.7	1	2	6,892	1	1	1980	56.7	51.9	61.5
110	27934.423	113.09483	18.2	0.0	2	2	8,861	40	1	2018	83.0	79.0	86.4

111	53259.501	215.62551	32.7	17.0	3	3	11,888	12	3	2019	94.4	91.5	96.1
112	27934.423	113.09483	32.0	4.3	1	2	7,132	1	1	1998	74.3	69.7	78.3
113	27934.423	113.09483	22.9	2.2	3	3	6,652	3	1	2000	11.6	9.0	15.3
114	27934.423	113.09483	18.1	28.0	1	3	6,044	1	1	1996	20.3	16.6	24.5
115	75097.761	304.03952	23.3	2.5	2	2	16,245	16	3	2015	46.1	41.2	50.9
116	212304.13	859.5309	5.3	27.1	2	2	55,111	32	5	2020	98.4	96.8	99.3
117	61902.137	250.61594	34.5	1.8	2	1	20,180	3	2	2020	97.7	95.8	98.8
118	88765.366	359.37395	44.8	0.0	4	1	17,949	6	4	2020	99.2	97.8	99.8
119	43080.784	174.41613	14.5	0.9	2	1	9,866	1	1	1996	45.4	40.7	50.4
120	154045.74	623.66697	18.1	16.4	0	5	12,439	22	7	2019	95.4	93.0	97.1
121	27934.423	113.09483	23.7	0.0	2	6	676	1	1	1982	61.1	56.1	65.7
122	1344655.4	5443.9491	40.5	51.9	6	6	217,263	183	46	2020	99.7	98.6	100.0
123	27934.423	113.09483	15.3	0.0	2	6	6,406	1	1	2003	72.4	67.9	76.6
124	56026.289	226.82708	20.3	0.6	3	6	5,885	4	3	2019	98.0	96.1	99.0
125	452962.34	1833.8556	19.6	9.1	3	2	94,653	112	19	2020	96.9	94.8	98.3
126	99051.095	401.01658	16.1	12.5	1	2	19,632	22	6	2018	55.3	50.4	60.0
127	24998.527	101.20861	7.4	37.6	0	6	1,656	3	2	2014	77.6	73.2	81.3
128	19949.279	80.76631	5.1	2.1	1	4	7,784	1	1	2018	90.2	86.9	92.8
129	139132.76	563.29053	22.8	34.9	3	3	27,473	84	11	2020	97.1	94.8	98.3
130	49403.717	200.01505	27.3	7.1	2	4	11,623	3	2	1990	54.8	49.9	59.6
131	601811.2	2436.4826	50.8	34.2	6	1	126,966	436	43	2021	100.0	100.0	100.0
132	171847.03	695.73697	34.8	7.3	4	2	19,867	47	16	2020	99.6	98.6	100.0
133	27934.423	113.09483	17.1	0.0	0	4	5,086	1	1	2014	78.3	73.9	82.0

[1] Representative units: 1 – Upper Coastal Plain (South Carolina); 2 – Upper Coastal Plain (GA/FL); 3 – Atlantic Coastal Plain (Carolinas); 4 – Atlantic Coastal Plain (GA/FL); 5 – FL Peninsula; 6 – AL/FL Panhandle; 7 – Upper AL

Table A-3. Florida pinesnake population summaries for currently extant populations across low urbanization/RCP 4.5 and high urbanization/RCP 8.5 scenarios for 2040, 2060, and 2080. Numbers are reported only for the 92 populations identified as extant in current condition.

