

**BEACH MOUSE HABITAT ON STORM-AFFECTED BARRIER ISLANDS: INFLUENCE
OF LANDSCAPE STRUCTURE, PERCEIVED PREDATION RISK, AND SEA LEVEL RISE**

By

MARGO ANORA STODDARD

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To my friends and family who have supported me throughout this process

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LIST OF ABBREVIATIONS

CLC	Cooperative Land Cover
DEM	Digital Elevation Model
EAFB	Eglin Air Force Base.
FGDL	Florida Geographic Digital Library
FNAI	Florida Natural Areas Inventory
FWC	Florida Fish and Wildlife Conservation Commission
GINS	Gulf Islands National Seashore
GIS	Geographic Information System
Gt	Great diurnal tide range: the difference in height between MHHW and MLLW
GUD	Giving up density: the density of food at which an animal “gives up” foraging because the risk of foraging outweigh the benefits
IPCC	Intergovernmental Panel on Climate Change
LiDAR	Light detection and ranging
MHHW	Mean high higher water
MLLW	Mean low lower water
MTL	Mean tide level
NAVD88	North American Vertical Datum of 1988
NWFWM	Northwest Florida Water Management District
NOAA	National Oceanic and Atmospheric Administration
NWI	National Wetlands Inventory
NWS	National Park Service
PK	Perdido Key
PKBM	Perdido Key beach mouse
SLAMM	Sea Level Affecting Marshes Model

SLOSH	Sea, Lake, and Overland Surges from Hurricanes
SLR	Sea Level Rise
SRBM	Santa Rosa beach mouse
SRI	Santa Rosa Island
UF	University of Florida
USFWS	United States Fish and Wildlife Service

Abstract of Dissertation Presented to the Graduate School
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By

Margo Anora Stoddard

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Chair: Lyn Branch

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Habitat fragmentation is a major wildlife conservation concern because it can lead to isolation of populations, loss of habitat necessary for survival, and changes in how animals move, forage, and otherwise use their environment. Coastal areas are susceptible to fragmentation from both development and climate-related processes like sea level rise and storm surge. Understanding how these processes affect dune-dependent species like beach mice (*Peromyscus polionotus* spp) is necessary to make conservation and management decisions that ensure survival of these species. I studied three topics that address this issue. First, I examined how factors related to habitat structure (e.g., vegetation cover, patch size, corridor length and presence) and factors that elevate perceived risk of predation (artificial light, predator cues, moonlight) act alone or together to affect the foraging behavior of the Santa Rosa beach mouse (*P.p. leucocephalus*). I used amount of food eaten by mice from foraging trays as a surrogate for perceived risk. At the microhabitat scale, I found mice perceived more risk (ate less) in open than vegetated habitat and when exposed to an owl cue than cues of other potential predators (cat, fox). At the intermediate scale, I conducted three experiments using different configurations of experimental vegetation plots in which I manipulated habitat structure and risk (light, predator cues). Factors that increased risk perceived by mice (mice ate less) were

artificial lights, moonlight, and an owl cue. Mice perceived less risk when a vegetated corridor was present (in summer but not winter). At the landscape scale (1-ha grids of foraging trays in natural landscapes), mice perceived less risk (ate more) closer to vegetation patches than further away, but this response depended on the proximity of an owl cue. Second, I modeled and mapped the effect of sea level rise (0.7 m, 1.0 m, and 2.0 m by 2100) and storm surge from Category 1-3 hurricanes on beach mouse habitat in the years 2025, 2050, 2075, and 2100. Models predict the proportion of beach mouse habitat on Perdido Key and Santa Rosa Island will decline by ~6% and ~15% by 2100 (1-m rate) respectively, corresponding to ~40% loss of beach dune and 10-13% loss of scrub habitat. Many areas characterized as scrub habitat will persist through 2100 at all sea level rise rates and have a high probability of occurrence of beach mouse burrows (>0.75-1.0), likely because these habitats have high elevation and slope which are important predictors of the presence of burrows in a habitat suitability model. Category 2 and 3 hurricanes will result in surge flooding of the majority of beach mouse habitat on Perdido Key (56-98%) (across all years modeled, 1-m sea level rise rate). Finally, I used tracking tubes to examine use of natural, open sand, and planted linear dune restoration plots by Perdido Key beach mice (*P.p. trissyllepsis*). Mice used restoration plots less than natural areas but more than open sand gaps, suggesting restoration plots may facilitate movement of mice across fragmented areas. Together, these studies identify risk factors that impede or facilitate movement by beach mice under current conditions, show how beach mouse habitat is expected to become more fragmented over broad spatial and temporal scales, and suggest a habitat restoration approach that may facilitate movement of beach mice and ameliorate the consequences of sea level rise and storms on beach mice into the future.

CHAPTER 1

GENERAL INTRODUCTION

Coastal systems are dynamic landscapes subject to both natural and anthropogenic forces that cause fragmentation, habitat loss, and changes in habitat connectivity. In natural areas, availability and distribution of vegetated habitat shift as wind and wave actions redistribute sand, shaping a mosaic of tall frontal dunes along the ocean, ephemeral incipient dunes adjacent to and landward from frontal dunes, or more stable secondary scrub dunes further inland, all in various stages of development (Hesp 1991). On barrier islands, wave and storm surges push ocean water over islands carrying sediment from the ocean to the lee sides, leading to a gradual shift in island landform (i.e., island migration) (Fitzgerald et al. 2002). In the short term, surges and overwash flatten and denude large areas, potentially isolating organisms that require some vegetation structure to move. In regions like the Gulf Coast of the U.S., large storms have increased significantly since 1995, relative to the preceding two decades (Goldenberg et al. 2001, Saunders and Lea 2008). These storms have dramatically affected the distribution of dune dependent species like beach mice (*Peromyscus polionotus* spp), in part by flattening frontal dunes that are considered critical habitat for mice (Pries et al. 2008, Pries et al. 2009).

Coastal development acts alone and together with these natural processes to fragment habitat on barrier islands, negatively affecting many organisms. By blocking or changing movements of wind, sand, and water, development can rearrange landscape patterns and affect habitat connectivity. Development may isolate subpopulations, precluding dispersal and thus demographic rescue following local extinction (Oli et al. 2001). Also, the effects of introduced predators, which are common in developed areas, may be intensified in small populations. This can lead to rapid extirpation of sub-populations if such pressures are not controlled. Feral cats and red foxes in particular are known to be major threats to coastal rodents (Van Zant and

Wooten 2003, Arthur et al. 2005, Winchester et al. 2009). The negative effects of artificial light pollution in developed areas is well documented for sea turtles (Longcore and Rich 2004, Harewood and Horrocks 2008), and research on this topic is lacking but increasing for other taxa (Longcore and Rich 2004). For beach mice, artificial lighting has been shown to reduce foraging and overall seed harvest (Bird et al. 2004). By affecting movement patterns of species like beach mice, predators and artificial lights may reduce the amount of habitat that otherwise would have been used.

The threat to wildlife species already at risk from fragmentation and development-related factors is expected to increase with sea level rise (Fish et al. 2005, LaFever et al. 2007). The predicted levels of global sea level rise over the next century range from 0.18 to 1.9 m (IPCC 2007, Pfeffer et al. 2008, Vermeer and Rahmstorf 2009, Gomez et al. 2010, Jevrejeva et al. 2010). On barrier islands in the Gulf Coast, the consequence of this process will include inundation of low-lying areas, and increases in erosion and flooding from storms (Farbotko 2010). How future storms and increasing sea levels will influence the distribution of beach mice is a major conservation concern because beach mice are restricted to narrow barrier islands and coastal peninsulas, require dunes and dune vegetation for burrowing, food, and movement habitat (Humphrey and Barbour 1981, Scheckenberger 2001), and some are separated geographically into small subpopulations vulnerable to extinction (Oli et al. 2001). Understanding effects of climate change on beach ecosystems is considered a major conservation need that has been overlooked (Schlacher et al. 2007, Dugan et al. 2010).

Habitat restoration in coastal areas is one strategy that can ameliorate the effects of fragmentation caused by climate-related factors (e.g., sea level rise and storms) and development. For example, by reestablishing the structure and composition of native vegetation

and trapping sand, restored dunes can protect coastlines from storm surge and may act as movement habitat across open areas or between developed areas for some species (Austin et al. 2015). Although hypothesized to provide habitat for beach mice, whether or not mice use restored dunes has not been assessed.

To understand how coastal wildlife is affected by development- and climate-related fragmentation, in my dissertation I evaluate three aspects of the biology and conservation of two subspecies of beach mice, the Perdido Key beach mouse (*P.p. trissyllepsis*) and the Santa Rosa beach mouse (*P.p. leucocephalus*). These subspecies are found only on Perdido Key and Santa Rosa Island, barrier islands along the Gulf Coast of Florida and Alabama. All seven extant subspecies of beach mice except for the Santa Rosa beach mouse are federally listed as threatened or endangered, but all are subject to the same threats, including fragmentation and habitat loss from development or climate-related disturbances, artificial lights, and introduced predators.

In Chapter 2 I describe several experiments in which I assessed whether factors related to habitat structure (vegetation cover, patch size, corridor length and presence) and factors that elevate perceived risk of predation (artificial light, predator cues, moonlight) affect foraging by beach mice. I used foraging trays to measure how much mice ate under different conditions and at three spatial scales. When mice perceive the environment to be risky, they eat less than when they perceive conditions to be relatively safe. So, both physical habitat structures and risk factors that affect movement decisions should be considered when managing habitat for wildlife.

In Chapter 3 I describe results from modeling how beach mouse habitat changes over time (by 2100) under three rates of sea level rise (0.7 m, 1.0 m, and 2.0 m) and when subjected to storm surge from category 1, 2, and 3 hurricanes. I created maps showing changes in habitat by

land cover type and probability of occurrence of beach mouse burrows, which I used as a measure of habitat quality. I discuss implications of changes in amount, type, and quality of habitat on movement, burrowing behavior, and persistence of beach mice over time.

In Chapter 4 I address how restoration might facilitate movement and improve habitat connectivity for beach mice. I compared the proportion of tracking tubes visited by mice in dune restoration plots with that in naturally-vegetated dunes and open sand gaps. I assessed whether perceived risk changes how mice use these sites by measuring responses under full moon and new moon conditions. I discuss implications of use of restoration plots by beach mice on management and conservation of mouse habitat in landscapes that will continue to be impacted by storms into the future.

Understanding how beach mice use natural and restored habitat in these landscapes will help managers identify areas where mice may be particularly at risk from storms and other threats, and how to improve habitat to hedge against effects of these threats on beach mouse populations into the future.

CHAPTER 2

INFLUENCE OF LIGHT, PREDATOR CUES, AND LANDSCAPE STRUCTURE ON FORAGING BEHAVIOR OF BEACH MICE

Introduction

Habitat fragmentation is a key conservation concern in many landscapes, particularly in developed areas where multiple factors can exacerbate the negative effects of fragmentation on biodiversity. In coastal areas, weather-related disturbances such as tropical storms and hurricanes affect patch size and connectivity (Pries et al. 2008, Pries et al. 2009); real estate development can create barriers to gene flow (Oli et al. 2001); and artificial lights and generalist predators may affect populations directly through predation or indirectly by altering behavior of prey species (e.g., reduce movement) (Van Zant and Wooten 2003, Bird et al. 2004, Arthur et al. 2005). Together, predators and artificial lights may also reduce the amount of habitat that otherwise would be used by affecting movements of species.

A main research focus in fragmented systems has been on how the physical connectedness of habitats facilitates or impedes animal movement (Bakker and Van Vuren 2004, Tischendorf et al. 2005). However, an organism's behavioral decisions regarding movement are influenced by both changes in habitat structure as well as the amount of risk it is willing to take to move and meet its ecological needs (Belisle 2005, Zollner and Lima 2005). This risk can be real and have lethal consequences, or be perceived, such that only the threat of predation affects behavior. For example, many taxa forage less as moon phase increases (Orrock et al. 2004, Kotler et al. 2010) or in habitats exposed to predator cues such as urine (Herman and Valone 2000, Brinkerhoff et al. 2005), presumably because perceived predation risk is high under these conditions. These non-lethal effects have been hypothesized to ultimately affect the fitness of individuals and persistence of populations by reducing the amount of landscape used and thus access to resources (Brown and Kotler 2004). Understanding whether and how habitat structure

and risk interact to explain wildlife movement is necessary to inform management decisions that benefit wildlife.

The goal of this study is to identify how characteristics related to habitat structure (e.g., vegetation cover, patch size, connectedness) act alone or together with factors that can elevate perceived risk of predation (predators and light) to affect the behavior of the Santa Rosa beach mouse (*Peromyscus polionotus leucocephalus*). Beach mice occur in coastal dune systems fragmented by development and storms. They forage and burrow in vegetated dunes and vegetation patches of different sizes within a matrix of open sand, so regularly make decisions at different spatial scales regarding which type of habitat to use and whether to move across gaps and through open or covered habitat. These decisions can depend on other factors mice are exposed to, like light, which they encounter along roads and near developed areas. Predators associated with development, like foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) currently or previously have occurred in the study area (as do owls) and would be expected to influence behavior of mice as well.

I developed a set of predictions on how risk perceived by mice would change across spatial scales and with changing habitat and risk. At the microhabitat scale, I predicted nonlethal cues of predation would influence risk perceived by mice and that vegetation cover would reduce the risk of perceived predation. For this and all experiments, I measured how much mice ate from foraging trays (see details under Methods) as a surrogate for perceived risk. I tested my prediction at the microhabitat scale (experiment 1, below) by measuring foraging in open and closed patches when exposed to the cues of three potential predators of beach mice, owls, foxes, and cats. I used the predator cue with the greatest influence on foraging from this experiment in all subsequent experiments that included a predator cue.

At the patch scale, I predicted connectivity and larger patch sizes would reduce perceived predation risk more than small patches, and that artificial lights, moonlight, and predator cues would alter how mice use the landscape because of changes in predation risk. I conducted three experiments at this scale, all using planted vegetation plots in which I manipulated a combination of patch size, corridor length, and corridor presence. The corridor was a planted strip of vegetation between a patch with a foraging tray and another planted patch or a natural vegetated patch. A combination of moonlight, artificial lights, and a predator cue were applied to these experiments as well, as described below. In the first patch-scale experiment (experiment 2, below), I measured foraging in patches of two different sizes that were connected or not connected to another patch and that were exposed to artificial lights, artificial lights plus a predator cue, a predator cue only, or a control. In the second patch level experiment, I manipulated presence and length of a corridor, and measured foraging under full and new moon conditions and in different seasons (experiment 4, below). In the third patch-level experiment, I kept corridor (or gap) length static, and measured the influence of a predator cue and corridor presence on foraging (experiment 5, below). Experiment 4 was conducted to distinguish more clearly the effect of vegetation corridor presence and length, which could not be determined from experiment 2, and to test whether moonlight had an effect on foraging similar to the effect of artificial light in experiment 2. Experiment 5 was conducted to clarify how corridor presence and a predator cue might interact to affect foraging, effects that were not clear after conducting experiments 2 and 4. Main differences among experiments are summarized in Table 2-1.

Finally, at the broadest scale, I predicted a greater proportion of the landscape would be used at ambient risk levels compared to heightened risk levels and that foraging would be positively associated with vegetation cover and increasing distance from a predator cue. I tested

this prediction by measuring foraging in trays throughout four, 1-ha landscapes, two of which were exposed to a predator cue, and measuring vegetation cover and distance to each tray from vegetation and an owl cue (experiment 3).

Methods

Study System

Beach mice are a complex of seven extant subspecies of the oldfield mouse (*P.p. polionotus*) restricted to barrier islands and coastal peninsulas along the Florida and Alabama coasts. All but the Santa Rosa Island beach mouse are listed as threatened or endangered. They occur on Santa Rosa Island (SRI), a 64-km barrier island located along the Florida panhandle in the northern Gulf Coast of Mexico. I conducted all experiments within a 21-kilometer restricted-access section of Eglin Air Force Base which is undeveloped except for several sparsely distributed paved military sites. This area is characterized by fragments of frontal and secondary dunes with herbaceous and woody vegetation, scrub-covered dunes along the sound (northern) side of island, and wetland habitat (i.e., swales) generally located in between frontal and scrub dunes. Open sand gaps and coastal grasslands make up the matrix of habitat between higher elevation vegetated dunes. The main herbaceous plant species are sea oats (*Uniola paniculata*), cakile (*Cakile edentula*), beach morning glory (*Ipomoea imperati*), and beach elder (*Iva imbricata*). Bitter panicum (*Panicum amarum*) and maritime bluestem (*Schizachyrium maritimum*) are also common. Woody species include false rosemary (*Ceratiola ericoides*), woody goldenrod (*Chrysoma pauciflosculosa*), sand live oak (*Quercus geminata*), and sand pine (*Pinus clausa*).

Giving-Up Densities

To assess risk perceived by mice under different conditions, I used foraging trays to measure the giving up density (GUD), the density of resources remaining after an animal has

stopped foraging in a patch (Brown 1988). To measure GUDs, a given density of food is mixed in a substrate such that search time increases and diminishing returns are experienced as an animal forages. So, where risk of predation is high, GUDs would be expected to be higher (more food remaining, or less eaten) than where risk is low. GUDs have been used extensively to titrate food for safety in habitats that have known or hypothesized differences in predation risk (Brown 1988, Kotler and Blaustein 1995, Jacob and Brown 2000, Brown and Kotler 2004).

I used GUD trays in three experiments to determine perceived predation risk under ambient and heightened risk conditions at three spatial scales (experiments 1-3), and in two follow-up experiments to assess the influence of connectivity on movement of and risk perceived by beach mice (experiments 4 and 5). GUD trays were covered bakery trays ($\frac{1}{4}$ sheet Pactiv ClearView® Showcake® Smartlock® Bakery Containers, 14" x 10" x 4.5") baited with 5 g of dried millet seeds mixed evenly into 3 liters of beach sand. Trays were elevated to prevent access of non-target animals (e.g., ghost crabs, *Ocypode quadrata*), and a wooden dowel was inserted into the sand and leaned against two entrance holes cut near the edge of the trays on opposite sides to serve as a ramp for mice (Figure 2-1). Trays were pre-baited and deployed for 5-7 days prior to data collection. I sifted seeds from each tray three times per sampling period, weighed the seeds, and used the mean of the weights as the response (i.e., mean GUD). GUD is a measure of how much risk mice perceive at a particular tray location. See methods under each experiment for additional details.

Experiment 1: Predator Cue and Microhabitat (Microhabitat Scale)

To assess how predator cue and microhabitat influence foraging, I installed GUD trays (paired in covered and open microhabitats) at multiple points and exposed trays to heightened risk (predator cue) or control conditions. I selected sites during surveys for mouse tracks along island-wide north-south transects separated by 175-225 m. When I encountered tracks, I

searched for a potential point within 20 m. Potential points contained a vegetated patch \geq 1.5 m in diameter and an open sand area \geq 6 m in diameter. I placed one tray in the open and the other in the vegetated plot, 3-5 m away. Points were separated by \geq 175 m to ensure spatial independence among mouse home ranges and thus mice that might visit the points.

I randomly assigned one of five predator cue treatments or controls to each point (Figure 2-1). These were: 1. 3-D model of a great-horned owl (*Bubo virginianus*) and hoot playbacks; 2. 3-D fox model and scent (fox urine on dowels); 3. 2-D cat model and scent (cat feces on dowels); 4. control call (seagull call playbacks); and 5. control scent (water on dowels). Calls were broadcast between dusk and dawn from an mp3/portable speaker setup within 5 m of the feeding tray. The predator model and/or speaker, or the control, was placed equidistant from the paired GUD trays at each point. These visual cues and speakers were moved daily to avoid acclimation of mice to models or speakers. I set up 17-20 paired trays (i.e., “points”) per predator cue or control.

During a six-day sampling period, I sifted seeds from trays and re-baited them every two days. So, GUDs were collected three times per treatment and the mean of these three values was used as a response (i.e., mean GUD). I evaluated GUDs during the new moon (between the last and first quarter of the moon). I modeled the influence of predator cue, microhabitat, and their interaction on mean GUD. Point (nested in predator cue) was a random variable in the model. All pairwise comparisons of differences of least squares means (LSM) were assessed using Tukey’s tests. For this experiment and all others described below, I examined Pearson residuals of the response to assess whether errors were normally distributed and model assumptions were met. I used $\alpha=0.05$ to assess whether or not results were significant in all experiments. I used

SAS v. 9.4 for all analyses and used the MIXED procedure (unless noted otherwise) to model the influence of main effects, interactions, and random effects on foraging.

Experiment 2: Predator Cue, Lights, and Connectedness (Patch Scale)

To determine how connectivity and patch size influence foraging under different levels of elevated risk (from artificial lights and an owl cue), I measured foraging in sites each with four experimental vegetation *patch types* (described below) that differed in connectivity and patch size and comprised an experimental plot. Elevated risk treatments or controls (no risk treatments) were applied to each experimental plot. I randomly selected locations of these plots from all potential open sand gaps on Santa Rosa Island identified from aerial photos of the study area and field surveys. Potential sites were large enough to allow separation of ≥ 15 m among patch types (see below), and among patch types and the nearest natural vegetation patch ≥ 1 m in diameter.

I planted 12 experimental vegetation plots using broom sedge (*Andropogon virginicus*), which structurally resembles dune grasses and is available in large quantities. Each experimental plot comprised 4 *patch types* (Figure 2-2): 1. connected small patch and 2. connected large patch, both configurations comprising one 1-m² and one 2.5-m² patch connected by a 0.5 x 10-m corridor (area = 12.25 m²; area/edge=0.53); and 3. isolated small patch and 4. isolated large patch, both configurations comprising one 1-m² and one paired 2.5-m² patch separated by 10 m of open sand and each with a 0.5 x 2.5 m patch extending in the direction opposite the small or large patch (area = 12.25 m²; area/edge=0.51). One GUD tray was placed in each patch type, and the order of patch types within an experimental plot was randomly determined. By varying connectivity and patch size but maintaining similar habitat area and edge (area/edge), this design separates the potentially confounding effects of habitat area and configuration common in manipulative studies of fragmentation (Haddad et al. 2003, Orrock and Danielson 2005, Ewers

and Didham 2006). Similar patch sizes and linear connections are common in regenerating and established dune fragments on Santa Rosa Island. I selected 10 m spacing because gaps < 8.38 m in width have been shown to be easily crossed by mice (Wilkinson et al. 2013).

At each site, 1-3 of 4 “risk” treatments were applied during new moon periods during the experiment: 1. control; 2. owl cues (model and hoot playbacks); 3. artificial lights; 4. owl cues and artificial lights. Treatments were assigned to most sites randomly and without replacement (i.e., no site received the same treatment over different sampling sessions). I was unable to randomly apply all treatments to each site because the lighting treatment could not be applied during the turtle nesting season (May to October). So, only the control treatment was applied to some sites when others could not be deployed. For treatments including the owl, two owl models were placed between patch types, each at a different location, and were moved daily to avoid acclimation by mice to the cue. The lighting treatment comprised three 60W incandescent bulbs suspended ~2.5 m over each patch type, with one bulb each over the small and large vegetation patches, and one over the gap or corridor between the vegetation patches (Figure 2-3). Lights were powered by portable generators placed > 60 m from trays.

I sifted seeds from trays and re-baited them every two days over a 6-day sample period. GUDs were collected three times and the mean of these three values was used as a response. GUDs were evaluated during the new moon (between the last and first quarter of the moon). I modeled the influence of the risk treatments (or control), patch size (1 m^2 v. 2.5 m^2), connections by vegetation corridors, and all interactions of these effects on mean GUD. Site and site by treatment interaction were included as random effects. I assessed all comparisons of differences of least squares means (LSM) using Tukey’s tests.