		Scenario								
		Current Condition			Low Urbanization/RCP 4.5			High Urbanization/RCP 8.5		
		Populations (% Total Extant)	Area acres (km2)	% Area	Populations (% Total Extant)	Area acres (km2)	% Area	Populations (% Total Extant)	Area acres (km2)	% Area
2040	Extremely Likely on Landscape (95-100%) High Resiliency	29 (31.5%)	10662434.4 (43167.8)	77.1	6 (6.5%)	6500689.9 (26318.6)	47	5 (5.4%)	6270996.4 (25388.6)	45.3
	Very Likely on Landscape (80-94.9%) Moderate Resiliency	37 (40.2%)	2232475.2 (9038.4)	16.1	24 (26.1%)	3627748.8 (14687.2)	26.2	25 (27.2%)	3857442.3 (15617.2)	27.9
	More Likely than Not (50-79.9%) Low Resiliency	26 (28.3%)	941039.6 (3809.9)	6.8	46 (50%)	3167397.9 (12823.5)	22.9	45 (48.9%)	3139463.6 (12710.4)	22.7
	Unlikely (<50%) Extirpated				16 (17.4%)	540112.5 (2186.7)	3.9	17 (18.5%)	568046.9 (2299.8)	4.1
2060	Extremely Likely on Landscape (95-100%) High Resiliency	29 (31.5%)	10662434.4 (43167.8)	77.1	3 (3.3%)	4985328.9 (20183.5)	36	3 (3.3%)	4985328.9 (20183.5)	36
	Very Likely on Landscape (80-94.9%) Moderate Resiliency	37 (40.2%)	2232475.2 (9038.4)	16.1	12 (13%)	3871689.2 (15674.8)	28	12 (13%)	3871689.2 (15674.8)	28
	More Likely than Not (50-79.9%) Low Resiliency	26 (28.3%)	941039.6 (3809.9)	6.8	40 (43.5%)	3437535.7 (13917)	25	38 (41.3%)	3332229.2 (13490.8)	24
	Unlikely (<50%) Extirpated				37 (40.2%)	1541395 (6240.5)	11	39 (42.4%)	1646701.9 (6666.8)	12
2080	Extremely Likely on Landscape (95-100%) High Resiliency	29 (31.5%)	10662434.4 (43167.8)	77.1	1 (1.1%)	3270478.9 (13240.8)	23.6	1 (1.1%)	3270478.9 (13240.8)	23.6
	Very Likely on Landscape (80-94.9%) Moderate Resiliency	37 (40.2%)	2232475.2 (9038.4)	16.1	10 (10.9%)	5308869.1 (21493.4)	38.4	10 (10.9%)	5308869.1 (21493.4)	38.4
	More Likely than Not (50-79.9%) Low Resiliency	26 (28.3%)	941039.6 (3809.9)	6.8	26 (28.3%)	2338159.1 (9466)	17	24 (26.1%)	1857262.4 (7519.3)	13.4
	Unlikely (<50%) Extirpated				55 (59.8%)	2918442 (11815.6)	21	57 (62%)	3399338.8 (1376.2)	24.6

Table A-4. Number of Florida pinesnake populations in each persistence probably category showing low (50-79.9%), moderate (80-94.9%), and high (95-100%) resiliency for current conditions and two scenarios at years 2040, 2060, and 2080, within each representative unit and range-wide. For future conditions population numbers presented for currently extant populations only (n=92).

Representative Unit	Scenario											
	Current Conditions (n=133)				Low Urbanization/RCP 4.5 (n=92)				High Urbanization/RCP 8.5 (n=92)			
Year - 2040	Extirpated	Low	Moderate	High	Extirpated	Low	Moderate	High	Extirpated	Low	Moderate	High
Upper Coastal Plain (South Carolina)	1	0	0	3	1	1	1	1	0	1	1	1
Upper Coastal Plain (GA/FL)	5	9	7	9	4	14	6	1	4	14	7	0
Atl. Coastal Plain (Carolinas)	3	0	2	0	0	2	0	0	0	2	0	0
Atl. Coastal Plain (GA/FL)	8	5	12	3	7	8	4	1	7	8	4	1
FL Peninsula	17	6	13	9	3	13	10	2	3	13	10	2
AL/FL Panhandle	6	4	2	5	2	5	3	1	3	4	3	1
Upper Alabama	1	2	1	0	0	3	0	0	0	3	0	0
Range-wide	41	26	37	29	16	46	24	6	17	45	25	5
Year - 2060												
Upper Coastal Plain (South Carolina)	1	0	0	3	1	1	1	0	1	1	1	0
Upper Coastal Plain (GA/FL)	5	9	7	9	10	1	4	0	10	11	4	0
Atl. Coastal Plain (Carolinas)	3	0	2	0	2	0	0	0	2	0	0	0
Atl. Coastal Plain (GA/FL)	8	5	12	3	11	7	1	1	11	7	1	1
FL Peninsula	17	6	13	9	7	14	6	1	8	13	6	1
AL/FL Panhandle	6	4	2	5	4	6	0	1	4	6	0	1
Upper Alabama	1	2	1	0	2	1	0	0	3	0	0	0
Range-wide	41	26	37	29	37	40	12	3	39	38	12	3
Year - 2080												
Upper Coastal Plain (South Carolina)	1	0	0	3	1	1	1	0	1	1	1	0
Upper Coastal Plain (GA/FL)	5	9	7	9	15	7	3	0	16	6	3	0
Atl. Coastal Plain (Carolinas)	3	0	2	0	2	0	0	0	2	0	0	0
Atl. Coastal Plain (GA/FL)	8	5	12	3	14	5	1	0	14	5	1	0
FL Peninsula	17	6	13	9	13	10	4	1	14	9	4	1
AL/FL Panhandle	6	4	2	5	7	3	1	0	7	3	1	0
Upper Alabama	1	2	1	0	4	0	0	0	4	0	0	0
Range-wide	41	26	37	29	55	26	10	1	57	24	10	1

Table A-5. Summary table showing number of Florida pinesnake populations at or above the 50% probability of persistence for current conditions and for two scenarios at years 2040, 2060, and 2080, within each representative unit. The number of populations with probability of persistence >80%, presented in parentheses. These represent the moderate and high resiliency populations.

Representative Unit	Current Conditions	2040		2060		2080	
		Low 4.5	High 8.5	Low 4.5	High 8.5	Low 4.5	High 8.5
Upper Coastal Plain (South Carolina)	3 (3)	3 (2)	3 (2)	2 (1)	2 (1)	2 (1)	2 (1)
Upper Coastal Plain (GA/FL)	25 (16)	21 (7)	21 (7)	5 (4)	15 (4)	10 (3)	9 (3)
Atlantic Coastal Plain (Carolinas	2 (2)	2 (0)	2 (0)	0	0	0	0
Atlantic Coastal Plain (GA/FL)	20 (15)	13 (5)	13 (5)	9 (2)	9 (2)	6 (1)	6 (1)
FL Peninsula	28 (22)	25 (12)	25 (12)	21 (7)	20 (7)	15 (5)	14 (5)
AL/FL Panhandle	11 (7)	9 (4)	8 (4)	7 (1)	7 (1)	4 (1)	4 (1)
Upper Alabama	3 (1)	3 (0)	3 (0)	1 (0)	0	0	0