Experiment 3: Predator Cue, Distance to Vegetation, and Proportion of Area Used (Landscape Scale)

To assess perceived predation risk in relation to landscape structure across broad spatial scales, I established four 10 x 10 grids of GUD trays with 10-m spacing (total area in a grid = 1 ha). Two grids were randomly assigned to receive the owl treatment and two to be controls (no treatment). I used the owl treatment because it had the strongest negative effect on GUDs, based on results from the first experiment (i.e., “predator cue and microhabitat”). Five 3-D models of a great-horned owl (*Bubo virginianus*) and speakers playing hoot playbacks were placed in the predator grids, with one owl at the center and one midway between the center owl and each of the four corners of the grid. I collected seeds from each GUD tray three times, once every two days during a six-day sampling period. The mean of these three GUDs was the response, and this is a measure of how much risk mice perceive at a particular tray location.

Using ArcGIS 10.5, I conducted an unsupervised classification of 2010 digital orthophotos of Santa Rosa Island to identify four levels of vegetation cover (open, sparse, full, dense), calculated the area of each level within each grid (# pixels) and also measured the distance from each tray location to the edge of the closer of either full or dense vegetation. These categories are based on a grouping of pixels in an image with similar spectral signatures. Based on my familiarity with the study area, the open category represented open sand, the sparse level usually represented grassland with open sand visible among grasses, and the full and dense vegetation represented vegetation that almost fully or fully covered the ground. Vegetation in the full category generally formed a buffer around the dense vegetation category. I also measured distance to owl from each tray location in ArcGIS. I first examined whether the owl treatment and percentage of vegetation in different cover categories in each grid influenced GUD averaged over each grid (n = 4). That is, I used the average of all GUD across each of the four

grids as a response, and percentage of vegetation cover in the sparse + full + dense categories, and cover in the full+ dense categories as fixed effects. To include potential local effects on foraging, I evaluated two other models using the GLIMMIX procedure; 1. The effects of owl presence and distance to full or dense vegetation on mean GUDs (mean of three GUDs collected from each tray during the sample period); and 2. The effects of distance from each tray to full or dense vegetation, distance from each tray to the owl cue, and the interaction of these factors. To account for spatial correlation among responses at trays, I included the row and column codes for each tray in a random effects model statement. The random effects variables included in these local models were grid, grid*column, grid*row, and grid*column*row.

Experiment 4: Connectedness, Distance from Dune, and Moon Phase

Because results from experiment 2 were equivocal regarding the influence of connectedness on foraging by beach mice, I conducted an experiment using a larger patch size and greater distance than those examined in experiment 2 and also examined whether moon phase (new or full) influenced foraging. I used a patch size greater than those used in experiment 2 (5 m^2 instead of 1- and 2-m^2 patches) to focus on testing the influence of connectedness on perceived risk, so I installed patches that I believed would not increase risk perceived by mice. Patches of approximately this size and greater were found by Wilkinson et al. (2013) to be used by beach mice more than smaller patches. The experiment was conducted in both the summer (June-July) and was repeated in the winter (December-January) to try to understand some unexpected results regarding the effect of corridor length in the summer, and to assess whether or not there were differences in responses across seasons. I modeled the influence of corridor presence and distance from source dune on mean GUD during each season and moon phase.

To select sites, I identified open sand areas $\geq 45\text{ m}$ in diameter and bordered by a source dune with evidence of regular use by mice or presence of an active burrow. Sites for installation

of artificial patches were randomly selected to be 10 m (n=10) or 20 m (n=10) from the source dune. In each site I installed a 5 x 5 m patch of broomsedge and placed a GUD tray at the center. I collected GUDs over 3 consecutive nights each when patches were either connected or not connected to the source patch with a narrow (~0.25 m-wide) corridor of broomsedge (Figure 2-4). So, two treatments were applied to the same site over six nights centered on the new moon or the full moon. Whether sites were connected during the 1st or 2nd 3-day period was randomly determined. Seeds were collected from foraging trays every day for three days (rather than every 2 days as in experiments 1-3). The mean weight of these three measurements (the mean GUD) was the response.

Due to some unusually cold weather during one of the winter new moon sessions, I also evaluated the influence of mean nightly temperature on GUDs. I obtained hourly nightly temperatures from NOAA's Aviation Weather Center Data site, using data collected at nearby Hurlburt Field (<http://aviationweather.gov/adds/metars/>). I averaged temperatures between 01800 and 0600 for each night of data collection and categorized them in cold ($\leq 12^{\circ}$ C) and average ($\geq 12^{\circ}$ C) temperature classes. I used individual GUDs (3 per treatment combination and site) rather than mean GUD as a response so that daily temperatures could be evaluated.

Experiment 5: Connectedness and Predator Cue

I used the same site selection and general design as that described for experiment 4. In this experiment, however, all patches (n=18) were 20 m from a source dune and the influence of an owl cue as well as that of corridors on foraging by beach mice were assessed. A site was randomly assigned to be connected or not by a strip and to have an owl cue or not for three nights. Then, the opposite combination of treatments was applied during the second three nights. So, 4 treatments combinations were evaluated (strip, owl; strip, no owl; no strip, owl; no strip, no owl). As in experiment 4, 2 treatments (3 nights each) were applied during one 6-night new

moon session (note experiment 4 also assessed the effect of moon phase, but this experiment did not). Seeds were collected every day for three days, and the mean of these three weights was used as the response (i.e., the mean GUD). I modeled the influence of owl, corridor, and the interaction of owl and corridor on mean GUD, and included site as a random effect.

Results

Experiment 1: Predator Cue and Microhabitat (Microhabitat Scale)

Mice always ate less in open compared to vegetated plots as predicted, and the negative influence of the owl treatment appeared to be stronger than that of other cues or controls, but only in open habitat (Figure 2-5). Model results show mean GUDs were influenced by the interaction of predator and microhabitat (Figure 2-5; $F=4.44$, $p = 0.003$) and microhabitat ($F=43.39$, $p <0.0001$), but not by predator cue treatments alone ($F =0.99$, $p=0.419$). Multiple comparisons verified that the owl cue had the greatest negative effect on GUDs in the open microhabitat compared to that of other treatments. In particular, after adjusting p-values for multiple comparisons, the LSM differences for the owl in the open treatment differed from that in the vegetated trays for all other predator cues or controls (cat: $t=3.80$, adj. $p=0.0097$; control call: $t=3.36$, adj. $p=0.0362$; control scent: $t=3.23$, adj. $p=0.052$; fox: $t=3.53$, adj. $p=0.022$).

Experiment 2: Predator Cue, Lights, and Connectedness (Patch Scale)

Artificial lights strongly influenced foraging activity of mice, with mice foraging less when lights were present compared to foraging under the owl treatment or the control (Figure 2-6). There was no trend indicating patch size and connections by vegetation corridors influenced mouse activity in the design used in this experiment. Model results show only treatment influenced mean GUDs ($F =7.05$, $p= 0.016$). No higher order main effects including risk treatments, patch size, connections by vegetation corridors, and all interactions were significant.

Comparisons of LSM differences verified the trend observed in Figure 2-6; only comparisons of responses from control plots differed from those in treatments with lights. These include single order comparisons of the control v. the artificial lights treatment ($t=-4.20$, adj. $p=0.016$), and v. the lights plus owl cue treatment ($t=-3.27$, adj. $p=0.053$). Comparisons of GUD responses among different treatments, patch sizes, and vegetation corridor in the control v. in the lighting treatments (e.g., difference of LSM in 1 m control patch v. 2 m artificial light patch) that differed are shown in Figure 2-6.

Experiment 3: Predator Cue, Distance to Vegetation, and Proportion of Area Used (Landscape Scale)

At the broadest spatial scale examined (4, 1-ha grids) I found no influence of the owl treatment on GUDs measured over entire landscapes (ANOVA: $F=1.34$, $p=0.247$) and there was no relationship between mean GUD and either proportion of grids containing any vegetation (all vegetation classes except open) or that of the full plus dense categories. Owl was also unimportant in explaining GUD responses at the local (tray) level ($F=0.04$, $p=0.86$). There was a strong effect of distance to full or dense vegetation ($F=22.25$, $p=<0.001$), and of the interaction between distance to owl and distance to vegetation ($F=8.33$, $p=0.004$), but not of the main effect of distance to owl ($F=0.60$, $p=0.441$). Mice perceive less risk (forage more) in trays closer to vegetation than further away, but this effect is weakened as proximity to an owl cue increases.

Experiment 4: Connectedness, Distance from Dune, and Moon Phase

In both summer and winter, GUDs in general were higher (i.e., perceived risk was higher) during the full v. the new moon (Figure 2-7). GUDs collected in each season during the full moon did not differ by distance or corridor treatment (Figure 2-7). In summer during the new moon, mice foraged more in plots with versus without a corridor (Figure 2-7A.: $F=10.95$,

$p=0.004$). Neither distance (10 versus 20 m from source dune) nor the interaction of the corridor treatment and distance was significant.

In winter during the new moon, none of the factors—distance (10 versus 20 m), vegetation corridor presence, and the interaction of distance and corridor— influenced foraging by mice (Figure 2-7B). Temperature class was the only significant effect in the model for data collected during the full ($F\text{-value}=70.59$, $p= <0.0001$) and new moon ($F = 8.05$, $F=0.006$); during both moon phases, GUDs were higher (i.e., mice foraged less) under cold versus average temperature conditions. There were no clear interactions between temperature and other effects.

Experiment 5: Connectedness and Predator Cue

GUDs generally were lower (lower risk perceived by mice) in plots with no owl versus those with an owl cue (Figure 2-8). The vegetated corridor did not influence foraging by mice. Among main effects, only the owl treatment significantly influenced the response ($F=12.18$, $p <0.001$).

Discussion

Some factors related to habitat structure examined in this study (e.g., microhabitat, corridor length and presence) were important in explaining risk perceived by beach mice. In some cases, these effects mediated the effects of a factor that elevates risk. For example, within natural landscapes (experiment 3), perceived predation risk decreases as distance to vegetation decreases, but the size of this effect depends on the proximity of an owl cue. In other cases, the influence of the risk factors seem to override any potential influence of habitat structure. This may explain results in experiment 2 in which the treatments with lights were significant in explaining GUD responses, but the owl plus lights treatment was not. The effect of lights may be so strong as to dampen effects of landscape structure (of which none were significant in this

experiment) and also override risk cues (i.e., the owl) that have been shown to be effective in reducing foraging and increasing risk perceived by mice in other studies.

By examining characteristics known to be important across three spatial scales, the experiments in this study also help identify the scales at which beach mice select habitat, and how some risk-elevating factors might affect those decisions. At the microhabitat scale, as expected, mice foraged less in open than in closed microhabitat, presumably because they perceive more risk in the open. The negative influence on foraging by the owl cue was only evident in the open microhabitat, where an aerial, visual predator like an owl would be most effective in capturing prey. Many studies show that foraging under or near vegetation cover is higher (and thus risk is lower) than in open habitats (Morris 1997, Mandelik et al. 2003), and that owls affect rodent responses to microhabitat cover, as we observed in this study (Abramsky et al. 1997, Abramsky et al. 2002, Berger-Tal et al. 2010, Thomsen and Green 2016). At the local scale in experiment 3, owls also had negative effects on foraging, but this effect depended on distance to vegetation.

Surprisingly, beach mice did not respond to cues of either of the terrestrial predators (cats and foxes), suggesting either mice did not recognize the cues of these predators and or they did not perceive high predation risk with the specific cues I used. Both of these species (*Felis catus* and *Vulpes vulpes*) are known predators of beach mice (Van Zant and Wooten 2003, Greene et al. 2016). If mice do not recognize the cues, they may not avoid areas used by these predators or alter other behaviors to reduce risk, which could make them particularly vulnerable to predation. In other sites where foxes are present and increasing, foraging (measured as the difference in foraging among exposed and sheltered microhabitats) by deer mice (*Peromyscus maniculatus*) decreased (perceived risk increased) as fox abundance increased (Orrock and Fletcher 2014).

Rodents in this system therefore must recognize and respond to fox cues, e.g., urine or feces. Deer mice at another site also avoided traps scented with fox feces (Orrock 2010). Several studies, however, show that rodents and other groups do not respond to single direct cues of predation risk like fox urine (Lima and Dill 1990, Fanson 2010). Although I used both visual and olfactory cues to represent foxes and cats, the stationary visual cues may not have been recognized by mice whereas the owl model, which moved in the wind, may have appeared more realistic. Owls were present in the study area, as were foxes but in low densities (Branch et al. 2011). Cats were not present, but have been in the past.

In patch scale experiments, I expected mice would forage more (i.e., GUDs would be smaller) in larger vegetation patches, when patches were connected, and when risk was not elevated by a predator cue and/or lights (artificial or moonlight). The lack of influence of patch size and of vegetation corridors (experiment 2) was unexpected given related factors (patch and gap size) are known to influence how mice use natural landscapes (Wilkinson et al. 2013). It is possible the artificial setup (i.e., using broomsedge) in this experiment did not include patch sizes and corridor dimensions that affect mice. For example, mice may not perceive a difference among the patch sizes I used, and the 10-m distance I evaluated may have been too short to affect movement and foraging behavior because mice may not have perceived crossing that distance as a risk. I also expected the owl cue to affect foraging alone and in combination with artificial lights, given results of experiment 1. I suspect the influence of lights is so strong that it overrides other risk factors to which mice might be exposed when present, such as owls. My observation that lights lead to reduced foraging by mice supports previous observations (Bird et al. 2004).

Strong negative light effects on predation risk were also observed in experiment 4 in which GUDs (and thus perceived risk) were higher during the full versus the new moon (Figure 2-7), an effect that Wilkinson et al. (2013) observed for beach mice and that has been observed for other species of rodents (Abramsky et al. 2002, Orrock et al. 2004, Thomsen and Green 2016). These effects of ambient and artificial light underscore the need to enforce strict lighting regulations on or near habitat that beach mice use. In areas where habitat is already limited, lighting could imperil beach mouse populations by restricting the amount of habitat available to use, even when vegetation or other characteristics appear to be adequate.

I expected mice to forage less in patches farther from (i.e., 20 m) than closer to (i.e., 10 m) natural dunes, and less in patches not connected by vegetation corridors compared to connected patches. However, I only observed a positive effect of the corridor on foraging during the summer under a new moon. Also, mice appeared to forage more with the 20-m compared to the 10-m corridors, although corridor distance was not a significant effect. In summer when food resources are sparse, mice may have had to move across the landscape to seek food regardless of whether or not the patches are connected by vegetation corridors, or whether or not gaps they cross are small or large. The trend in winter may suggest mice perceive higher risk when habitat is connected under the particular conditions of this experiment. It is possible that in some novel habitats (e.g., artificial planted patch/strip), mice may be warier of risks associated with vegetation (e.g., lie-in-wait predators) than those of open habitat (e.g., exposure to other predators) until the mice become familiar with the new habitat.

During summer, predation risk perceived by mice may be higher in general because some predators, e.g., snakes, are more active in spring and summer than in colder months. Mice may associate vegetation with protection from predators that may be active at dusk, even when

ambient light is low (new moon). Mice were more active during the full moon in the summer than in the winter. If food is scarce in summer, then mice may have to forage even under high risk (i.e., full moon) conditions to meet their energy needs.

At the landscape scale, results regarding the influence of distance to vegetation support those observed for movement of beach mice in natural landscapes. The lack of influence on GUDs of vegetation or owl cues measured at the grid scale and the stronger local effects of these factors suggest mice make behavioral decisions regarding risk and movement at fine to intermediate scales. Observations at the local (tray) scale included only distance to vegetation (and to an owl cue) in the analysis because of the importance of these variables in other studies, and due to lack of orthogonality among other variables I originally considered (e.g., distance to sparse vegetation, vegetation cover at tray). It is possible, however, that mice may respond to factors measured over a broader spatial scale that I did not consider in this experiment.

This work highlights several areas for future work and management concern. The negative effect of light on beach mice is unequivocal and strict enforcement of lighting regulations must be a priority. Testing whether mice use patch sizes and corridor lengths different from those I examined may help identify how to manage beach mouse habitat to facilitate movement, particularly after tropical storms and hurricanes that flatten dunes and reduce connectivity. This is especially important for threatened and endangered subspecies of beach mice, some of which inhabit smaller and more unstable island habitats than the Santa Rosa beach mouse. My data also suggest there may be seasonal differences in corridor use associated with resource abundance, and this weak trend needs to be evaluated further. Finally, additional research also is needed to evaluate the influence of other potential risk factors, such as snakes, on use of corridors.

Results from this study also underscore the importance of evaluating factors that can elevate risk perceived by wildlife, in addition to quantifying physical habitat characteristics related to landscape structure. For example, habitat that provides cover or facilitates movement, such as vegetated corridors, under low risk conditions can be made less effective if additional risk is imposed on the environment by predators or lights. Understanding the interaction between habitat structure and risk factors can help guide management and conservation efforts that reduce fragmentation and improve habitat connectivity for species in fragmented landscapes.

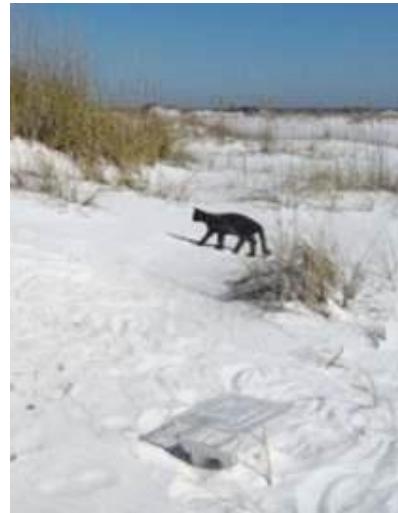


Figure 2-1. Clockwise from top left: 1. Foraging (GUD) tray used to measure the amount of risk (of predation) beach mice perceive under different conditions; 2. Owl, 3. Cat, and 4. Fox cues were placed 3-5 m from foraging trays to identify whether these cues influence foraging by mice, and thus how much risk they perceive. Photos courtesy of the author.

Table 2-1. Characteristics of five experiments described in Chapter 2.

Experiment	Spatial scale	# sites	Predator cue	Patch size (m ²)	Corridor (l x w in meters)	corridor presence	Moon phase
1	microhabitat	93	fox, cat, owl	n/a	n/a	no	new
2	patch	12	owl	1 & 2.5	10 x 0.5	yes	new
3	landscape	4	owl	n/a	n/a	no	new
4	patch	20	n/a	5	10 x 0.25 & 20 x 0.25	yes	new & full
5	patch	18	owl	5	20 x 0.25	yes	new

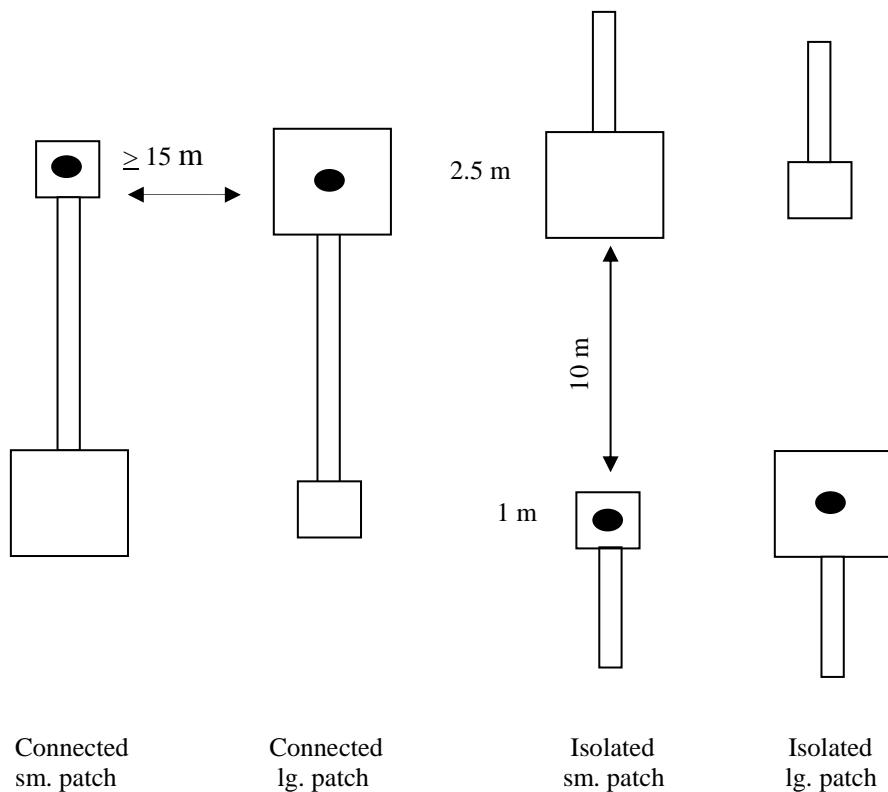


Figure 2-2. Diagram of experimental plot comprising 4 patch types. One GUD tray (black dot) was placed at the center of one square vegetation patch in each patch type.



Figure 2-3. From experiment 2: Experimental plots of broomsedge (*Andropogon virginicus*) were used to measure how patch size, connectedness of patches, a predator (owl) cue, and artificial lights influence the amount of predation risk mice perceive. A connected (left) and isolated plot (right) with the lighting treatment are shown. Photos courtesy of the author.



Figure 2-4. Experimental plot setup used in experiments 4 & 5 in which the influence of patch size, connectedness to a natural dune, and a predator (owl) cue on predation risk mice perceived by mice was assessed. Photo courtesy of the author.

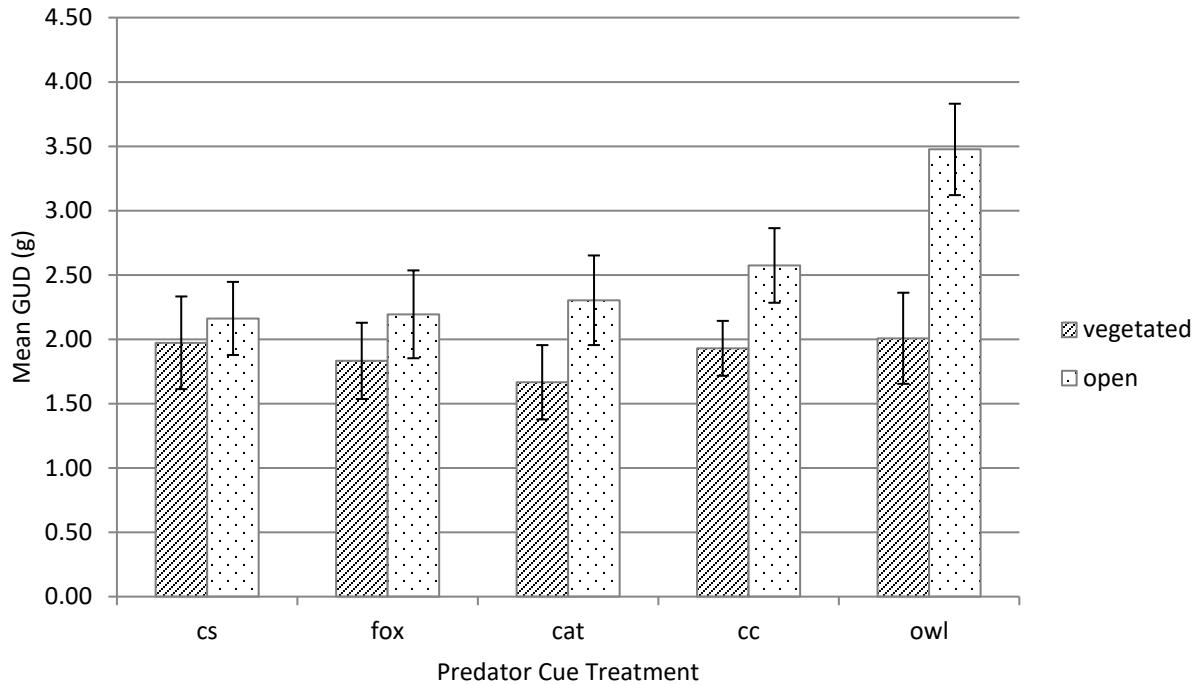


Figure 2-5. Mean giving up densities (GUD, ± 1 SE) in foraging trays in different microhabitats (vegetated vs. open) and under different predator cue treatments. Trays were originally baited with 5 g of millet seeds. Lower GUDs (fewer seeds left in tray by mice) correspond to lower perceived predation risk. cc = control scent (water); cs = control call (sea gull). The interaction of predator cue and microhabitat, and microhabitat alone were significant. Only the owl cue in open habitats negatively influenced foraging, based on differences of least squares means and Tukey's tests.

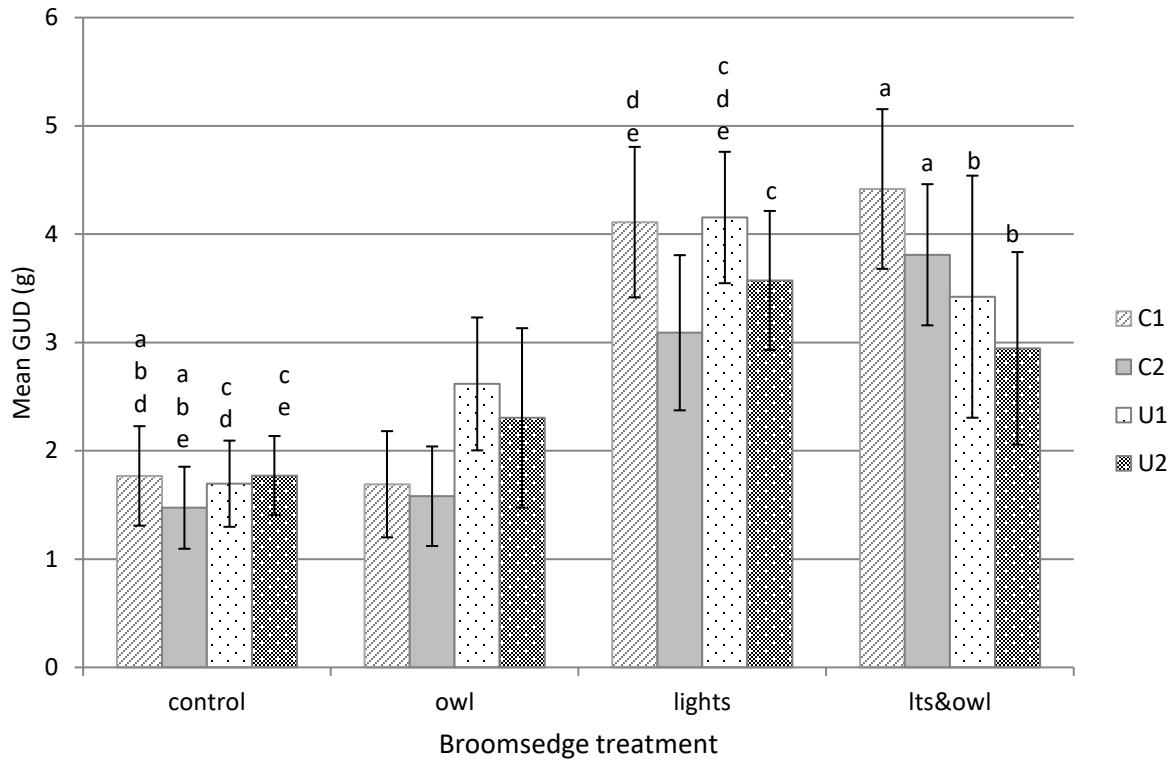
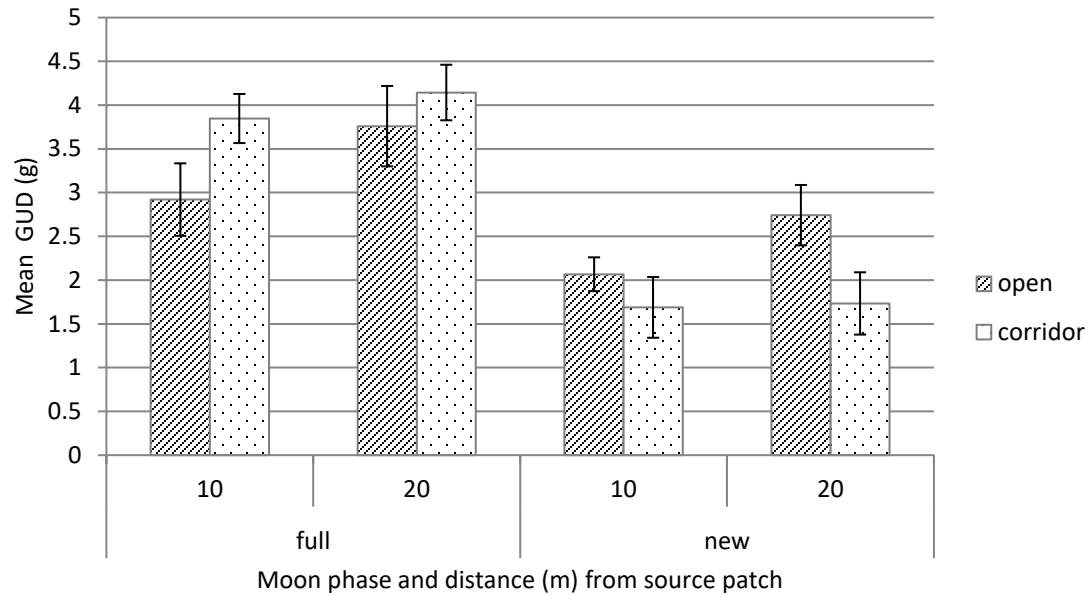


Figure 2-6. Mean giving up densities (GUD, ± 1 SE) in foraging trays in experimental vegetation plots with different patch sizes and connectedness, and under predator and/or lighting treatments. Trays were originally baited with 5 g of millet seeds. Lower GUDs (fewer seeds left in tray by mice) correspond to lower perceived predation risk. C = connected; U = unconnected; 1 = 1 m^2 patch; 2 = 2.5 m^2 patch. Note: 1-3 treatments were applied to each patch size/connectedness combination. Treatment was significant in explaining GUDs; only the control v. lights and control v. lights & owl comparisons were significant. Letters represent least squares means comparisons across treatments that were significantly different (based on Tukey's tests). For example, the "a" represents the comparison of connected patches in the control v. the connected patches in the lights & owl treatment.

A. Summer



B. Winter

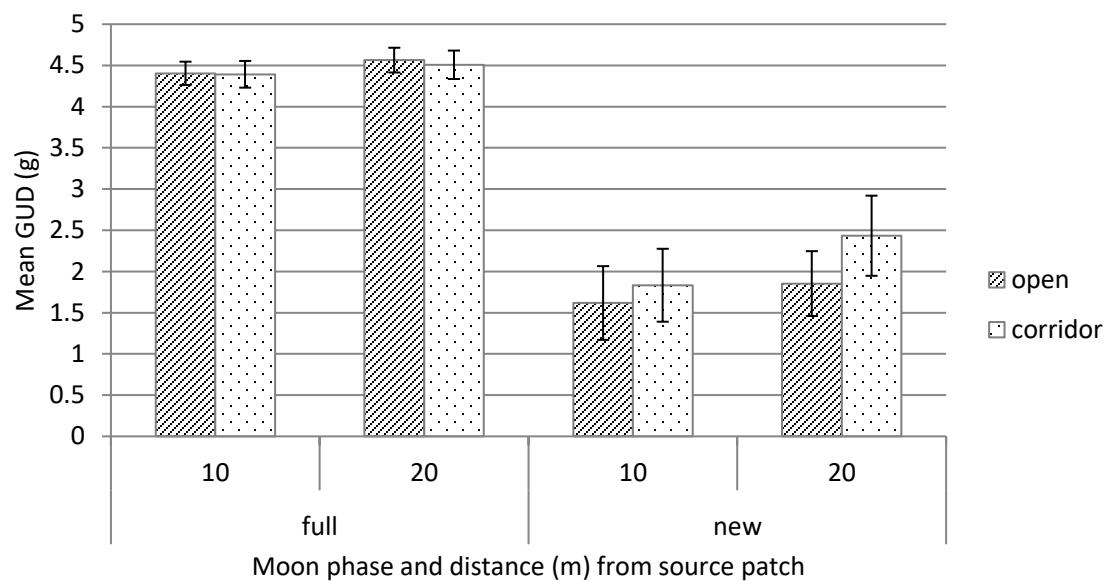


Figure 2-7. Mean giving up densities (GUD, ± 1 SE) in experimental vegetation (broomsedge) patches in two treatments at two distances (10 v. 20 m) from a natural source patch for mice, measured during full and new moon phases during summer 2012 (A.) and winter 2012-13 (B.). During the “open” treatments there was an open sand gap between broomsedge and source patches, i.e., the vegetated corridor was absent.

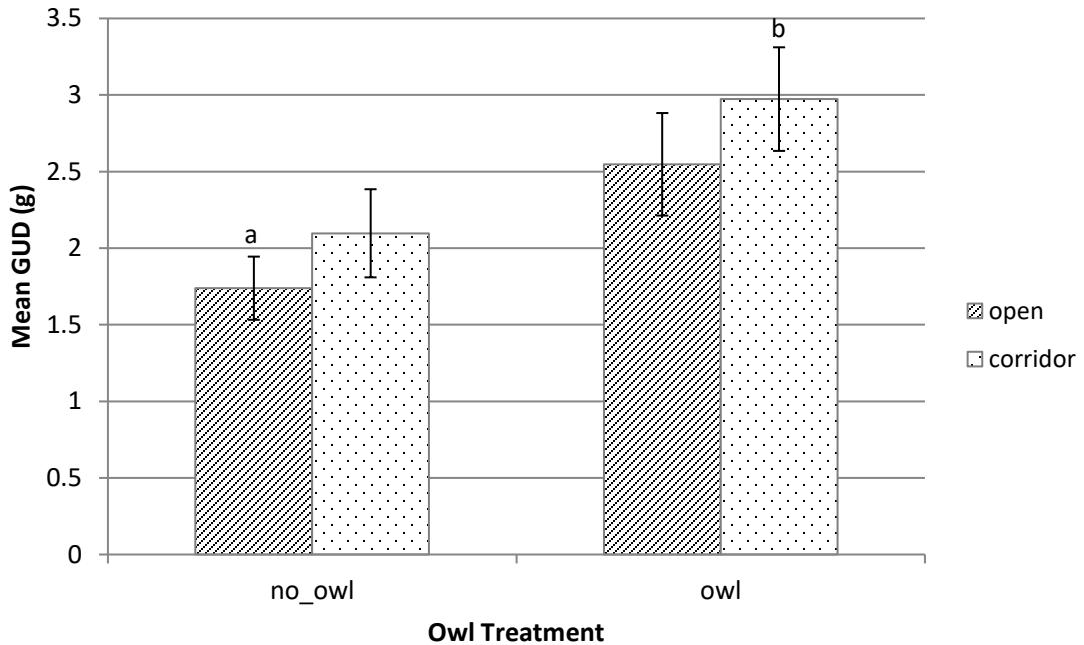


Figure 2-8. Mean giving up densities (GUD, ± 1 SE) in experimental vegetation (broomsedge) patches 20 m from a natural source patch for mice. GUDs were measured when patches were connected by a vegetated corridor or not, and/or were exposed to an owl cue (or not). During the “open” treatments there was an open sand gap between broomsedge and source patches, i.e., the vegetated corridor was absent. The owl treatment was significant. Only the bars with a different letter above them differed based on differences of least squares means and Tukey’s test.

CHAPTER 3

INFLUENCE OF SEA LEVEL RISE AND STORM SURGE ON BEACH MICE

Introduction

Habitat loss, fragmentation, and degradation are threats to biodiversity that will worsen under climate change, particularly in coastal areas. Consequences of sea level rise are already apparent along developed coastlines where infrastructure prevents natural migration inland of beaches, resulting in constriction of dune habitat between the rising ocean and built, static structures (Feagin et al. 2005, Pontee 2013). Even along undeveloped coastlines, sea level rise, which is projected to be between 0.18 and 1.9 m by 2100 (Pfeffer et al. 2008, Vermeer and Rahmstorf 2009, Gomez et al. 2010, Jevrejeva et al. 2010, IPCC 2013) will affect low-lying coastal areas as the threshold at which land floods is lowered (Nicholls 2011, Nicholls et al. 2011, Dahl et al. 2017, Vitousek et al. 2017). Exacerbating the effects of sea level rise, surge from hurricanes and tropical storms can erode and flood coastal habitat, sometimes flattening and denuding large areas. This process is especially noticeable on low-lying barrier islands where surges can push ocean water completely over islands, in effect bisecting them until water recedes. Along the Gulf Coast, large storms have increased significantly since 1995 (Goldenberg et al. 2001, Saunders and Lea 2008), and the number of intense storms and their frequency are predicted to increase through the 21st century (Bender et al. 2010, Knutson et al. 2010). Together, this increasing “storminess” and sea level rise will result in more inundation, erosion, and flooding (Otvos and Carter 2008, Moore et al. 2010, Priestas and Fagherazzi 2010) which will greatly affect the coastal landscape and species that depend on it.

Coastal wildlife will be affected by development and climate-related threats at fine to broad spatial scales. Flooding and erosion can destroy habitat for foraging, cover, and local and long-distance movements, potentially reducing fitness if organisms cannot meet their biological

needs. Development rearranges landscape patterns and affects habitat connectivity by blocking movements of wind, sand, and water, thereby altering the development of dunes and vegetation (Feagin et al. 2005, Schmidt et al. 2012). In general at broad scales, habitat loss and fragmentation, reduction in habitat connectivity, and isolation of populations can all have consequences for the long-term viability of wildlife populations (Wilson et al. 2016). For example, subpopulations that become isolated because of development, surge, or sea level rise are at risk of becoming extirpated because genetic and demographic rescue through dispersal are not possible (Oli et al. 2001, Benscoter et al. 2013). Also, small populations are more susceptible to the effects of stochastic events like hurricanes. Faced with these threats, species that depend on coastal systems will have to adapt to changing and reduced habitat, be displaced, or potentially be extirpated from some areas.

The goal of this study is to assess how sea level rise plus hurricane surge will affect the amount and location of potential habitat for beach mice (*Peromyscus polionotus* spp.). Beach mice are at risk from the combined effects of development, sea level rise, and hurricanes because 1. They occur only on a narrow low-lying barrier islands or coastal peninsulas in Florida and Alabama, areas that are especially susceptible to storm surge and sea level rise; 2. They have subpopulations separated geographically by development, meaning habitat connectivity that could facilitate dispersal and rescue of isolated subpopulations is already compromised; and 3. They rely on stable, vegetated dunes for foraging, burrowing, and movement (Humphrey and Barbour 1981, Scheckenberger 2001), habitat that can be severely degraded or destroyed by storm impacts such as wind, waves, and surge (Pries et al. 2008, Pries et al. 2009).

Information from this study will help managers identify areas where mice may be particularly at risk from storms and other threats, areas that may serve as refugia, and how to

improve habitat to hedge against effects of threats to beach mouse populations into the future. The approach I describe can be used to assess risk of sea level rise and storm surge for other coastal organisms of conservation or management concern as well.

Methods

Study Area and Study Species

My study area comprised two barrier islands in the northern Gulf of Mexico, Perdido Key and Santa Rosa Island. Perdido Key (Figures 3-1 and 3-2) is a 26 km-long barrier island located south of Pensacola, FL. Commercial development and privately-owned homes are interspersed along the island between stretches of undeveloped publicly owned or administered lands, e.g., (from west to east) the Florida Point Unit of Gulf State Park (in AL), Perdido Key State Park, and the Perdido Key Area of Gulf Islands National Seashore. These public areas comprise primarily small foredunes and low scrub dunes landward of the beaches. Gulf State Park is located at the western end of Perdido Key and is ~1.8 km in length and 480 m in width at its widest point. Perdido Key State Park is ~3.3 km in length by ~500 m in width and is bisected lengthwise by State Road 292. This area includes a 400 m² area of development on the north side of the road. The Perdido Key Area of Gulf Islands National Seashore covers the easternmost 10.7 km of the island and most is 140-280 m in width. At the eastern end, the island curves to the north as it widens to 800-900 m across and then curves to the west, forming a peninsula within the sound. The hardscape footprint of Fort McCree (active 1834-1947) is located at the eastern end of the island. A 4-km two lane road runs from the western entrance of the park eastward (Figure 3-2).

Santa Rosa Island (Figure 3-3) is a barrier island 64 km long and ~0.5 km wide located just to the east of Perdido Key (Figure 3-1). Like Perdido Key, Santa Rosa Island comprises a patchwork of private, government, and publicly administered land. Public and government areas

mentioned in this chapter include (from west to east): 1. the Fort Pickens Area of Gulf Islands National Seashore (~12 km in length). The westernmost 4 km in this area are ~600-950 m in width and the rest is ~350 m in width; 2. the Santa Rosa Area of Gulf Islands National Seashore (~15.7 km x ~0.5 km); and 3. a contiguous 21-km section of Eglin Air Force Base (~400-900 m in width). Towns located between public and private areas are (from west to east) Pensacola Beach, Navarre Beach, and Ft. Walton Beach. On Gulf Islands National Seashore and Eglin Air Force Base, Santa Rosa Island is characterized by fragments of frontal and secondary dunes with herbaceous and woody vegetation, scrub-covered dunes along the sound (northern) side of island, and wetland habitat (i.e., swales) generally located in between frontal and scrub dunes.

The main herbaceous plant species on Perdido Key and Santa Rosa Island include sea oats (*Uniola paniculata*), cakile (*Cakile edentula*), beach morning glory (*Ipomoea imperati*), and beach elder (*Iva imbricata*). Bitter panicum (*Panicum amarum*) and maritime bluestem (*Schizachyrium maritimum*) are also common. Woody species include false rosemary (*Ceratiola ericoides*), woody goldenrod (*Chrysoma pauciflosculosa*), sand live oak (*Quercus geminata*), and sand pine (*Pinus clausa*).

Since 1851, 114 tropical storm systems have passed within 100 miles of the study area, with 21 passing since 1990 (NOAA 2017d). The most recent, severe storms (all Category 3) to have struck the area were Hurricanes Opal (1995), Ivan (2004), and Dennis (2005). Ivan and Dennis caused large areas to be overwashed by seawater and destruction of up to 70% of frontal dunes (Pries et al. 2008). Hurricane Katrina (2005) severely impacted Perdido Key in particular, even though its path was ~120 miles to the west. Tropical Depression 10 (2007), and Tropical Storm Claudette (2009) are other recent storms that resulted in major flooding and erosion of beach habitat in the area (D. Miller, pers. comm.).

The Santa Rosa beach mouse (*P.p. leucocephalus*) is the only subspecies of beach mice (among seven) that is not listed as threatened or endangered and is found throughout public lands on Santa Rosa Island. Perdido Key beach mice (*P.p. trissyllepsis*), found on Perdido Key, are restricted to three main areas, Gulf State Park, Perdido Key State Park, and the Perdido Key area of Gulf Islands National Seashore. Perdido Key beach mice were extirpated from Gulf State Park following Hurricane Opal in 1995 (Oli et al. 2001), and remaining sub-populations on Perdido Key State Park and Gulf Islands National Seashore have fluctuated, sometimes occupying only a small percentage of available habitat (USFWS 2007). Hurricanes Ivan, Dennis, and Katrina (2004 and 2005) were especially destructive of Perdido Key beach mouse habitat; following these storms, only thirty individuals were captured on public lands in 2005. Subsequent trapping efforts showed that by 2009, Perdido Key beach mice were present throughout Gulf Islands National Seashore and were present in Perdido Key State Park (D. Greene, pers. comm.). In 2010, captive-bred Perdido Key beach mice were reintroduced to Gulf State Park (Greene et al. 2016), and the population increased steadily over the five years following the reintroduction (Greene et al. 2017). Whether or not this subpopulation is now breeding with individuals from the adjacent Perdido Key State Park sub-population is currently unknown.

Sea Level Rise Modeling Approach

I used the Sea Level Affecting Marshes Model (SLAMM)(Clough et al. 2012) to assess sea level rise on Perdido Key and Santa Rosa Island. I chose to use SLAMM initially because it accounts for the dynamic nature of the effect of sea level rise on habitat. Unlike other approaches, it simulates dominant processes associated with sea level rise, including inundation, erosion, overwash, and saturation. The processes applied and how they are applied depend on the habitat categories present in the study area, the elevation and slope values assigned to model

input data cells (raster files), and the values of other input variables (Clough et al. 2012). Using this input data, SLAMM is able to calculate how habitat types change dynamically as sea level rise progresses (Appendix A, Figure A-1).

A main drawback of the SLAMM approach to modeling is that SLAMM focuses primarily on wetlands (as opposed to inland, dry habitats), and SLAMM categories for upland habitats are coarse. Because of this, output of the model can lack specific habitat information for species that use uplands. To create the SLAMM land cover input file, available land cover information must first be converted into the specific SLAMM cover types. I converted habitat information for the study area into SLAMM categories by combining (if necessary) and re-coding (into SLAMM categories) Cooperative Land Cover (CLC) habitat types (Knight and Knight 2010) per methods described in Geselbracht et al. (2013; See below for a description of the Cooperative Land Cover data). Some SLAMM habitat categories comprise more than one CLC habitat type that beach mice potentially use, as well as some CLC types that are likely not used by beach mice. For example, the SLAMM category “undeveloped dry land” comprises nine CLC cover types, but only three of these are considered potential beach mouse habitat, i.e., coastal grassland, coastal scrub, and scrub (Table 3-1). To be able to include some habitat detail lost by having to use the coarse SLAMM categories in the model, I converted SLAMM model output *back* into the more detailed Cooperative Land Cover (CLC) habitat types to present in the results. So, the constriction of island area is still based on the simulation of inundation, erosion, overwash, and saturation of the SLAMM model (Appendix A, Figure A-1). By overlaying CLC habitat categories on SLAMM results, I show more habitat detail and also remove areas categorized as beach mouse habitat in SLAMM that should not be categorized as such, e.g., hammocks and flatwoods (Table 3-1). For the barrier island habitat I modeled, results based on

SLAMM and those based on the CLC overlay of habitats were very similar, and the benefit of the increase in habitat detail using CLC overrode the drawback of not showing the few dynamic changes in habitat SLAMM showed in my study area. See Appendix A for more details on the comparison of SLAMM- and CLC-based results.

Since its initial development (Park et al. 1986), SLAMM has been released with a graphical user interface SLAMM 6.2 beta (Dec. 2014: <http://warrenpinnacle.com/prof/SLAMM/index.html>), and has been applied to model sea level rise of other coastal species in Florida (Aiello-Lammens et al. 2011, Chu-Agor et al. 2012, Convertino et al. 2012). Performance of the SLAMM model was recently assessed for part of the Florida Gulf Coast through a retrospective analysis of sea level rise based on changes in coastal wetland systems that have been rigorously measured and documented (Geselbracht et al. 2011). Global uncertainty of the model (v. 5) and sensitivity of input factors have also been assessed for the study area (Chu-Agor et al. 2011). I followed in large part the methodology outlined in a broad-scale analysis of estuarine systems in the Gulf of Mexico, including one that covered part of the study area (Geselbracht et al. 2013, Geselbracht et al. 2015).