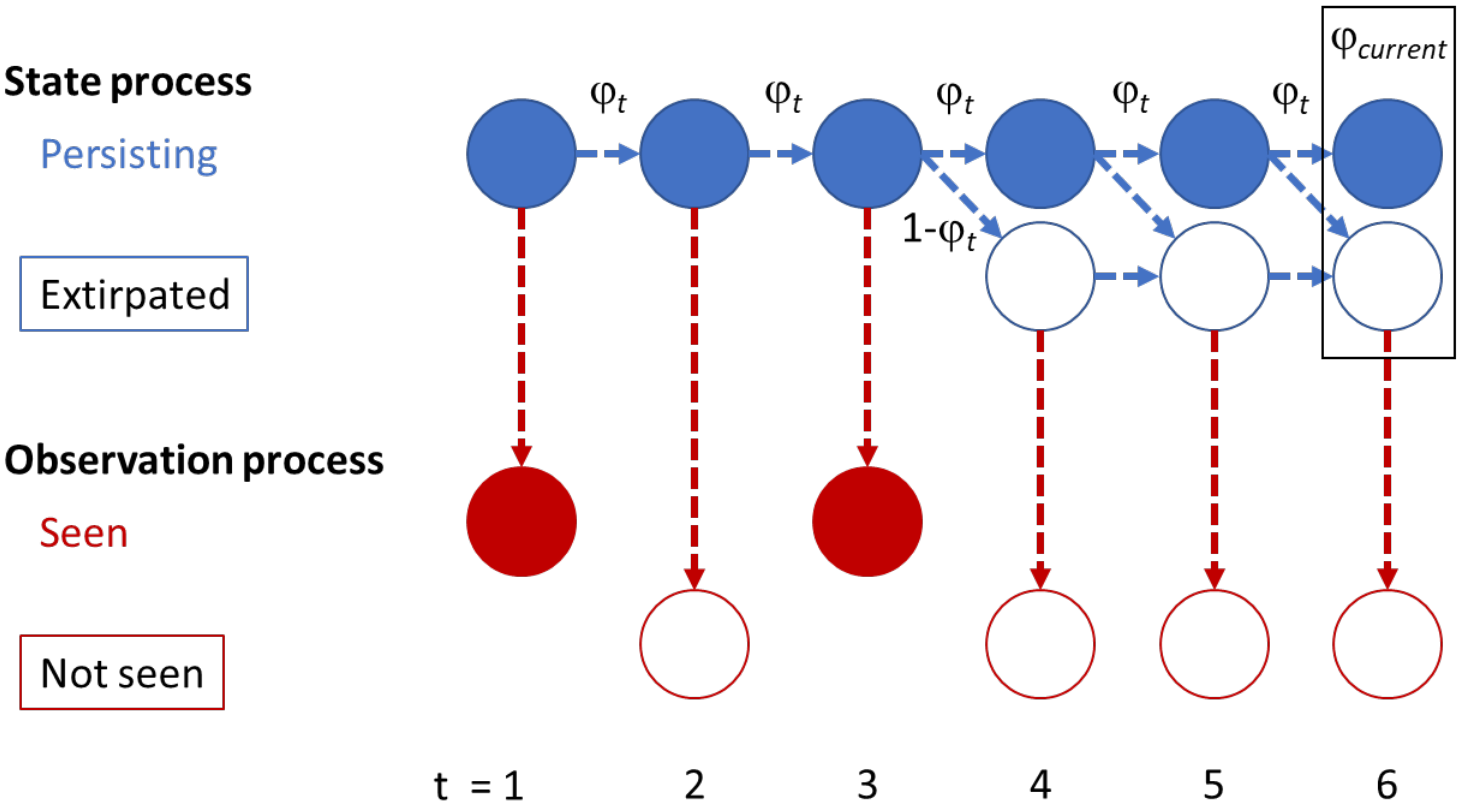

Figure A-1. Example of the state process (true state of the system) and observation process (what we observe) of a population over time for the persistence model. The population survives or becomes extirpated each year based on the annual persistence probability (φ_t), which is estimated in the model. This probability is used to derive the current persistence probability ($\varphi_{current}$) – i.e., the probability a site has survived over the entire time period modeled and is currently occupied in 2018. Its complement ($1 - \varphi_{current}$) is interpreted as the probability a population has become extirpated. If the population became extirpated in time period 4, then the sequence of the true state history would be z = [1, 1, 1, 0, 0, 0]. The observed capture history (when individuals were seen in this population) would be y = [1, 0, 1, 0, 0, 0]. Original source: Kéry & Schaub, 2012, p. 176.