SLAMM requires a minimum of three input data files to model changes in sea level rise: 1) elevation (in m) from a digital elevation model (DEM); 2) slope (in degrees); and 3) land or vegetation cover (converted to SLAMM wetland/cover categories):

1. Digital Elevation Model. I used a DEM mosaic of Florida (5-m cell size, elevation units in cm) created by the University of Florida GeoPlan Center. The entire DEM was created using data from four sources, with the primary source for the study area being the Northwest Florida Water Management District (NFWFMD) DEM (Appendix B, Table B-1). The NFWFMD DEM was derived from bare-earth Light Detection and Radar (LiDAR) points

collected in early 2006. I used the mosaic containing this LiDAR dataset because it had the most recent, complete coverage for the study area when the analysis was conducted. Because this dataset only covered Florida, I mosaicked it with another (Appendix B, Table B-1) to have complete coverage of Perdido Key in Florida and Alabama, I clipped the combined DEMs to the study area and converted elevation units from cm (whole numbers) to meter units with decimals by multiplying the DEM by 0.01 using Raster Calculator in ArcGIS 10.5. I used the original cell size (5.43 m) and reference system (Albers Conical Equal Area) of the input mosaic DEM for all input files. Using the extent and cell size of the study area DEM, I used the slope function in ArcGIS 10.5 to create a slope file.

2. Land cover data. Land cover data were based on the Florida Cooperative Land Cover Map (CLC v. 2.3, available at: <http://www.fnai.org/LandCover.cfm>), which combines information from statewide and local land cover sources (Knight 2010). Original statewide source data comprised the Florida Fish and Wildlife Commission Florida Vegetation and Land Cover data set (2003) and the Water Management District Land Use Land Cover data (2004 for the NFWFMD). Land cover data for Eglin Air Force Base was based primarily on field work conducted in 1997. Updates to CLC v. 2.3 completed in 2009-2010 included careful revisions of coverage of FNAI focal natural communities, including scrub, as well as improvements to coastal habitat maps (FNAI 2010, Knight and Knight 2010). Most CLC habitat types are fully described by the Florida Natural Areas Inventory (FNAI 2010). Data were downloaded as polygons and were clipped to the study area. CLC landcover categories were reclassified into SLAMM wetland categories based on crosswalk criteria defined by Geselbracht et al. (2013), and then data were converted into a raster file. I chose to use v. 2.3 rather than a more recent

update of CLC (v. 3.1) because classification of some habitat types in v. 2.3 better matched known habitat boundaries in the study area than did that in v. 3.1 (pers. obs.).

3. Parameter inputs. The DEM, slope, and land cover files were all converted to .txt files, as required by SLAMM. I used the clipped DEM of the study area as a template to ensure the slope and land cover files had the same extent and cell size. Additional parameter inputs required by or used to calculate parameters for SLAMM are listed in Tables B-2 and B-3 (Appendix B). I used 2007 as the NWI photo date as an approximate midpoint between the origin of much of the (state wide) source data for the CLC coverage (2003-2004) and updates to more local parts of the study area completed in 2009-2010. I selected the option to protect developed dry land from inundation under the different sea level rise scenarios because it was reasonable to expect sea walls and other flood-protection structures would be installed to protect developed dry land in the study area, e.g., Perdido Key Beach, Pensacola Beach, Navarre Beach, and Fort Walton Beach. Model performance was assessed partially by comparing the change in wetland categories between the initial condition (2007, vegetation cover date) and Time zero of the model (1990) (Appendix B, Table B-4), as suggested in the Technical Documentation for the model program (Clough et al. 2012).

Sea-level rise scenarios modeled. I assessed effects of three rates of sea level rise, all by the year 2100, 0.7 m—which corresponds to the A1B (maximum) scenario described by the Intergovernmental Panel on Climate Change (IPCC 2007)—and fixed 1-m and 2-m rates. Like some scenarios defined by the IPCC, models in SLAMM begin in 1990. So, sea level rise is already assumed to have occurred between 1990 and the starting year of a model run, or the “initial condition” as defined by the dates of the DEM and land cover data for a study site. If the initial condition date is after 1990, the amount of sea level rise by 2100 will be slightly lower

than the rate being modeled. For each scenario and time step specified, SLAMM outputs tabular data of the number of cells in each SLAMM cover category and a raster file of SLAMM cover categories for the entire study area. Using this output, I identified SLAMM cover categories within the study area that were potential beach mouse habitat (Table 3-1) based on my understanding of and prior research on the ecology of beach mice. Results presented in this chapter (i.e., the CLC overlay of habitat types on potential habitat output by SLAMM) are based on this potential beach mouse habitat and not SLAMM output for the entire study area. Some summary results of changes in all SLAMM habitat types for Santa Rosa Island (i.e., including non-beach mouse habitat) are presented in Appendix C, Figures C-1 and C-2.

Beach Mouse Habitat Maps

To create CLC-based habitat maps for beach mice, for each sea level rise scenario and time step, I removed areas comprising SLAMM categories that were not beach mouse habitat (see Table 3-1) from the SLAMM raster output of the entire study area. This resulted in raster GIS maps of SLAMM cover types representing potential beach mouse habitat. I used these maps as templates on which to overlay a raster map of CLC habitat categories, adding the habitat detail provided by CLC to the original SLAMM output. I then removed CLC categories that were not beach mouse habitat (Table 3-1), leaving a raster GIS map of CLC cover types of potential beach mouse habitat. To make maps, I used a multi-step process in the Model Builder application of ArcGIS 10.5 and ran the process iteratively for each sea level rise scenario and time step (2007, 2025, 2050, 2075, and 2100). Graphs were created using tabular data associated with these maps.

Beach Mouse Habitat Suitability Models and Probability Maps

I created habitat suitability models using data associated with 43 beach mouse burrow locations on the Eglin Air Force Base section of Santa Rosa Island and 43 randomly-generated

points in the same area. Burrow locations were originally identified in 2009 using telemetry to follow radio-collared mice to their daytime locations (Branch et al. 2011). The set of random points was generated with Hawth's Tools (Beyer 2004) in 2011 with the objective of comparing characteristics at random v. known burrow sites and identifying which habitat characteristics are important in burrow site selection. Almost all burrows from 2009 (29 of 43) were found in 2011 when habitat at known burrow sites and random points was characterized in the field. There was little storm activity between 2009 and 2011 and this may explain the longevity of the burrows and suggests there was little change in burrow site characteristics between 2009 and 2011. I considered all active burrows identified in 2009 to be surrogates for mouse presence and all randomly selected points to represent lack of presence, and modeled the influence of slope and elevation at each point on mouse presence using the GENMOD procedure in SAS 9.4 (SAS Institute 2013), specifying a binomial distribution and logit link function. Elevation values were obtained from a LiDAR-derived DEMs from 2007-2010 (Appendix B, Table B-1) and slope values were calculated from these elevations using the slope tool in the Spatial Analyst extension of ArcGIS 10.5. I used only these variables because they have been shown in other studies to be important in describing beach mouse burrow locations (Lynn 2000, Scheckenberger 2001) and they could be estimated objectively for the entire study area using widely available LiDAR data. Using known (active) burrow locations as an indicator of species presence or of habitat selection has been used to model habitat suitability and/or selection by other burrowing wildlife species (Baskaran et al. 2006, Roger et al. 2007, Lau and Dodd 2013).

I evaluated the fit of the model using three methods. In the first, I examined the agreement between predicted values (i.e., probability of burrow presence) and observed values using a 2 x 2 classification table to calculate four indices; specificity and sensitivity (proportion

of sites where predicted and observed values agreed), and false positive and false negative fractions (proportion of sites where predicted and observed values did not agree). To fill in the table, a probability cutoff value had to be specified. Those values above this cutoff would be classified as predicting a burrow to be present and those below as predicting a burrow to not be present (Pearce and Ferrier 2000). Rather than using 0.5 arbitrarily as a cutoff, for each 0.10 increment on a probability scale, I calculated sensitivity, specificity, the number of false positive and false negative values, and accuracy (the percentage of the burrow data points correctly predicted by the model to be present or not present). I selected the cutoff where accuracy, sensitivity, and specificity were maximized and the number of false-positive and false-negative values were minimized (this ended up being 0.40). Using these criteria to select a cutoff value, the cutoff neither defines habitat very leniently (low threshold), as might be used for a very common widespread species, nor very strictly (high threshold), as might be done for a very endangered species (Pearce and Ferrier 2000). The second approach used to measure model fit was calculation of the Kappa statistic, which is a measure (between 0 and 1) of agreement between model prediction and observed data, not considering the agreement due to chance. A Kappa of 0 suggests poor (or chance) agreement and 1.0 suggests perfect agreement (Landis and Koch 1977). Third, I calculated the area under the curve (AUC) for a receiver operating characteristic (ROC) curve (using options in the LOGISTIC procedure in SAS), which plots sensitivity (true positive rate) against 1-specificity (false positive rate) across multiple thresholds. AUC essentially represents the proportion of time the classification of a positive or negative instance (i.e., presence or absence of a burrow) is assigned correctly, given the model. An AUC of 0.5 is equivalent to random assignment and of 1.0 suggests a model is perfect. I assessed model performance using all available data rather than validating with a partitioned subset to best

estimate realistic error rates for this small dataset (Fielding and Bell 1997, Pearce and Ferrier 2000).

To calculate probabilities of burrow presence, I input the elevation values from the digital elevation model of the study area and the slope values from the slope file representing the study area into the GLM model statement (i.e., the habitat suitability model described above). I used the Raster Calculator tool in the Spatial Analyst extension of ArcGIS 10.5 to do this. I calculated the proportion of the study area in each of four probability categories; 0-0.25; >0.25-0.50; >0.50-0.75; >0.75-1.0 for each sea level rise scenario and time step. These probabilities correspond to the likelihood of finding a beach mouse burrow in a particular area. Areas with lower values (i.e., ≤ 0.25) are presumed to be less likely to have burrows and be habitat that could sustain beach mice than those with higher values (i.e., >0.25). Probability maps of potential beach mouse habitat, sea level rise scenarios, and time steps (2007, 2025, 2050, 2075, and 2100) were created with the Model Builder application in ArcGIS 10.5. I created graphs of probability categories using tabular data associated with these probability maps.

Hurricane Surge Modeling Approach

To model the impacts of storm surge on potential beach mouse habitat on Perdido Key, I used the National Hurricane Center's Sea, Lake, and Overland Surges from Hurricanes (SLOSH) model display program (v. 1.78) to obtain surge values for Categories 1-3 hurricanes (Saffir-Simpson Wind Scale, Appendix D). SLOSH is a model developed by the National Oceanic and Atmospheric Administration (NOAA) and the National Weather Service (NWS) to estimate hurricane surge heights from historical, hypothetical, or predicted hurricanes (Jelesnianski et al. 1992). It has been used extensively by resource managers and planners and has been validated multiple times since it was first developed in the mid-1980s (Jelesnianski et al. 1992, Houston et al. 1999, Glahn et al. 2009, Forbes et al. 2010, Forbes et al. 2014).

In the SLOSH display program, two products based on multiple SLOSH runs can be generated. The Maximum Envelope of Water (MEOW) is the maximum storm tide elevation for a particular storm category, forward speed, trajectory, and tide level (mean or high); and the Maximum of MEOWs (MOM) is the composite of maximum storm tide elevations (i.e., MOMs) for all hurricanes of a specified category and initial tide. MOMs do not take into account one particular forward speed, storm trajectory, and landfall location as do MEOWs but rather provide the maximum surge height for each cell based on multiple MEOWs. I modeled the MOM of Pensacola Bay Basin which is available to download (named Pensacola Bay v. 4, epn3) (NOAA 2017e) at high tide to generate worst-case high water values across Perdido Key for each hurricane category modeled. I wanted to model this worst possible scenario because SLOSH does not model the effects of waves (on top of surge) or flooding from precipitation, so output depicts flooding less severe than would occur if these factors were considered. Also, MOMs incorporate results from multiple storm trajectories, speed, and landfall locations, which I preferred to selecting specific but potentially arbitrary input parameters, which MEOWs would require.

I combined output from SLOSH and SLAMM (1-m sea level rise rate only) to create maps showing surge heights on Perdido Key in 2025, 2050, and 2100. For each hurricane category (i.e., 1-3), SLOSH output water levels (i.e., surge heights) as a GIS shapefile which I converted to a GIS grid in ArcGIS 10.5. I subtracted the DEM elevations from the surge heights (referenced to NAVD88) for different 1-m sea level rise rate timesteps (i.e., output from SLAMM) to create depth grids (i.e., surge heights above ground), and then clipped habitat maps for all timesteps under the 1-m sea level rise rate scenario by these depth grids. I used tabular

data associated with these maps to calculate the amount of each CLC beach mouse habitat (Table 3-1) flooded by each hurricane category.

Results and Discussion

Changes in Beach Mouse Habitat over Time: Perdido Key and Santa Rosa Island

Under all sea level rise scenarios, the total proportion of beach mouse habitat on Perdido Key and Santa Rosa Island (i.e., area of beach mouse habitat divided by total area of each island) declines between 2007 and 2100. On Perdido Key, 6-15% will be lost, and 9-38% will be lost on Santa Rosa Island, depending on rate of sea level rise. In 2007, beach mouse habitat on Perdido Key comprises close to 53% of island area and declines to ~47% (0.7-m and 1-m rate of sea level rise) and ~38% (2-m rate) by 2100 (Figure 3-4). A greater proportion of Santa Rosa Island initially is classified as beach mouse habitat (~70%), but this proportion declines to ~61%, ~54%, and ~32% under the 0.7-, 1-m, and 2-m sea level rise rates (Figure 3-5).

On both islands, changes in the percentage of particular CLC habitat types as a percentage of island area are relatively small under the 0.7- and 1-m rates and are most noticeable under the 2-m sea level rise rate, beginning in 2075 (Figures 3-4 and 3-5). On Perdido Key, coastal grassland and scrub remain relatively constant (~13 to ~15%) or increase slightly between 2007 and 2100 under the 0.7-m and 1.0-m rates, but decrease under the 2-m rate from ~13% to 9% (Figure 3-4). On Santa Rosa Island, coastal grassland initially comprises a larger proportion of habitat than on Perdido Key (~30-35%) and also decreases under the 2 m rate (to ~18%) by 2100 (Figure 3-5). Coastal interdunal swale almost disappears by 2100, losing 93% of its area on Perdido Key (Figure 3-4 and Figure 3-6, top) and 94% of its area on Santa Rosa Island (Figure 3-5 and Figure 3-6, bottom) under a 2-m rate. On both islands, the percentage of beach dune decreases under all sea level rise rates between 2007 and 2100. On Perdido Key 24% of island habitat is beach dune in 2007, and this decreases by 5% (0.7-m rate),

7% (1-m rate), and 7% (2-m rate) by 2100. On Santa Rosa Island, starting at 22%, beach dune decreases by 6% (0.7-m rate), 7% (1-m rate), and 12% (2-m rate) by 2100 (Figures 3-4 and 3-5).

Patterns of habitat loss are similar when comparing changes in habitat as a percentage of island area (discussed in previous paragraph and shown in Figures 3-4 and 3-5) with changes in the total area of each habitat between 2007 and 2100 (Figure 3-6). Losses are similar under or increase slightly between the 0.7- and 1.0-m rate and are largest under the 2-m rate (Figure 3-6).

Rate of habitat loss across SLAMM model timesteps is faster on Santa Rosa Island than on Perdido Key, especially under the 2-m sea level rise scenario (Figure 3-7). Although the number of ha lost is relatively small for some habitats, particularly under the 0.7- and 1-m sea level rise rates, even a small change in area can account for a large percentage loss of that habitat. Loss of habitats that are not widespread but are important to beach mice may have serious consequences. For example, interdunal swale declines by >45% on Perdido Key and Santa Rosa Island under all scenarios, and almost disappears under the 2-m rate (Figure 3-6). During wet periods only the edges of these areas may be used for foraging or as burrow sites (Branch et al. 2011) and the rest is covered by water, whereas during dry periods, beach mice may use the entire area. Swale edges may be particularly important when burrow site options are limited for some individuals, e.g., where dunes have been flattened by storms. Similarly, scrub is not widespread, but may be important as burrow habitat (see next section) and serves as refugia during storms (Pries et al. 2009). Scrub remains relatively constant on Perdido Key and Santa Rosa Island, with \leq 26 ha ($<13\%$) lost under both the 0.7- and 1.0- m sea level rise rate, but this loss quadruples under the 2-m sea level rise rate (Figure 3-6).

In addition to potentially functioning as burrowing and foraging habitat, habitat types that are more widespread such as coastal grassland (CLC category) or beach dune (see Figures 3-8 to

3-13), provide the matrix through which mice move for dispersal and foraging. Loss of contiguous areas of this habitat due to sea level rise can lead to physical separation of subpopulations of beach mice. Such a gap (~140 m in width) is predicted to develop on the eastern end of Perdido Key (Figures 3-11 to 3-13). By 2100 (under the 2-m rate), this gap will be ~450 m across, cutting off the eastern tip of the island which is believed to have functioned as refugia for mice during catastrophic hurricanes like Opal (1995) and Ivan (2004). Similarly, on the western end of Santa Rosa Island in the Fort Pickens section of Gulf Islands National Seashore (Figure 3-3B), three low-elevation, island-wide sections of land ~550~1600 m in length will be lost to sea level rise (2-m rate, unpubl. data), isolating the Fort Pickens site from the rest of Santa Rosa Island. Like the eastern tip of Perdido Key, this part of Santa Rosa Island is wide (~1 km across) and may be refugia for Santa Rosa beach mice during large storms. Subpopulations of beach mice isolated by these gaps will be susceptible to extirpation from stochastic events like hurricanes, similar to what happened to Perdido Key beach mice in Gulf State Park, which were extirpated following Hurricane Opal in 1995. Permanent gaps will prevent dispersal from contiguous areas that can lead to demographic and genetic rescue of small populations (Oli et al. 2001, Hufbauer et al. 2015).

Habitat Suitability Model and Changes in Burrow Occurrence Probabilities: Perdido Key and Santa Rosa Island

Habitat suitability model. Elevation and slope were both important in explaining the occurrence of beach mouse burrows. The fitted habitat suitability model statement was “logit(p) = -3.6187 + 1.0365*elevation + 0.2923*slope.” I used maximum likelihood estimation to estimate the model parameters and the Wald statistic to assess the significance of coefficients for slope ($\chi^2=10.94$, df=1, p=0.001) and elevation ($\chi^2=4.99$, df=1, p=0.03) at $\alpha=0.05$. The model performed well when using a cutoff value of 0.40 on a probability scale; this is the value at

which fitted values for the burrow data were considered correctly classified as present if >0.40 or not present if <0.40 . The percentage of the burrow data points correctly predicted by the model as present or not present was 84% (i.e., accuracy), with 86% of the points predicted to have a burrow correctly classified (sensitivity) and 82% of locations predicted to have no burrow (specificity) correctly classified. The Kappa statistic was 0.67, suggesting there was substantial agreement between model prediction and observed data (Landis and Koch 1977). The AUC for the ROC curve was 0.86, which also suggests the fit of the model to the data was good.

The model shows that as elevation and slope increase in an area, the probability of finding a beach mouse burrow increases. For example, varying only the parameter in question, the odds of finding a beach mouse burrow would increase by 182% (odds = 2.82, 95% CI: 1.13-7.00) with each meter increase in elevation, and by 34% (odds = 1.34; 95% CI: 1.13-1.59) with degree increase in slope. The importance of elevation and slope for beach mouse burrow habitat has been documented, as has burrowing on tall established dunes that have a stable substrate and provide cover, nesting habitat, and refugia from predators (Lynn 2000, Scheckenberger 2001). Because beach mice depend on these functions for survival, I define valuable habitat as areas with high elevation and slope. There are known exceptions to this definition that should be kept in mind; beach mice have been observed to occasionally burrow in low-elevation areas e.g., swale edges (Branch et al. 2011).

General changes in habitat quality categories. Perdido Key and Santa Rosa Island had similar proportions of the probability categories I defined (0-0.25, >0.25 -0.50, >0.50 -0.75, >0.75 -1.0) (Figures 3-14 to 3-16). The two highest categories represented the smallest percentage of island habitat in 2007 (3.6% to 7.6%) and these values decrease until 2100 to between <1% and 2.1%. The most noticeable decrease in percentage of habitat occurs in the

category with >0.25-0.50 probability of burrow presence, decreasing from between ~17% and 20% in 2007 to between 4% and 12% in 2100, depending on island and rate of sea level rise. On both islands, areas with high probability of burrows (>0.50-0.75 and 0.75-1.0) lose less habitat (in ha) between 2007 and 2100 than areas with >0.25-0.50 probability of burrows. Also, area of habitat lost generally increases with rate of sea level rise (Figure 3-17). Percentages of habitat lost across burrow probability categories are very similar, especially for Perdido Key, and values are high. For example, the predicted loss of 22 ha (Perdido Key, 0.7-m rate, >0.75-1.0 category) equates to halving the amount of best habitat available to mice. Under a 2-m rate, a loss of 36 ha translates to an 83% reduction of this habitat.

Habitat in the lowest probability category (≤ 0.25) comprises the majority of beach mouse habitat remaining on Perdido Key and Santa Rosa Island over the years modeled (between ~30%- and 50%, Figure 3-16) relative to that in other probability categories where the percentage of habitat ranges between <1% and 20% (Figures 3-14 and 3-15). Areas in these categories may be overestimated because data for the digital elevation model I used in the SLAMM model were collected in 2006, shortly after an especially active hurricane season in 2004 and 2005 during which many dunes were flattened and vegetation was denuded from parts of Perdido Key and Santa Rosa Island. Therefore, some areas that appear to have low habitat quality based on this model (i.e., with <25% probability of containing a beach mouse burrow) likely have improved in habitat quality because dunes and vegetation have recovered (i.e., grown in elevation) in the last decade (Houser et al. 2015). The SLAMM model does not account for this type of land accumulation over time (Clough 2012). On the other hand, climate change studies predict that the frequency and intensity of severe weather events will increase (Bender et

al. 2010). Thus, hurricane seasons similar to the ones observed in 2004 and 2005 likely will reoccur and could reset conditions to a baseline similar to the one I modeled.

High to intermediate quality burrow habitat (>0.75-1.0 and >0.50-0.75 probability of beach mouse burrows). The best burrow habitat for beach mice overlapped in several areas with scrub habitat (Figures 3-8 to 3-13), likely because these areas are higher in elevation and steeper in slope than other habitat types. This pattern is clear on Santa Rosa Island (Figure 3-18) where scrub is less abundant than patches of other habitat types and is generally sparsely distributed (Miller et al. 2016), but the pattern is not as consistent on Perdido Key. For example, in Perdido Key State Park many patches of high quality habitat (especially south of the road) are categorized as scrub, but so are areas of low quality habitat on the north side of the road (Figures 3-2B and 3-19). Also, on the eastern half of Perdido Key, three large areas of scrub on the sound side are categorized as having poor quality burrow habitat (Figures 3-8 to 3-13). Although many areas of scrub are associated with high elevation and slope, scrub (or habitat with some woody vegetation) can occur in lower elevation areas as well, and these low areas would be associated with low burrow probability in my habitat suitability model.

Generally, scrub in coastal areas comprises upland habitat—particularly sandy ridges parallel to coastlines—with open or dense shrubs (e.g., Florida rosemary) and shrubby oaks (FNAI 2010). On Santa Rosa Island and Perdido Key, some areas classified in the CLC dataset as scrub, however, are actually remnants of large frontal or secondary dunes (i.e., seaward of sandy ridges) stable enough to have survived storms that regularly have impacted the Florida Gulf Coast, and contain woody and mixed-herbaceous vegetation (D. Miller, pers. comm.). The CLC designation of scrub appears to be influenced more by presence of woody vegetation than by location, and can include large contiguous patches along the sound side, e.g., those found in

the eastern part of Perdido Key and within the Eglin Air Force Base section of Santa Rosa Island; remnant patches closer to the middle of the island, e.g., those found in the northern part of Perdido Key State Park and on the Eglin Air Force Base section of Santa Rosa Island; and remnant frontal dunes with woody vegetation closer to the Gulf of Mexico, e.g., some patches south of the road in Perdido Key State Park and many areas in Eglin Air Force Base on Santa Rosa Island (Figures 3-18 to 3-20)(Miller et al. 2016).

Some areas not classified as high quality in this analysis may be critical beach mouse habitat for reasons other than providing burrow habitat. For example, the three large patches of scrub on eastern Perdido Key (Figure 3-8, top) were not identified as high quality habitat because they are low in elevation and slope, but at least one of these sites is known to have functioned as refugia for Perdido Key beach mice during and after Hurricane Ivan (2004). Mice may use this area only to forage but build burrows in better higher elevation habitat nearby, or mice may build burrows in low-elevation sites such as in these patches. If the latter explanation is correct, this suggests the habitat suitability model may not be as effective for predicting burrow occurrence on Perdido Key as it is for Santa Rosa Island, and the model should be validated for Perdido Key.

Some individuals also survived in the area west of the Fort McCree area (Figure 3-2C) which has an intermediate burrow probability (>0.50 to 0.75) and where no habitat is classified as scrub. Being on the sound side of the island and covered by vegetation (including woody vegetation), these sites—particularly west of Fort McCree—likely were more protected from the brunt of the hurricane, which struck the island from the south. Survivors in these areas are believed to have repopulated Gulf Islands National Seashore, possibly preventing the subpopulation of Perdido Key beach mice on Gulf Islands National Seashore from extirpation

(D. Greene, pers. comm.). Many of these sites persist until 2100 under the 1-m sea level rise rate (Figures 3-8 to 3-13, bottom), so may serve as refugia during future storms.

In addition to dunes with woody vegetation, several roadside areas on Perdido Key and Santa Rosa Island have intermediate to high quality patches of habitat. Roads on Eglin Air Force Base and those passing through public areas like Gulf Islands National Seashore and Perdido Key State Park are cleared of accumulating sand as necessary to allow traffic to pass. Mixed-herbaceous vegetation that become established on sand captured by sand fences and sand cleared from roads facilitates the growth of new dunes (Miller et al. 2001). As a result of sand accumulation and subsequent colonization of vegetation, linear bands of dunes develop along roadsides, such as those in along Perdido Key State Park (Figure 3-19), the Johnson Beach section of Gulf Islands National Seashore on Perdido Key (Figures 3-2C and 3-20), and the Santa Rosa area of Gulf Islands National Seashore on Santa Rosa Island (Figure 3-18). These bands form good burrow habitat because these areas are high in elevation and slope. Areas of lower-intermediate quality habitat (>0.25-0.50) often surround or help form these linear bands, increasing the amount of potential burrow habitat. By 2100, probabilities associated with habitat along some roads decrease, but are high relative to surrounding areas (Figure 3-20, bottom). Many linear berms or restored dunes in front of developed areas along the Gulf side of Perdido Key and Santa Rosa Island were classified as having intermediate-high quality habitat, and like roadside habitat may be used for burrowing and movement by mice.

Intermediate to low quality burrow habitat (>0.25-0.50 and 0-0.25 categories). Most potential habitat on Perdido Key and Santa Rosa Island was classified as coastal grassland or beach dune, and most of these areas were unlikely to contain burrows (i.e., 0-0.25 or >0.25-0.50 probability categories). Beach dunes in particular had lower burrow probabilities than did

coastal grasslands or scrub. This is likely related to the fact that these dunes, as categorized by CLC and based on FNAI descriptions, comprise an herbaceous coastal plant community on the beach and first dune above the beach whereas scrub is a community of evergreen shrubs on sandy ridges (FNAI 2010). The lower probability of burrows on beach dunes may be related to the fact that they are lower in elevation and slope than the areas classified as scrub. Larger dunes, including those near the Gulf in Perdido Key State Park (Figure 3-19) and on Santa Rosa Island (Figure 3-18), usually had some woody vegetation and, therefore were probably classified by CLC as scrub rather than beach dunes.

Although generally classified in this analysis as having low burrow probabilities, coastal grassland and beach dune clearly are important to mice as foraging habitat, matrix habitat for movement, and under some conditions, burrow habitat and refugia. Where there are scattered habitat patches with a high probability of burrows (e.g., some parts of Santa Rosa Island, Figure 18), habitats with a low probability of burrows are important for mice to move among burrow habitat. Where scrub vegetation is sparse or non-existent, beach mice likely use these areas for burrow habitat as well. On narrow and flat islands like Perdido Key, this habitat may be of even higher value than suggested by the habitat suitability model, and compared to the value of the same habitats on larger, wider islands like Santa Rosa Island simply because there are fewer large remnant dunes on Perdido Key than on Santa Rosa Island, and Perdido Key beach mice must use coastal grasslands and beach dunes by necessity. Also, as sea level rise progresses, these habitats will comprise the majority of remaining beach mouse habitat (Figures 3-4 and 3-5) and be the only habitats that remain unflooded following large storms (Figures 3-21 to 3-24 and E-1 to E-6), thereby serving as temporary refugia. Changes in individual habitat types by

hurricane category and SLAMM model step are summarized in Appendix F, Figures F-1 and F-2.

Effects of Hurricane Surge on Habitat: Perdido Key

Flooding from storms will at least temporarily reduce the amount of habitat available to Perdido Key beach mice, and the scope of this change and duration of lasting effects on habitat will depend on storm intensity and sea level rise scenario (Figures 3-21 to 3-24 and E-1 to E-6). Under a 1-m rate of sea level rise, area affected by flooding from surge generally increases over time and with hurricane intensity. For most Category 2 and 3 hurricanes, area affected decreases after 2050, however under all intensities and across all sea level rise model time steps, the percentage of available beach mouse habitat affected increased. Over half of all Perdido Key beach mouse habitat will be flooded under storms > Category 1, and >86% of habitat available to beach mice will be affected under all Category 3 storms in all years (Figure 3-24). Following a Category 1 storm, habitat on some parts of Perdido Key will remain relatively connected (Figures 3-21, E-1, and E-4), although the island is breached in a low-lying area near the eastern end. Under all storm intensities, flooding of this gap will be temporary through at least 2075 (Figure 3-11), but by 2100 inundation from sea level rise will separate the eastern tip of Perdido Key from the rest of the island (Figure 3-12). Surge from a Category 3 storm is predicted to flood most available habitat (Figures 3-23 and E-6), although in Perdido Key State Park, some habitat will remain dry, even after a Category 3 storm in 2100 (Figures E-1 to E-3). Perdido Key beach mice were previously extirpated from Perdido Key State Park, but under the conditions I modeled, patches of habitat that presumably would provide some protection during storms would persist. The high relative elevation of this habitat likely mitigates the risk of storm damage associated with the narrow width (i.e., ~ 100-150 m) and close proximity to the Gulf.

Facilitating the establishment, growth, and anchoring of dunes in Perdido Key State Park to hedge against storm damage in the future should be a management priority in this area.

Areas identified as potential refugia (see previous section) will be severely compromised by storm surge. For example, in the northern peninsula west of the Fort McCree site (Figure 3-21) surge from a Category 1 storm in 2025 will begin to separate the habitat patches in which Perdido Key beach mice were able to survive the devastating effects of Hurricane Ivan in 2004. Inundation from sea level rise occurs between these patches by 2100 (Figure 3-21). A Category 3 hurricane will flood all habitat in this area (Figure 3-23). Some Perdido Key beach mice were believed to weather Hurricane Ivan as well in the peninsulas designated as scrub along the sound (Figures 3-8 to 3-13, middle). All or nearly all of this habitat will be affected by flooding from any storm (Figures 3-21 to 3-23) and flood depths will be relatively high, suggesting mice that use this area likely would not survive a Category 2 or 3 storm.

The duration of flooding should be considered when interpreting potential effects of storms on beach mouse habitat. Duration will vary by surge depth, island topography, and the presence of structures that prevent flood waters from receding. In most cases, those areas affected by shallow flooding (lighter shades of blue on surge maps), will recover relatively quickly and provide habitat through which mice can move. High surge (darker shades of blue on surge maps) will take longer than shallow flooding to recede, and could result in flattening of dunes, flooding beach mouse burrows, and killing vegetation on which mice depend for cover and food.

Limitations of SLAMM and SLOSH

Like any effort involving prediction of future conditions, assumptions and methodological limitations for making precise predictions should be kept in mind. Although the SLAMM model is better than some approaches that do not model processes associated with sea

level rise (i.e., inundation, erosion, overwash, and saturation), it does not account for important upland processes such as growth of dunes over time and uses only wetland and coarse coastal dry habitat categories (e.g., undeveloped dry land). Thus, changes in more finely divided upland habitats (e.g., the CLC categories coastal grassland and scrub which are part of the course SLAMM category undeveloped dry land) cannot be modeled individually. That is, I could not predict how these habitats might move or change to another habitat type with time. Because the assignment of SLAMM categories was originally based on CLC categories, in most areas the two datasets overlap. My use of the overlay of CLC categories on SLAMM results added some habitat detail that would not have been seen using the original SLAMM output. In my study area, the main dynamic change in upland habitat using only SLAMM results showed beach dune moving inland where undeveloped dry land used to be. There were few areas where this occurred.

At the same time, modeling changes in beach mouse habitat with sea level rise based on CLC has limitations. Changes in CLC habitat with sea level rise that I present are based on static habitat data collected between 1997 and 2010 with different rates of sea level rise imposed on these static maps using SLAMM. In the absence of complex models that predict changes in the geomorphology of the island (e.g., dune accretion or loss of dunes) and how this affects different habitats, transitions of initial CLC habitats to other habitats cannot be predicted into the future. A dynamic model with much finer resolution in upland habitat types and the potential to transition these habitats to new habitats would aid in understanding and predicting impacts of sea level rise on species that use uplands.

The SLOSH model I used to estimate flooding after taking into account sea level rise effects should also be interpreted cautiously. By modeling an amalgamation of storms at high

tide, my intention was to present the worst-case effects on beach mouse habitat for each hurricane category examined. These results present only a snapshot of the potential effects of one type of storm (per hurricane category) on habitat remaining after predicted sea level rise. In reality, the Northern Gulf Coast is often affected by multiple storms within a season and across years, so habitat may be more degraded due to residual or cumulative effects of past and subsequent storms than presented in my model. Models with mechanisms for incorporating periodic impacts of tropical storms and hurricanes on geomorphology and habitat, (Aiello-Lammens et al. 2011, Dai et al. 2015) and wave effects on surge (Chen et al. 2014) should be used if understanding effects of potential combinations of storms and resulting surge on habitat is a priority.

General Conclusions and Management Considerations

In this project, I identify how the amount and distribution of beach mouse habitat is likely to change under different rates of sea level rise and how remaining habitat is likely to be affected by storm surge. This information can be used to help understand broad patterns in habitat change for beach mice and the types of problems mice will face at the landscape level (e.g., loss of landscape connectivity).

Both high quality burrowing habitat, found primarily in dunes with relatively high elevation and woody vegetation, and habitat that may not support burrows but can be important for foraging and movement between burrow sites (e.g., lower elevation beach dunes) will be lost. As higher elevation habitat is expected to persist longer, and low elevation beach dunes are lost, prime burrowing habitat will become more isolated. Maintaining these patches as well as creating connectivity between these patches should be a management priority before this fragmentation of habitat isolates subpopulations of beach mice. As sea level rise progresses, this will be particularly important along narrow parts of Perdido Key and Santa Rosa Island where

storm impacts (e.g., reduction in dune height and the resulting increase in susceptibility to inundation) can be especially severe on beach mouse habitat.

Dunes that form along roads, such as those in Perdido Key State Park and Gulf Islands National Seashore, as well as dune habitat parallel to development along the Gulf (i.e., natural incipient dunes, manmade berms, or restored dunes) provide good burrowing habitat for mice based on the habitat suitability model I used, and many are predicted to persist under a 1-m rate of sea level rise and when subjected to surge from Category 1-3 hurricanes. Given their location, risk of disturbance of these dunes by people is higher than in more isolated parts of Perdido Key and Santa Rosa Island. Physical barriers, structures that facilitate growth of dunes, dune restoration, enforcement of ordinances that protect dunes, and public education are all strategies that should be implemented to ensure the persistence of these areas.

Finally, given the frequency of tropical storms is predicted to increase throughout the 21st century (Bender et al. 2010) and associated surge will worsen (Grinsted et al. 2013), managers should consider preemptive actions that ensure access by mice to potential refugia during storms is maintained. A first step is to assess the feasibility of dune restoration or creation in sites especially at risk of overwash and eventual inundation. For example, the eastern tip of Perdido Key, believed to be refugia for Perdido Key beach mice, will eventually be separated from the rest of the island. Restoration that facilitates growth of stable and well-anchored dunes in the overwash area may prevent or at least delay isolation of this important habitat by inundation and flooding from storms. At the same time, ameliorating habitat within the refugia site itself—in this case along the peninsula west of Fort McCree—could prolong the function of this area as refugia for Perdido Key beach mice until 2100 and beyond. Balancing the habitat needs of species that use overwashed areas, e.g., shorebirds, with the needs of those that would benefit

from increased connectivity of dunes should be considered when deciding how to best manage such gaps.

Sea level rise is progressing and its rate may increase (IPCC 2014), the intensity of storms is predicted to rise (Bender et al. 2010), and development in some coastal areas continues unabated, despite the obvious evidence of climate change (Schrope 2010). Efforts to reduce habitat loss and fragmentation under these conditions is made even more complex in coastal systems because of the dynamic nature of these systems. Coasts shift, barrier islands migrate, and habitat loss from natural processes like overwash will continue to occur (Whittaker 1995, Lentz et al. 2013). Studies like this one are helpful for understanding general patterns of habitat change for different species. Linking habitat change models, population models (Oli et al. 2001, Falcy and Danielson 2014) and models that address dynamic changes in coastal morphology (Masetti et al. 2008, Lindemer et al. 2010, Dai et al. 2015) is an important next step in understanding extinction risk of coastal species, and how to best to manage coastal systems to hedge against biodiversity loss.

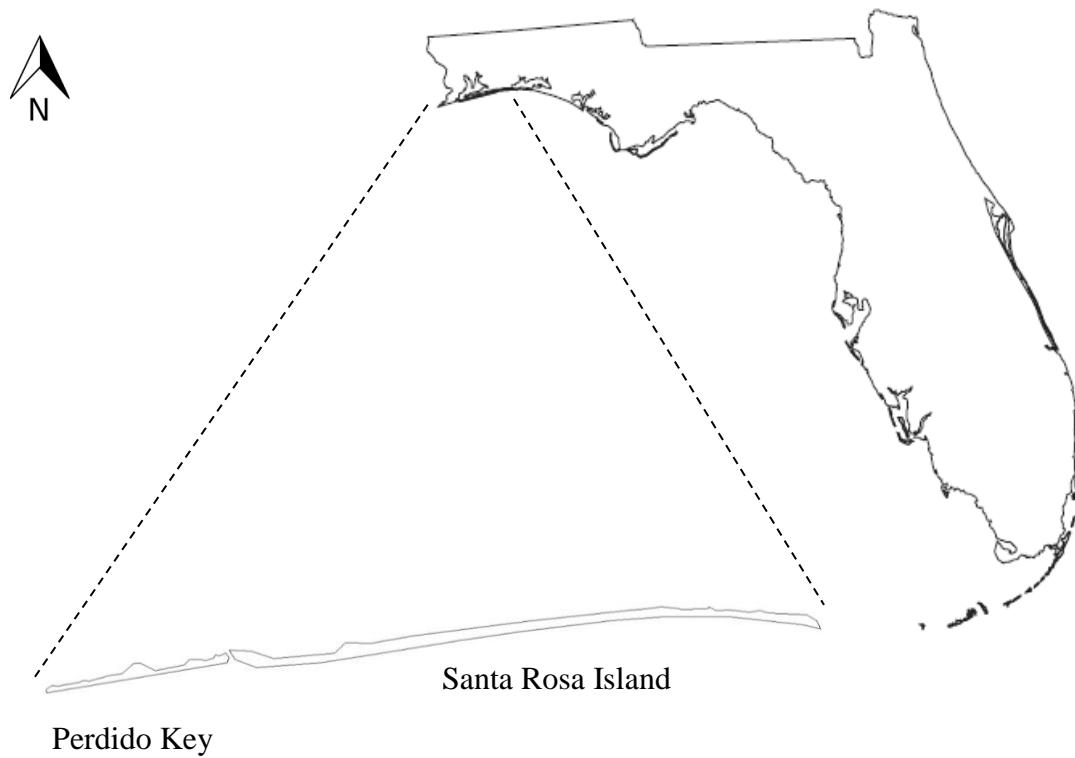


Figure 3-1. Diagram showing approximate location of Perdido Key and Santa Rosa Island relative to Florida. The westernmost three miles of Perdido Key are part of Alabama.



Figure 3-2. Map of Perdido Key (A.) (Google Maps 2017a) and photos showing Perdido Key State Park (B., ~3.3 km) (NOAA 2017c) and the Perdido Key Area of Gulf Islands National Seashore (C., ~10.7 km) (NOAA 2017a). Habitat distribution and probability model results for the area shown between the orange dashed lines in (C., ~5.3 km) are shown in Figures 3-8 through 3-13. Some results for the area in (B) and the area between the blue dotted lines in (C., ~3.4 km) are shown in Figures 3-19 and 3-20 and Appendix E. The yellow star shows the location of Fort McCree.

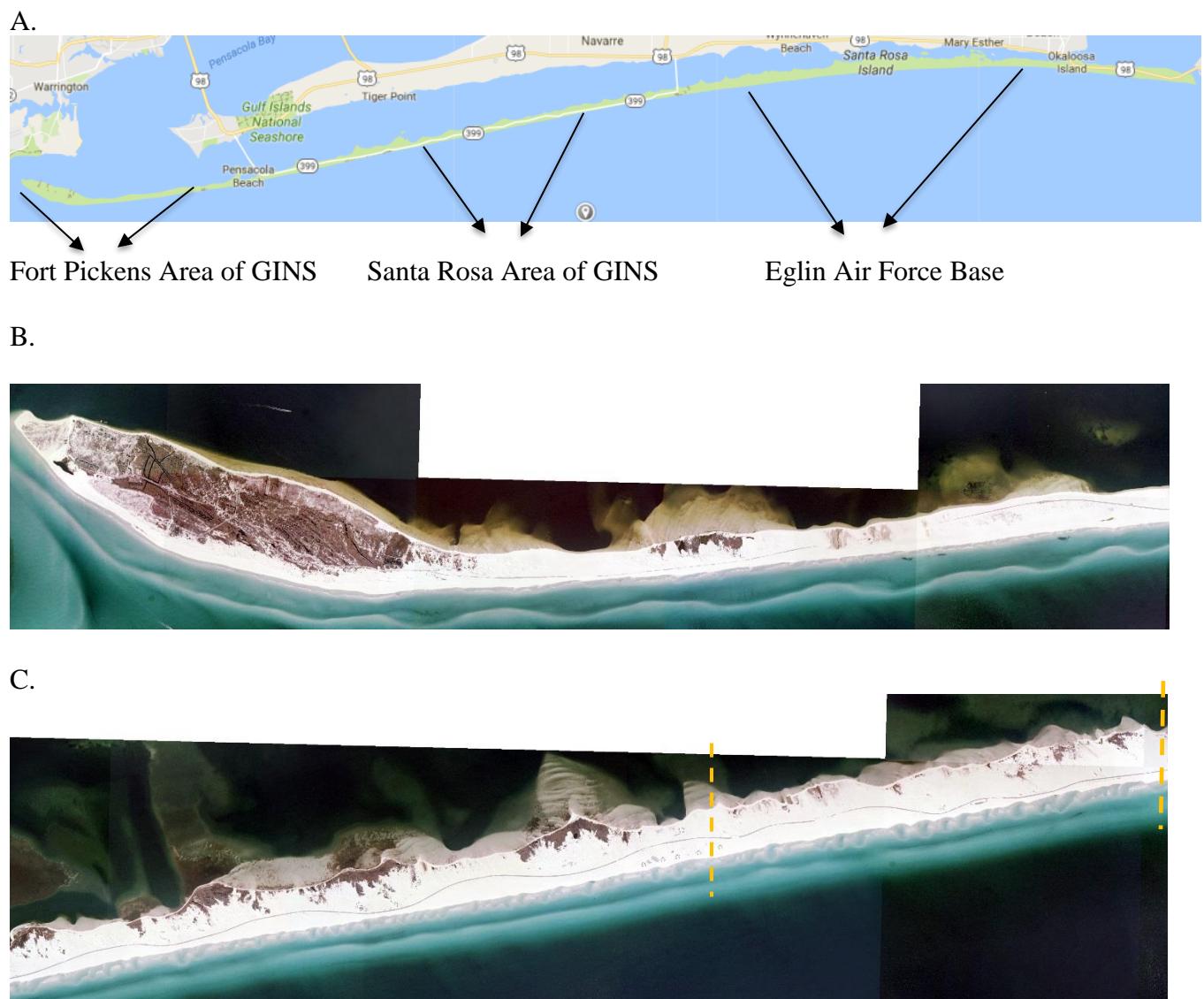


Figure 3-3. Map of Santa Rosa Island (A.) (Google Maps 2017b) and photos (NOAA 2017a) showing the Ft. Pickens (B.) and Santa Rosa (C.) areas of Gulf Islands National Seashore (GINS). The area between the orange dashed lines in (C.) is the approximate area shown in Figure 3-18.

Table 3-1. Land (or wetland) cover categories (from the sea level rise program, SLAMM) present on Perdido Key. Categories that I defined as including potential beach mouse habitat are italicized. The Cooperative Land Cover (CLC) categories present on Santa Rosa Island and Perdido Key that correspond to the SLAMM categories are shown. Only areas represented by categories I defined as beach mouse habitat are presented in all results (including those showing habitat probabilities), unless otherwise noted. “Beach dune” and “sand beach” values, and “coastal scrub” and “scrub” values were combined in results charts based on CLC data.