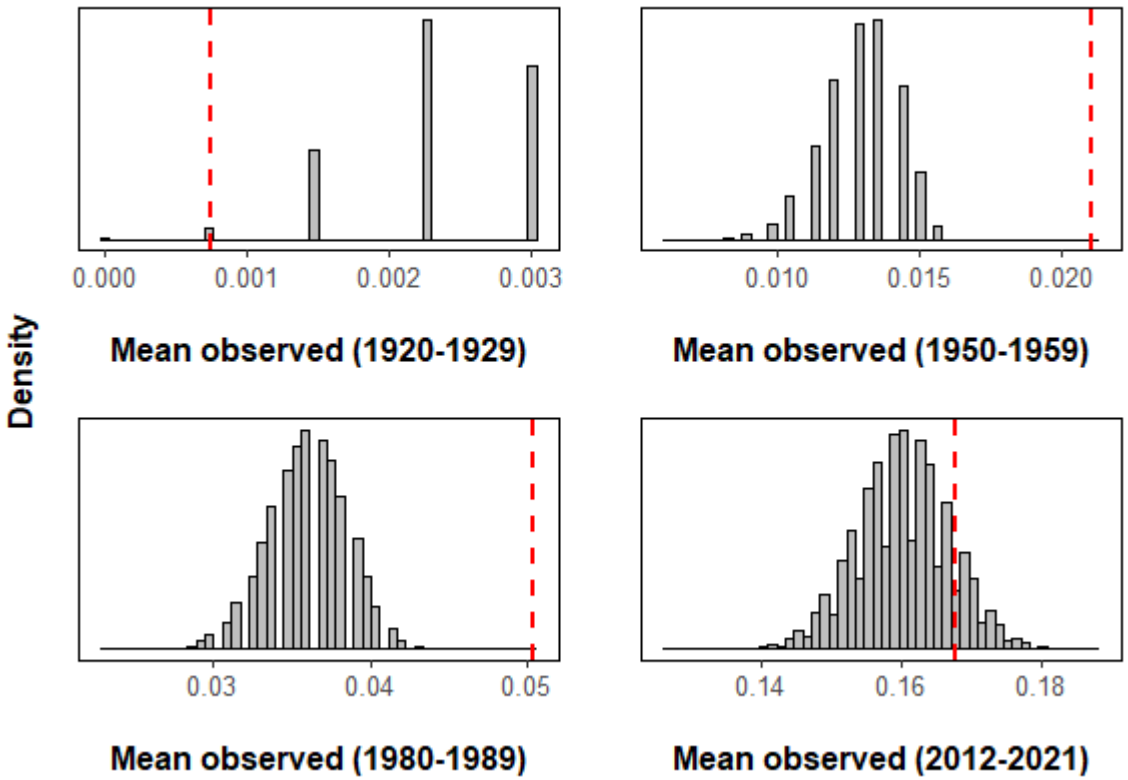

Figure A-2. Results of posterior predictive checks used to assess goodness-of-fit where we simulated datasets using parameters estimated in the model, calculated the mean number of populations with simulated detections of Florida pinesnakes in four time periods (1920-1929, 1950-1959, 1980-1989, and 2012-2021), and compared mean observations in these periods from the real dataset with values from simulated datasets. Gray bars represent the distribution of mean observations simulated within each period (n = 300,000 MCMC samples) and red dashed lines represent the mean observations in these periods from the real dataset.