SLAMM category	CLC category
<i>Inland Freshwater Marsh</i>	<i>Coastal Interdunal Swale</i>
<i>Inland Freshwater Marsh</i>	Mixed Scrub Shrub Wetland
<i>Inland Freshwater Marsh</i>	Nonvegetated Wetland
<i>Inland Freshwater Marsh</i>	Wet Prairie
<i>Ocean Beach</i>	<i>Beach Dune</i>
<i>Ocean Beach</i>	<i>Sand Beach (Dry)</i>
<i>Undeveloped Dry Land</i>	<i>Coastal Grassland</i>
<i>Undeveloped Dry Land</i>	<i>Coastal Scrub</i>
<i>Undeveloped Dry Land</i>	<i>Scrub</i>
<i>Undeveloped Dry Land</i>	Maritime Hammock
<i>Undeveloped Dry Land</i>	Mesic Flatwoods
<i>Undeveloped Dry Land</i>	Mesic Hammock
<i>Undeveloped Dry Land</i>	Rural Open
<i>Undeveloped Dry Land</i>	Scrubby Flatwoods
<i>Undeveloped Dry Land</i>	Xeric Hammock

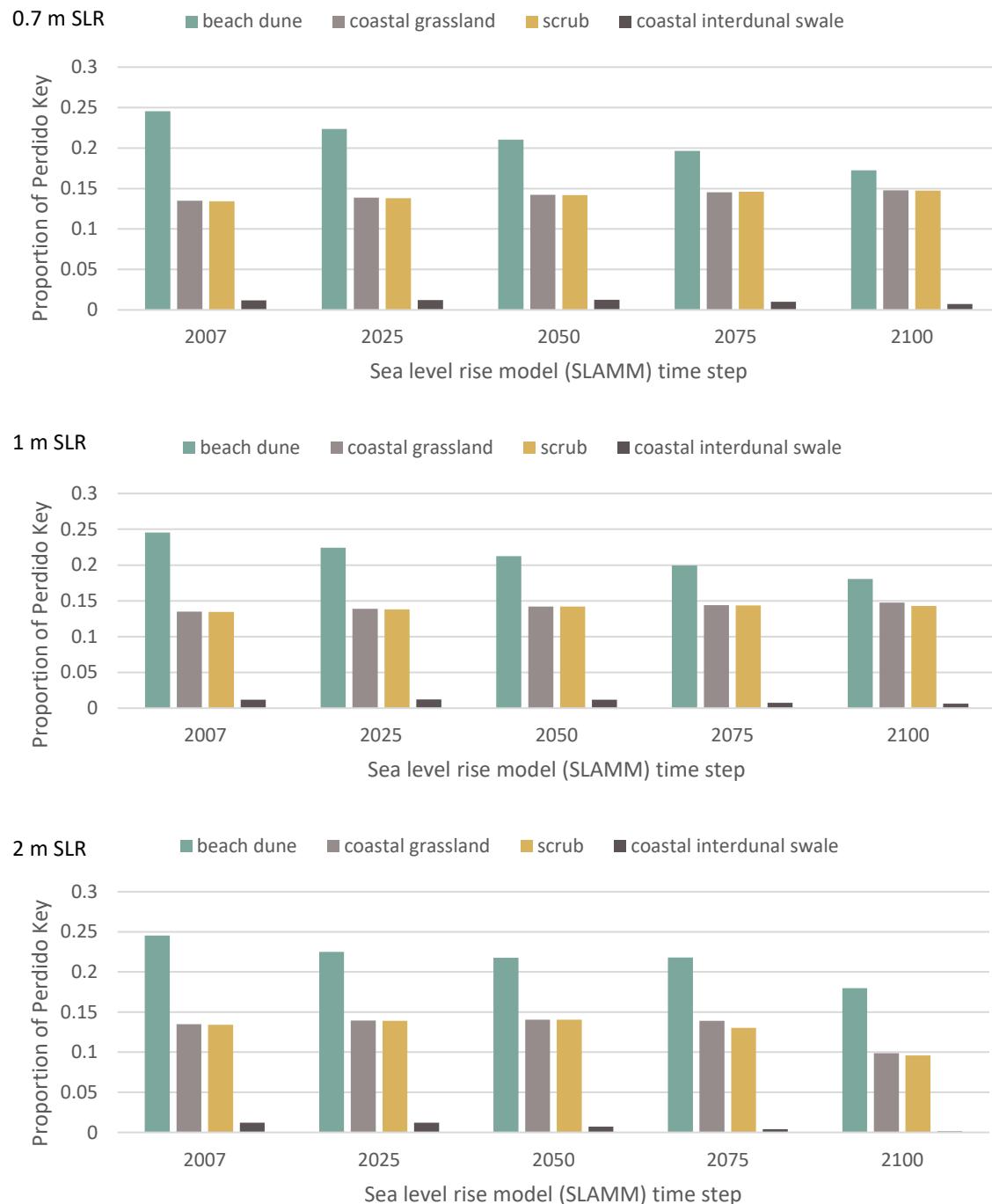
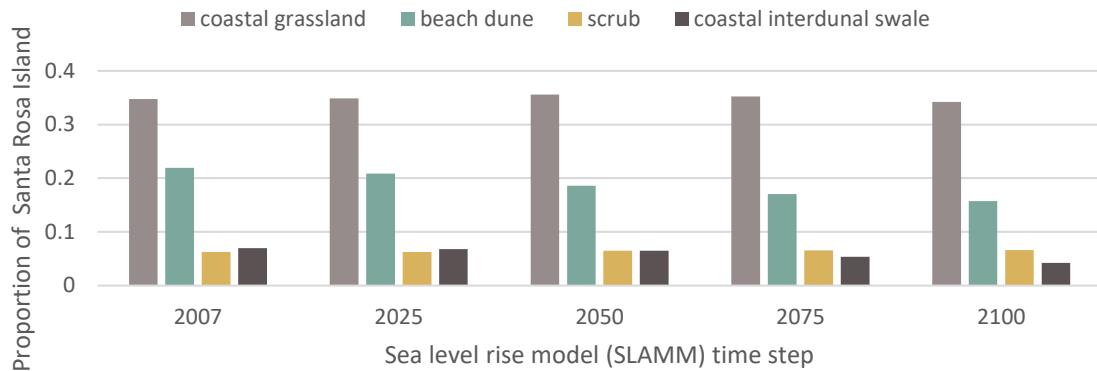


Figure 3-4. Proportion of beach mouse habitat on **Perdido Key** in different habitat categories after sea level rise at three different rates (top: 0.7 m, middle: 1 m, & bottom: 2 m, by 2100) and for different years. Results are based on Cooperative Land Cover (CLC) habitat categories (Table 3-1).

0.7 m SLR



1 m SLR



2 m SLR

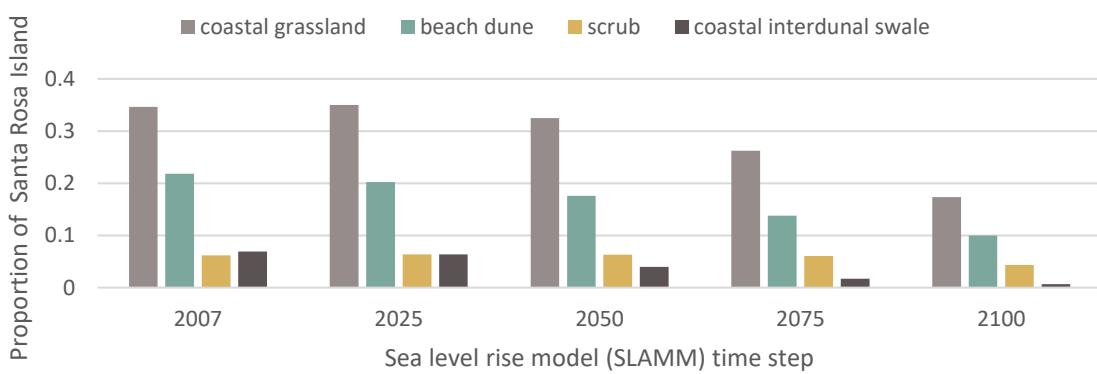


Figure 3-5. Proportion of beach mouse habitat on **Santa Rosa Island** in different habitat categories after sea level rise at three different rates (top: 0.7 m, middle: 1 m, & bottom: 2 m, by 2100) and for different years. Results are based on Cooperative Land Cover (CLC) habitat categories (Table 3-1).

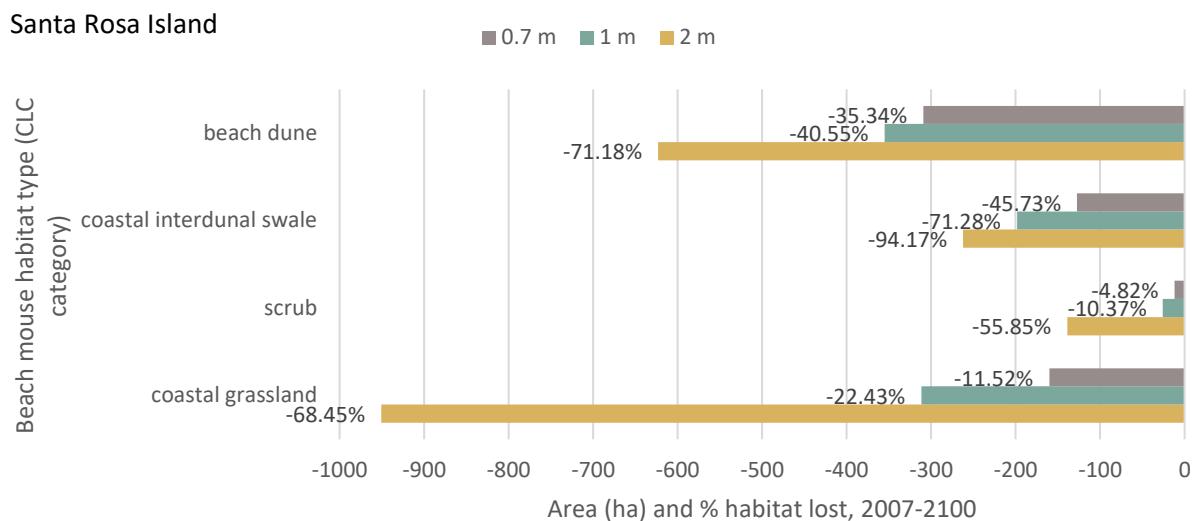
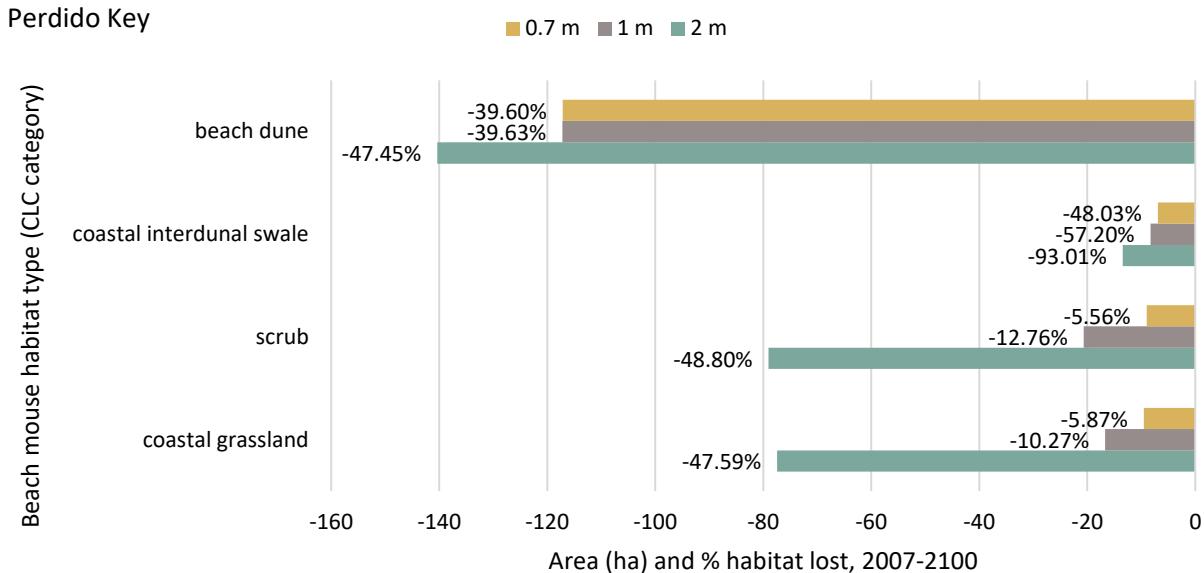


Figure 3-6. Area (ha) of beach mouse habitat on Perdido Key (top) Santa Rosa Island (bottom) in different habitat categories after sea level rise at three different rates (0.7 m, 1 m, & 2 m by 2100). Numbers (%) next to bars represent the percentage change in area for a habitat category between 2007 and 2100. Results are based on Cooperative Land Cover (CLC) habitat categories (Table 3-1).

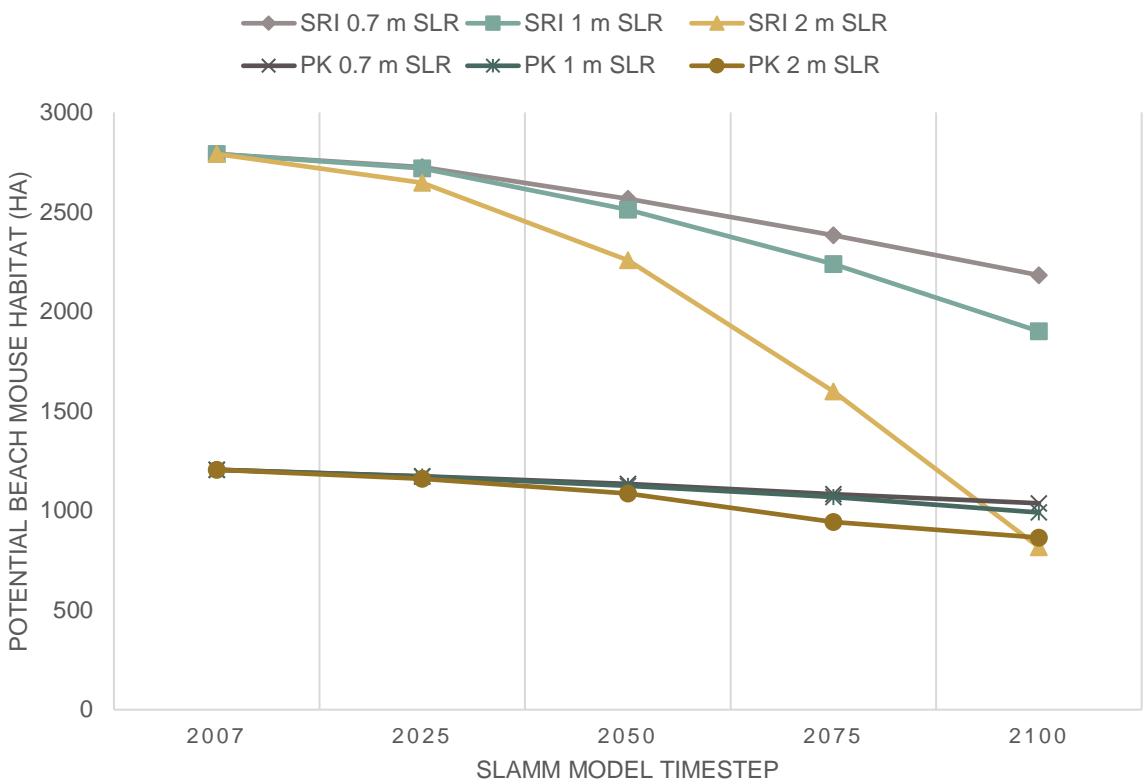


Figure 3-7. Change in area of beach mouse habitat on Perdido Key (PK) and Santa Rosa Island (SRI) under three rates of sea level rise (0.7-m, 1.0-m, and 2.0-m by 2100) and across timesteps of the SLAMM sea level rise model.

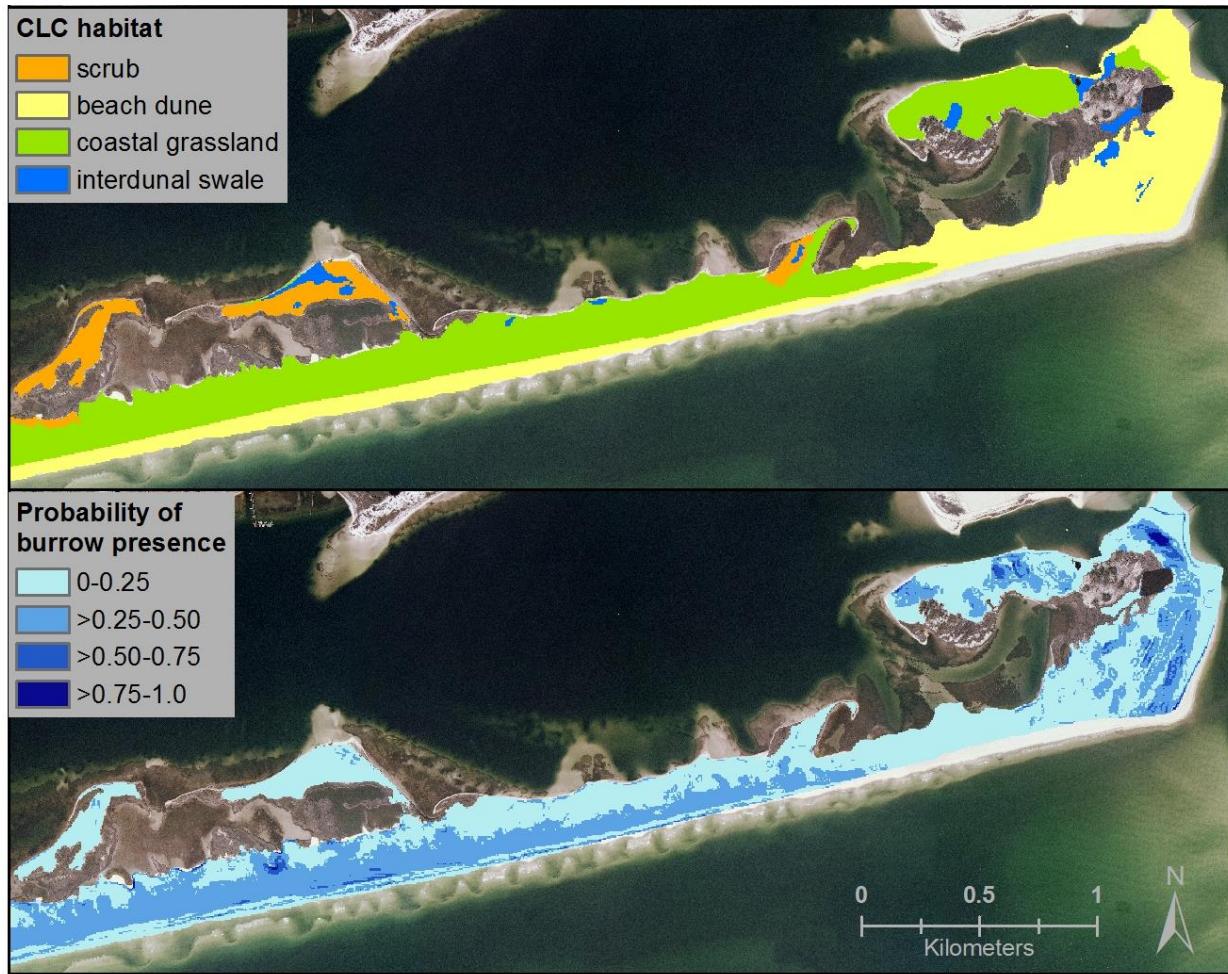


Figure 3-8. Beach mouse habitat represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C) in **2007**, the initial condition for my assessment of sea level rise using the SLAMM model. The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat, have changed to non-habitat, or have been lost to sea level rise (inundation).

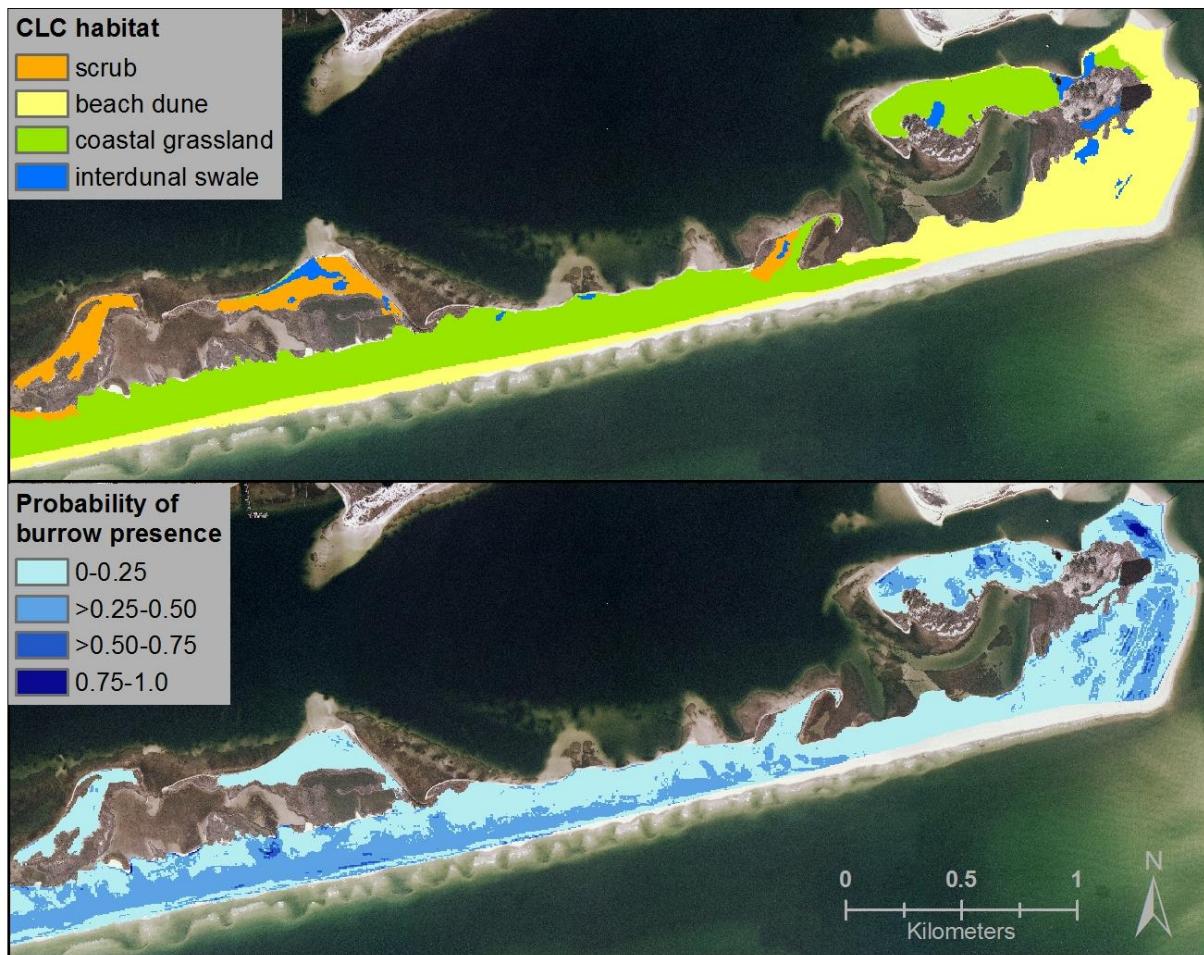


Figure 3-9. Beach mouse habitat remaining in **2025** after sea level rise at **rate of 1 m** (by 2100), represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat, have changed to non-habitat, or have been lost to sea level rise (inundation).

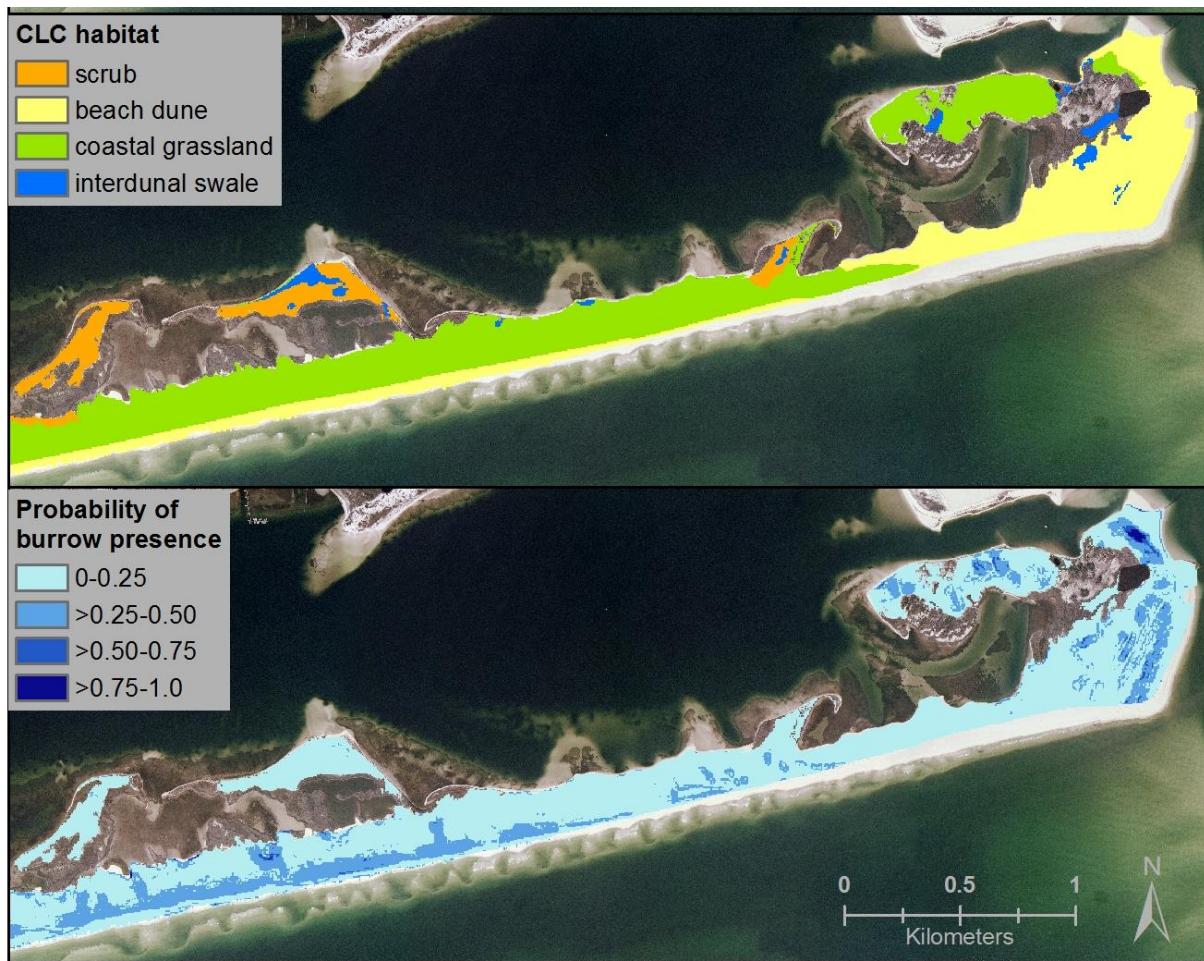


Figure 3-10. Beach mouse habitat remaining in **2050** after sea level rise at **rate of 1 m** (by 2100), represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-beach mouse habitat, or have been lost to sea level rise (inundation).

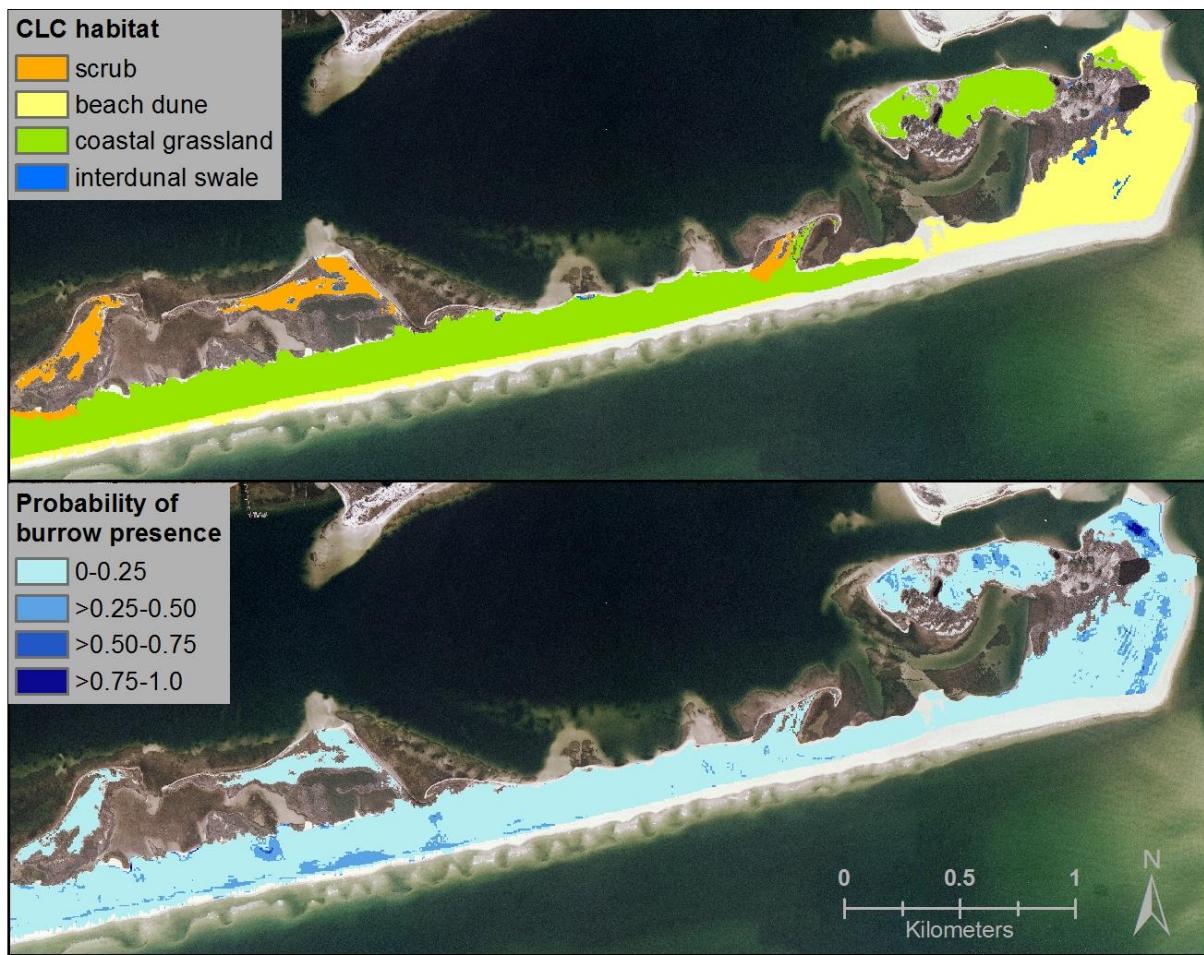


Figure 3-11. Beach mouse habitat remaining in **2075** after sea level rise at a **rate of 1 m** (by 2100), represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

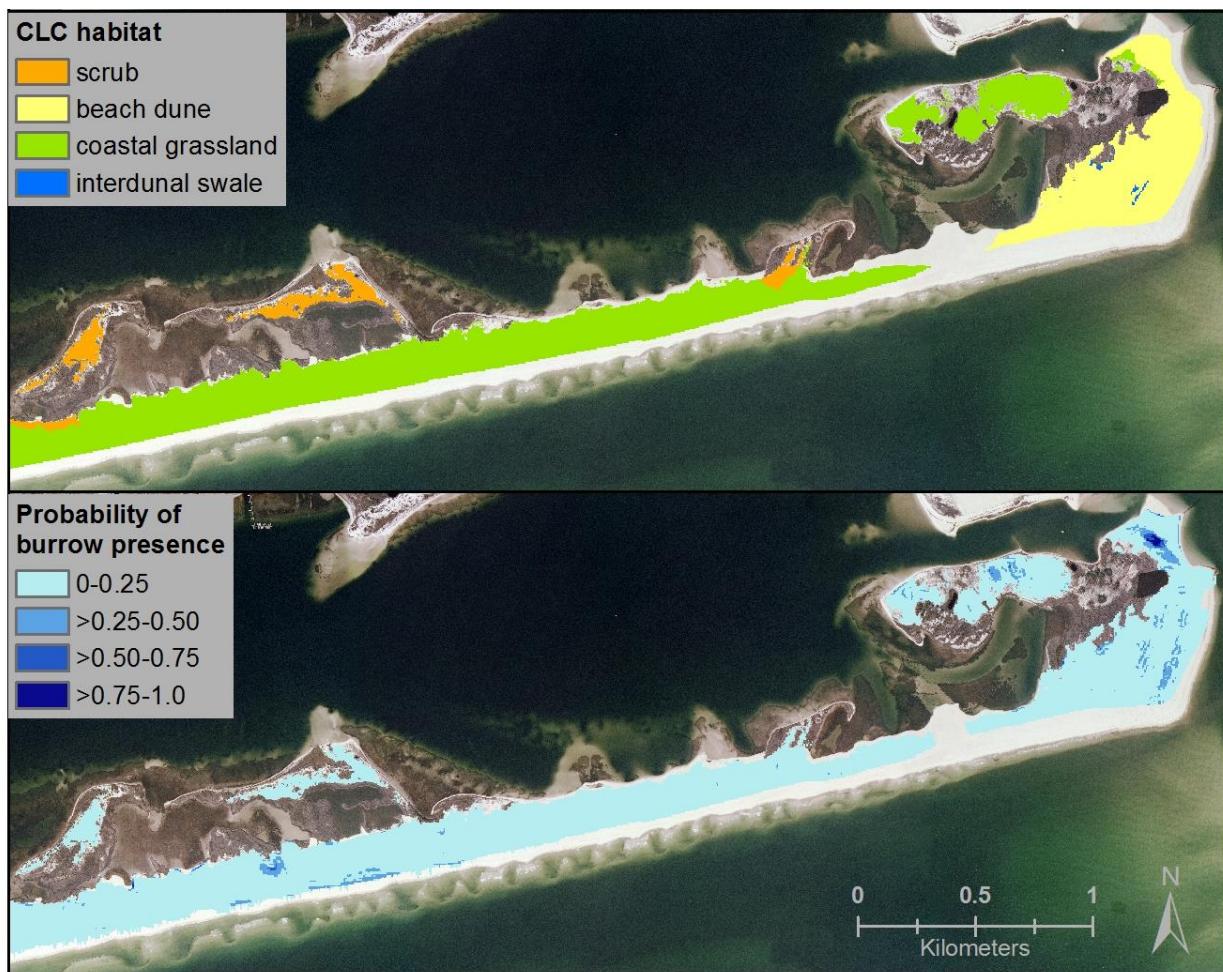


Figure 3-12. Beach mouse habitat remaining in **2100** after sea level rise at a **rate of 1 m** (by 2100), represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, see Figure 3-2C). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-beach mouse habitat, or have been lost to sea level rise (inundation).

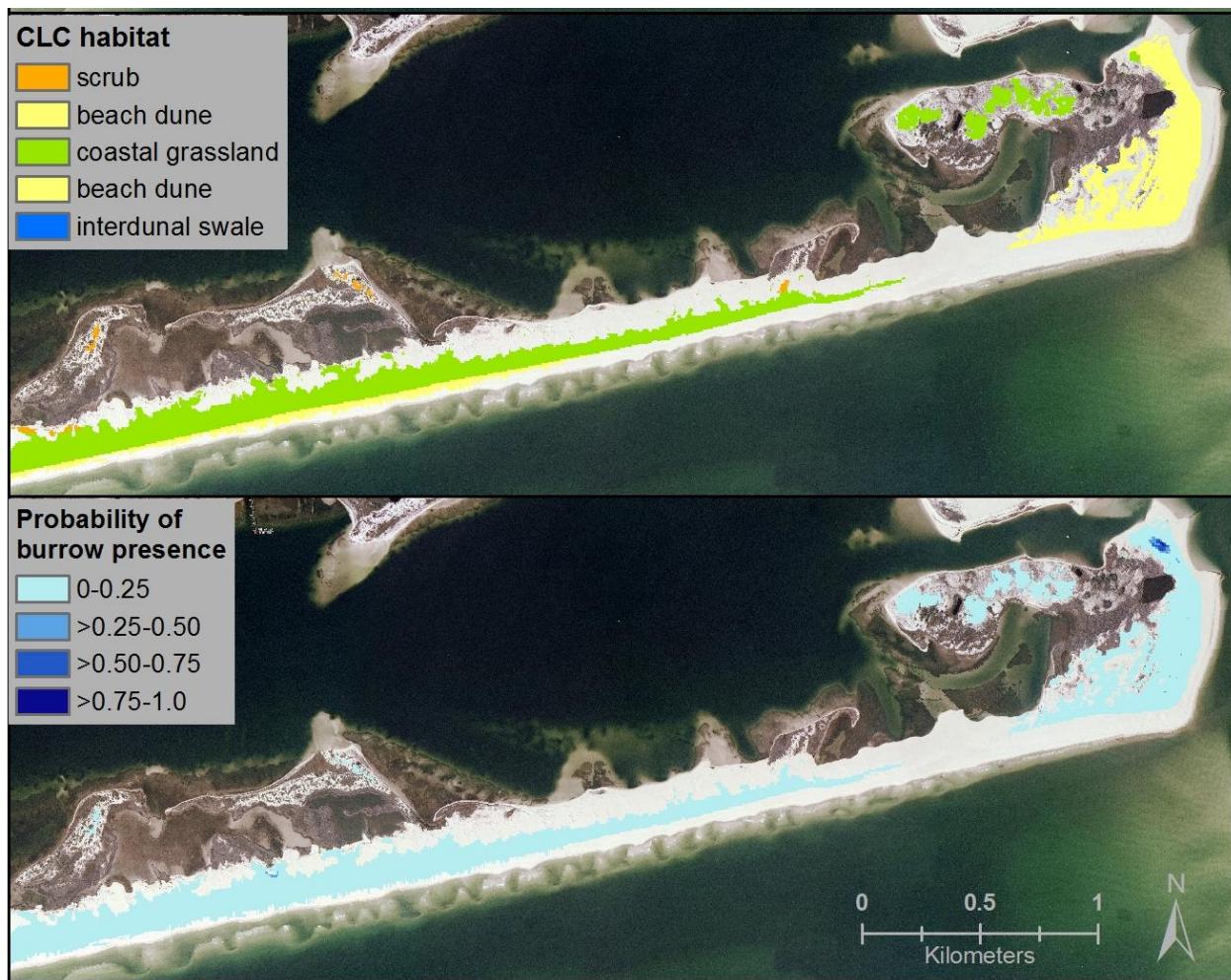


Figure 3-13. Beach mouse habitat remaining in **2100** after sea level rise at a **rate of 2 m** (by 2100), represented by CLC cover classes (top) (Table 3-1) for part of Gulf Islands National Seashore (~5.3 km, Figure 2C). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-beach mouse habitat, or have been lost to sea level rise (inundation).

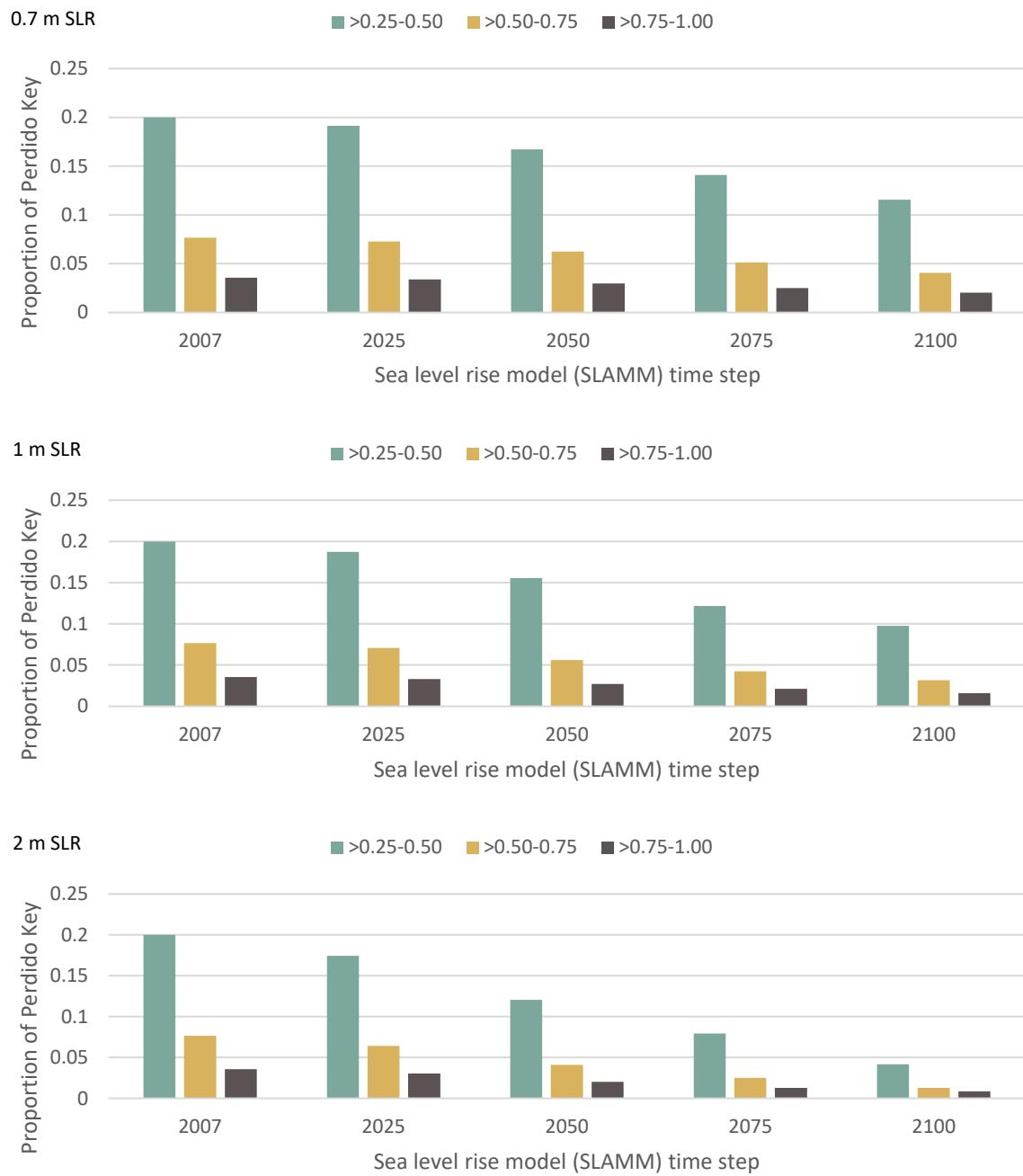


Figure 3-14. Proportion of beach mouse habitat on **Perdido Key** with a >25% probability of containing a beach mouse burrow. SLAMM model results from three sea level rise rates (top: 0.7 m, middle: 1 m, & bottom: 2 m, by 2100) are shown, based on CLC habitat categories (Table 3-1). Probabilities were determined by running a habitat suitability model for each cell of a spatial map covering the study area, using environmental parameter values specific to each cell for each model run.

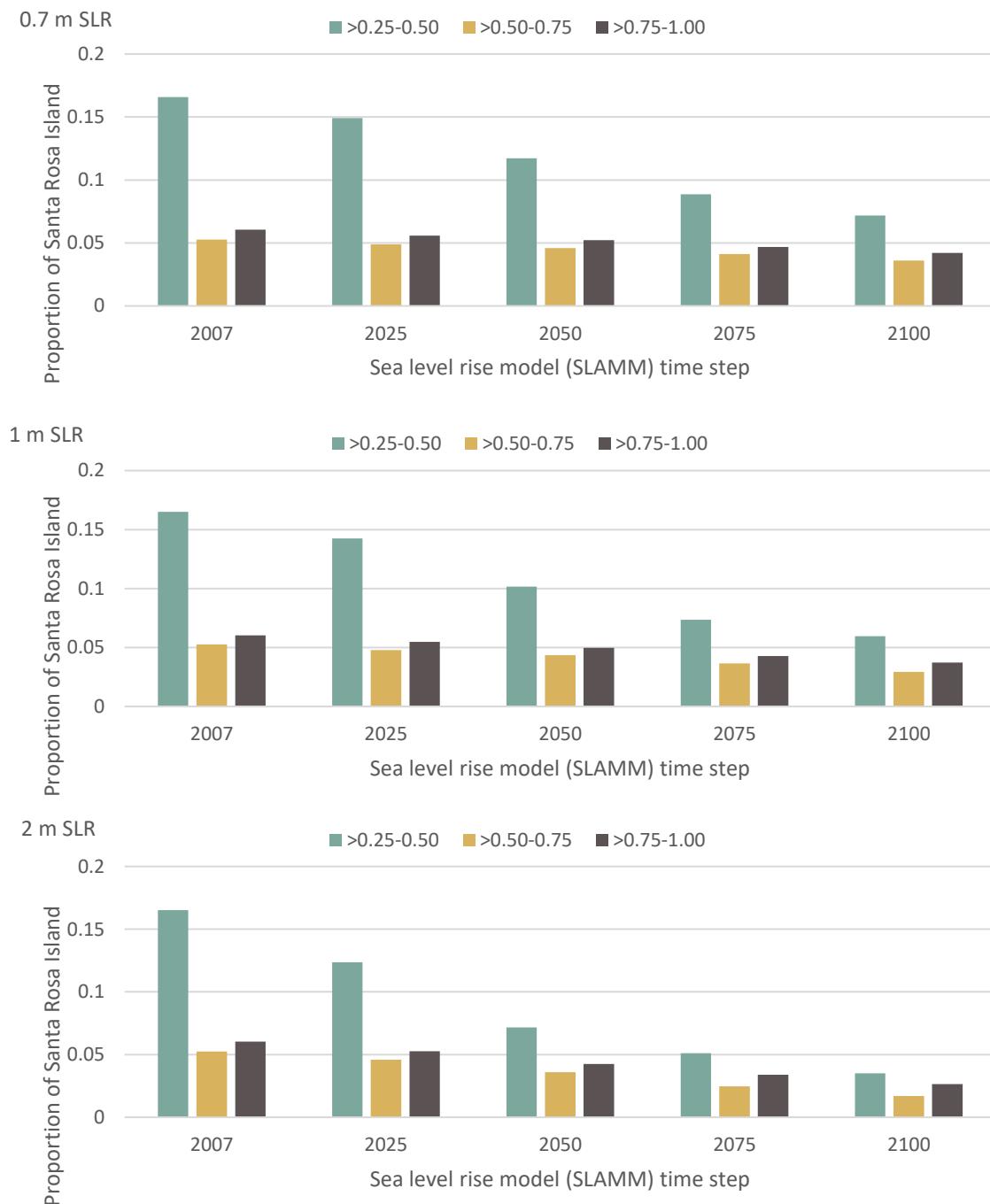


Figure 3-15. Proportion of beach mouse habitat on **Santa Rosa Island** with a >25% probability of containing a beach mouse burrow. SLAMM model results from three sea level rise rates (top: 0.7 m, middle: 1 m, & bottom: 2 m, by 2100) are shown, based on CLC habitat categories (Table 3-1). Probabilities were determined by running a habitat suitability model for each cell of a spatial map covering the study area, using environmental parameter values specific to each cell for each model run.