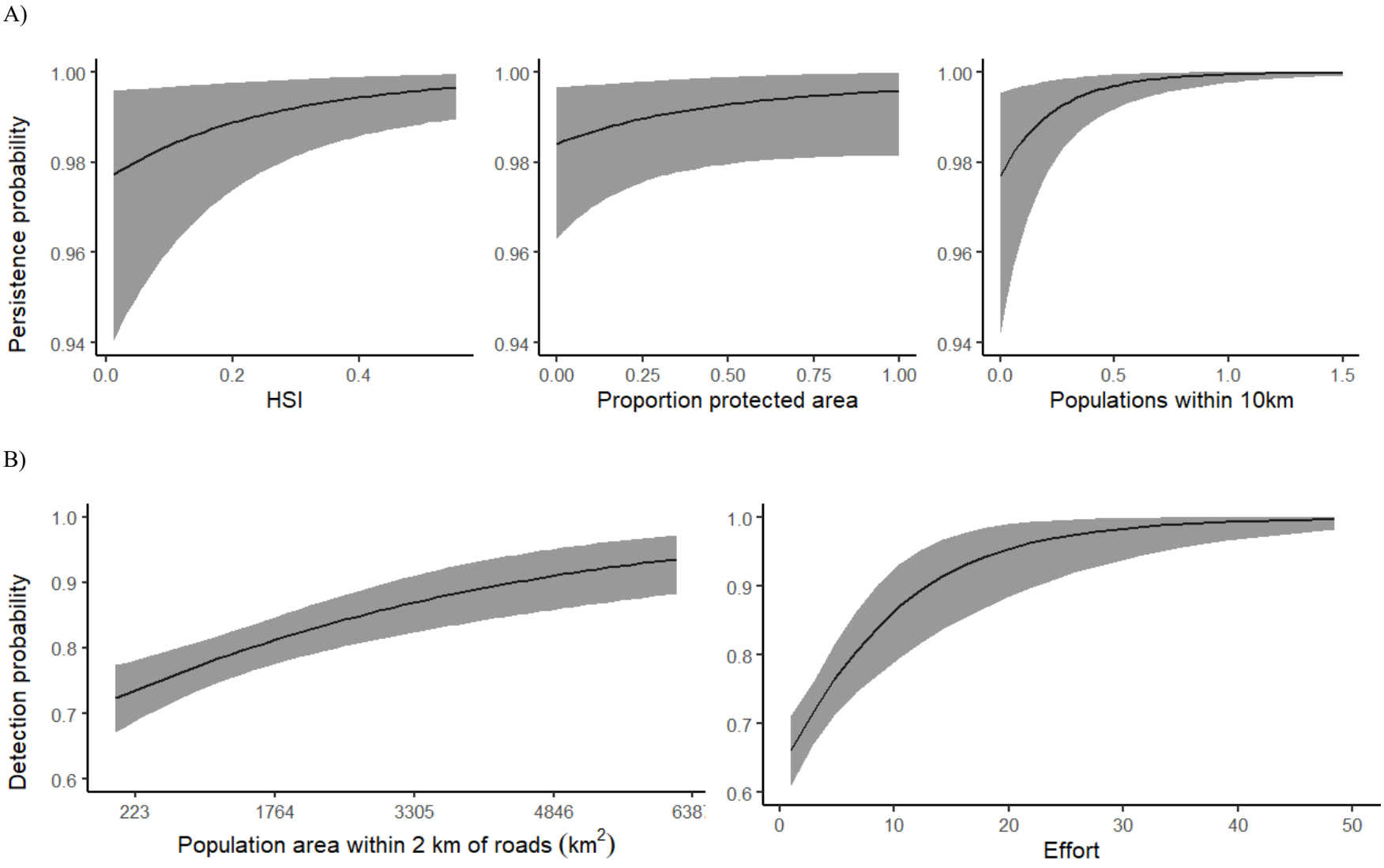

Figure A-3. Effects of each predictor variable on (A) persistence probability and (B) detection probability, holding values of other predictor constant at their respective means. For persistence probability, results are displayed for the Upper Coastal Plain (GA/FL) representative unit, which has a mean persistence estimate similar to that of most other representative units.

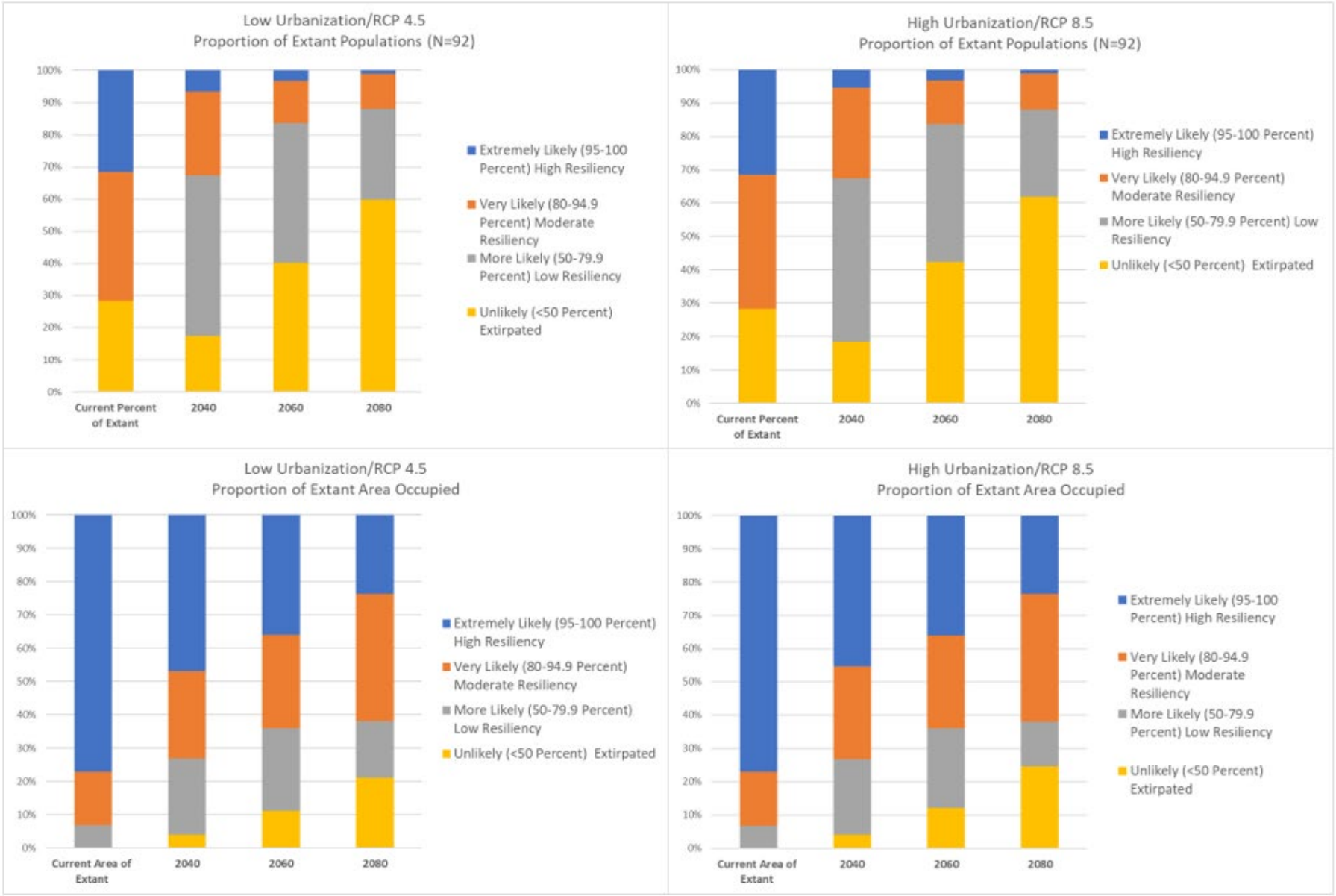

Figure A-4. Proportion of populations and percent of area occupied for all extant populations (n=92) in current condition, 2040, 2060, and 2080 for two future scenarios. Percent of area occupied calculated with area of each population for each resiliency category, provided in full in Table A-2.