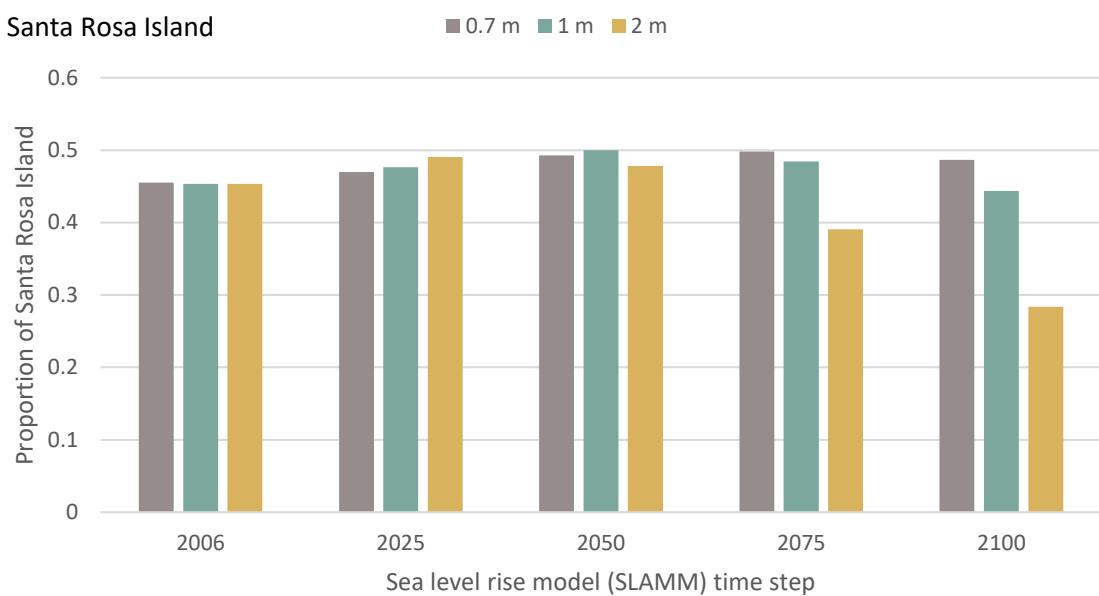
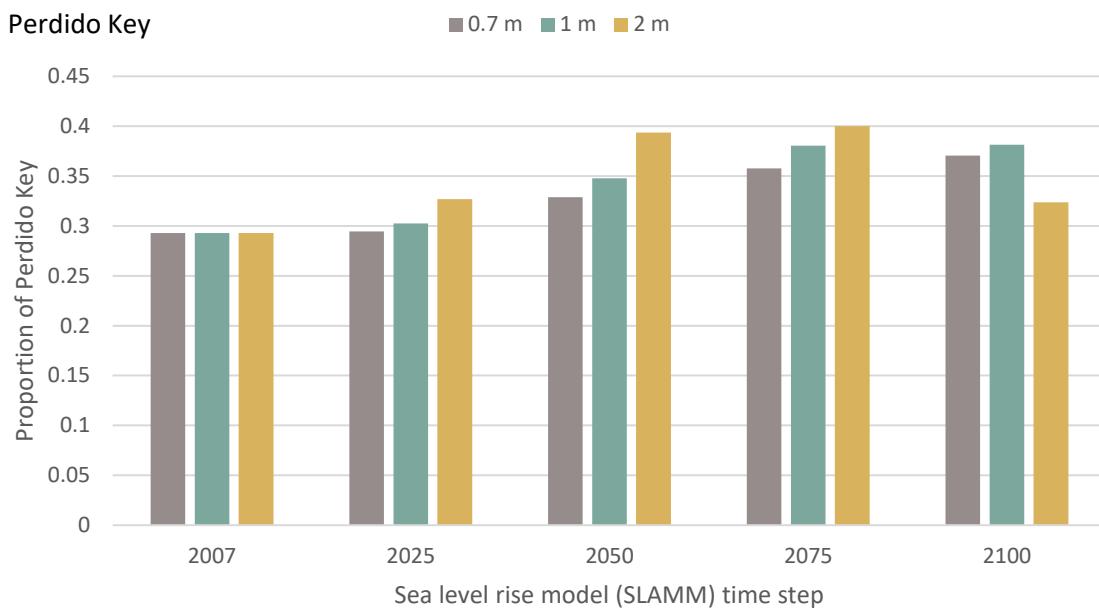


Figure 3-16. Proportion of beach mouse habitat on Perdido Key (top) and Santa Rosa Island (bottom) with a $\leq 25\%$ probability of containing a beach mouse burrow. Model results from three sea level rise rates (0.7 m, 1 m, & 2 m by 2100), based on default CLC model output. Probabilities were determined by running a habitat suitability model for each cell of a spatial map covering the study area, using environmental parameter values specific to each cell for each model run. Habitat categories represented in this chart are described under Table 3-1.

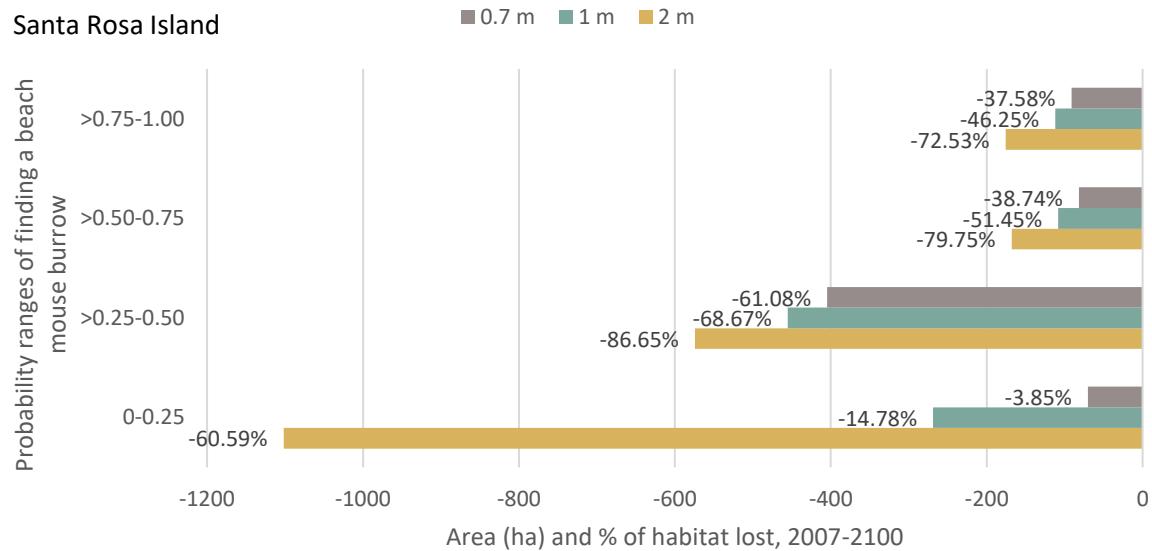
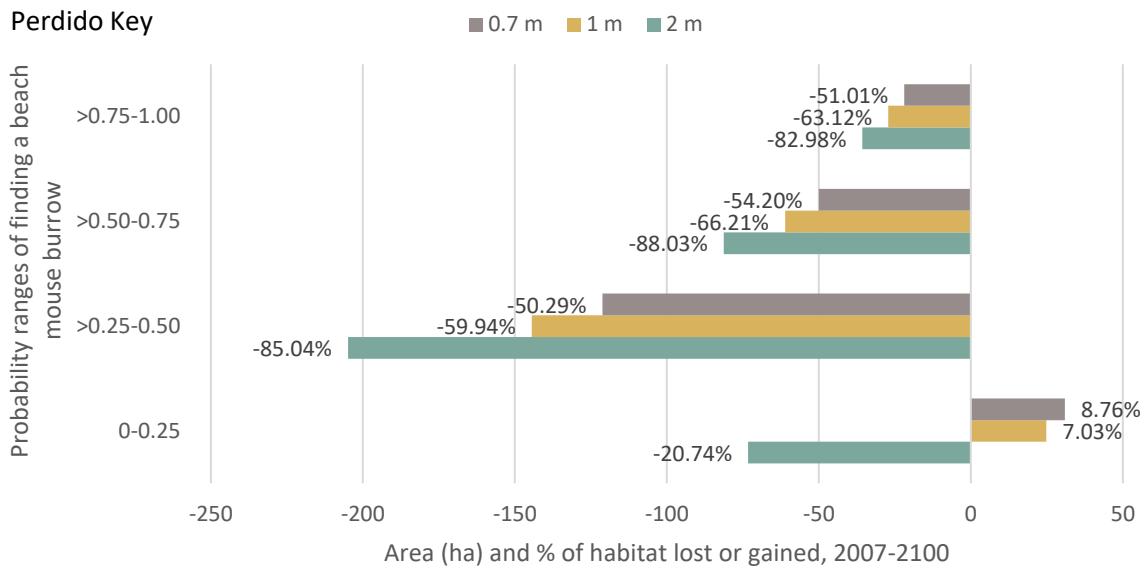


Figure 3-17. Area (ha) of beach mouse habitat on Perdido Key (top) and Santa Rosa Island (bottom) in different probability categories after sea level rise at three different rates (0.7 m, 1 m, & 2 m by 2100). Probability categories represent the probability of finding a beach mouse burrow in an area. Probabilities were determined by running a habitat suitability model for each cell of a spatial map covering the study area, and using environmental parameter values calculated for each cell in the model. Numbers (%) next to bars represent the percentage change in area for a probability category between 2007 and 2100. Results are based on Cooperative Land Cover (CLC) habitat categories (Table 3-1).

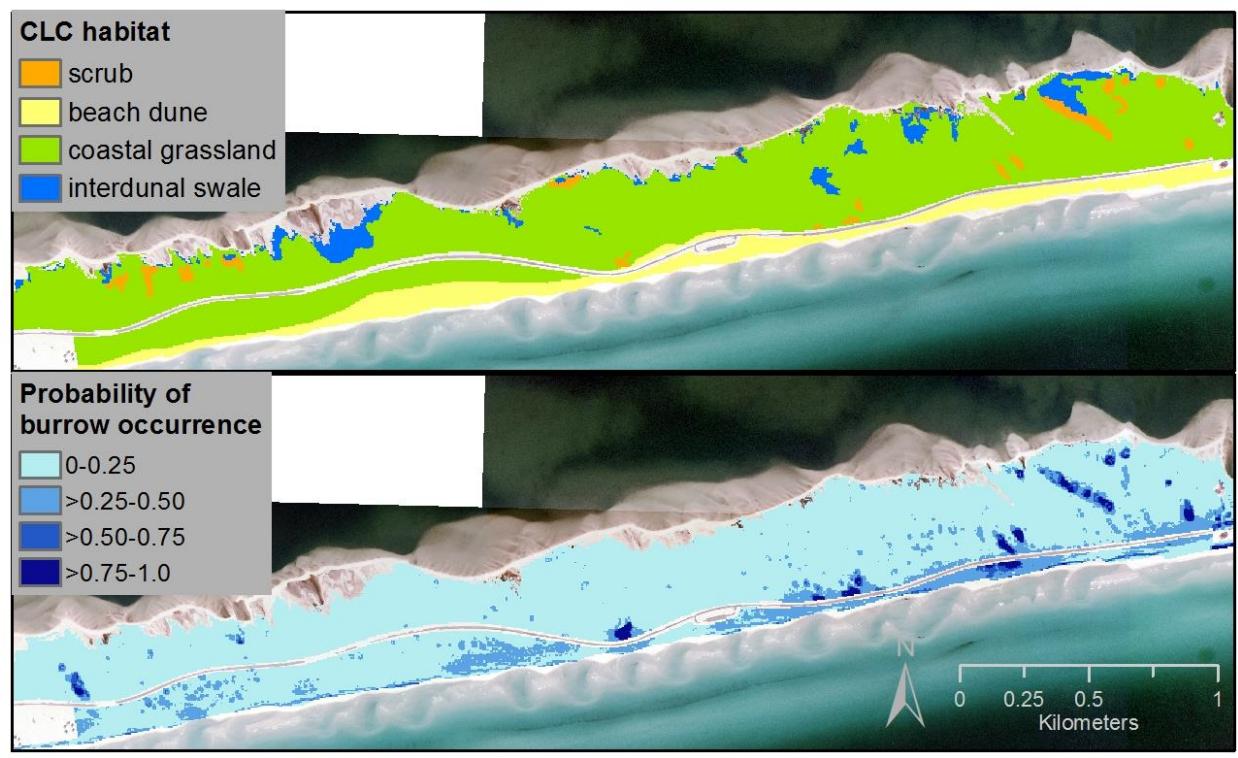


Figure 3-18. Beach mouse habitat remaining in **2025** after sea level rise at **rate of 1 m** (by 2100), represented by CLC (top) cover classes (Table 3-1) for part of Gulf Islands National Seashore (~4.5 km in length) on **Santa Rosa Island**. The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat, have changed to non-beach mouse habitat, or have been lost to sea level rise (inundation).

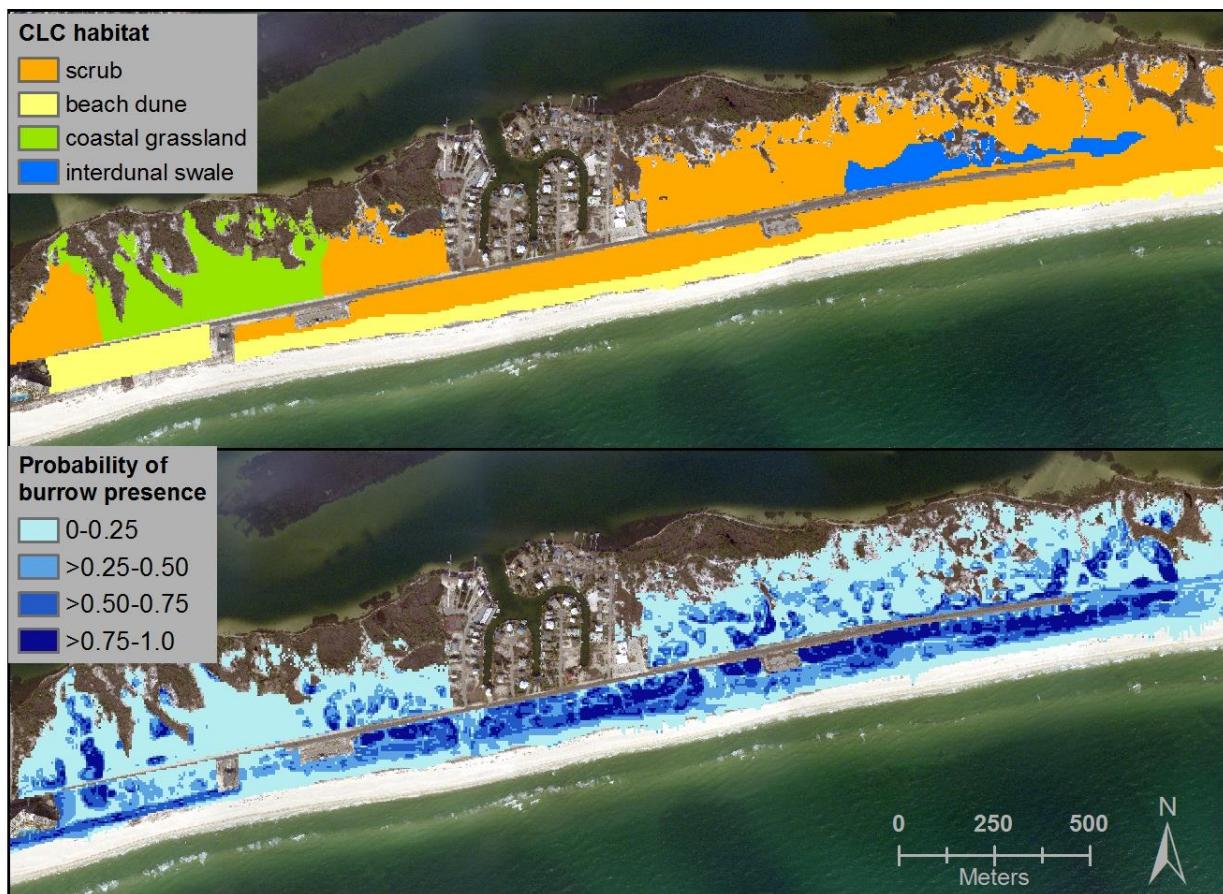


Figure 3-19. Beach mouse habitat remaining in **2100** after sea level rise at a **rate of 1 m** (by 2100), represented by CLC (top) cover classes (Table 3-1) in Perdido Key State Park (~3.3 km, Figure 3-2B). The bottom panel shows probabilities of finding a beach mouse burrow. Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

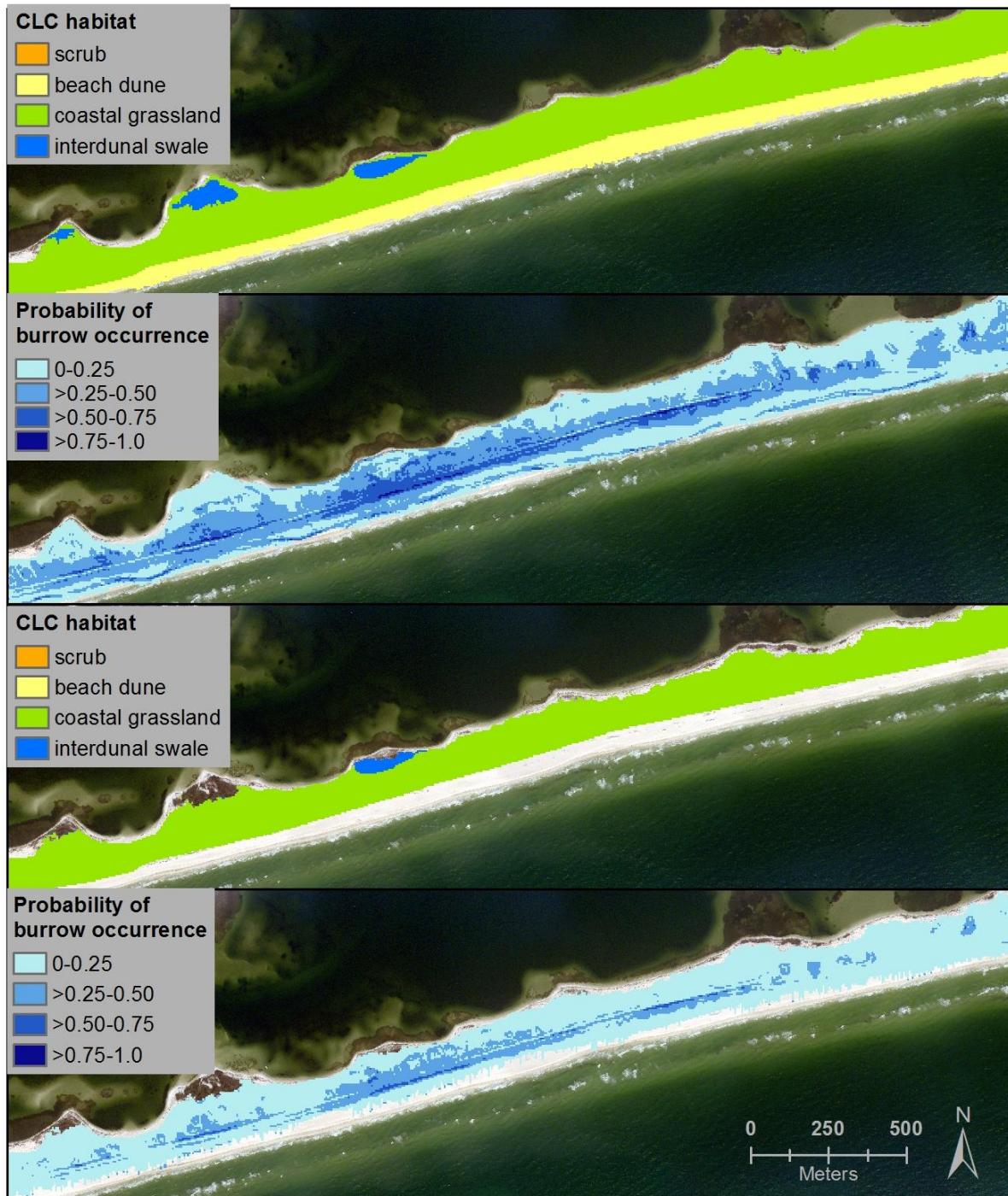


Figure 3-20. Beach mouse habitat remaining in **2025** (top panel) and **2100** (3rd panel) after sea level rise at a **rate of 1 m** (by 2100), represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.3 km, Figure 3-2C). The 2nd and bottom panels show probabilities of finding a beach mouse burrow in 2025 and 2100. Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

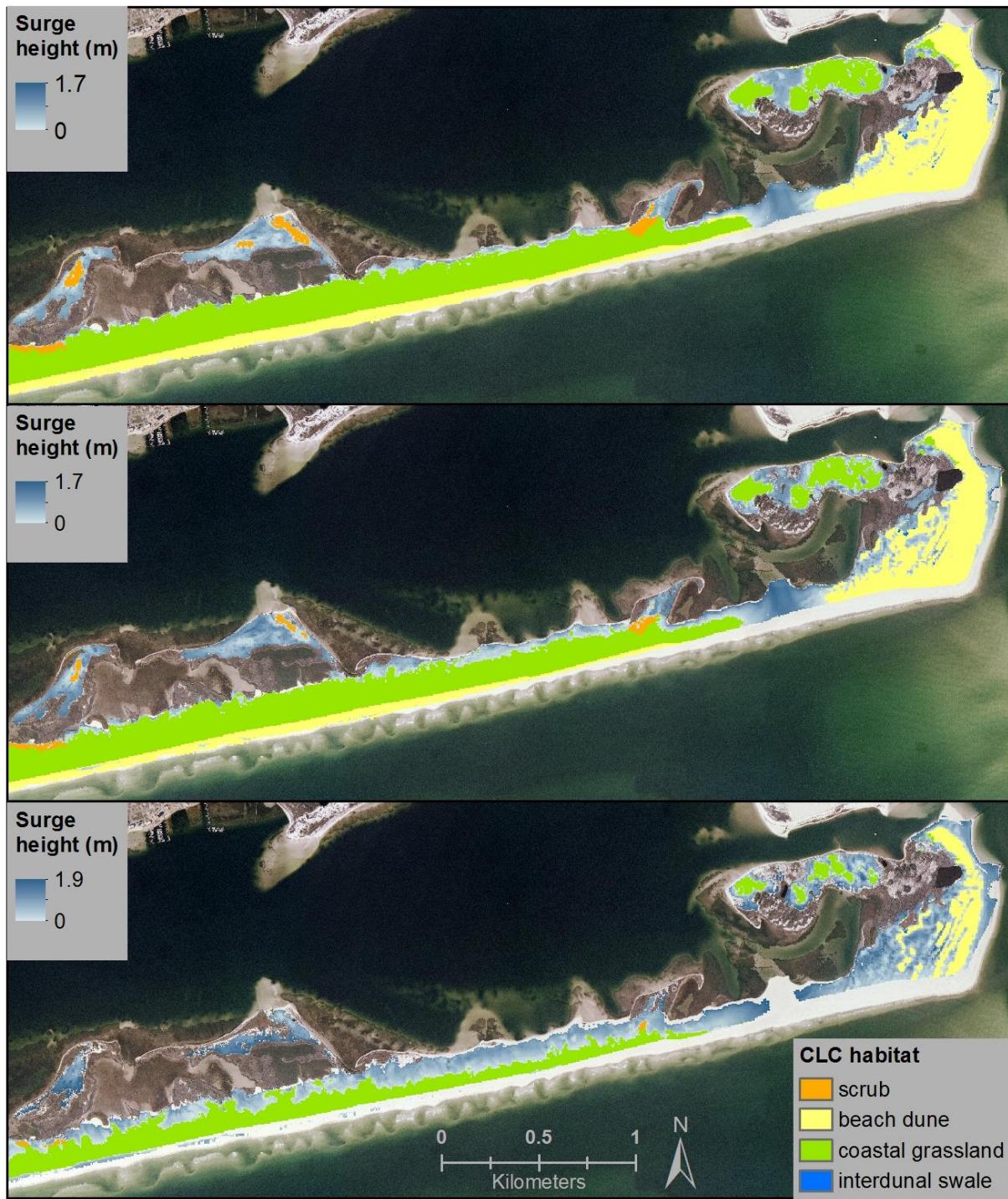


Figure 3-21. Beach mouse habitat remaining in 2025 (top), 2050 (middle), and 2100 (bottom) after sea level rise at a **rate of 1 m** (by 2100) and a **Category 1 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

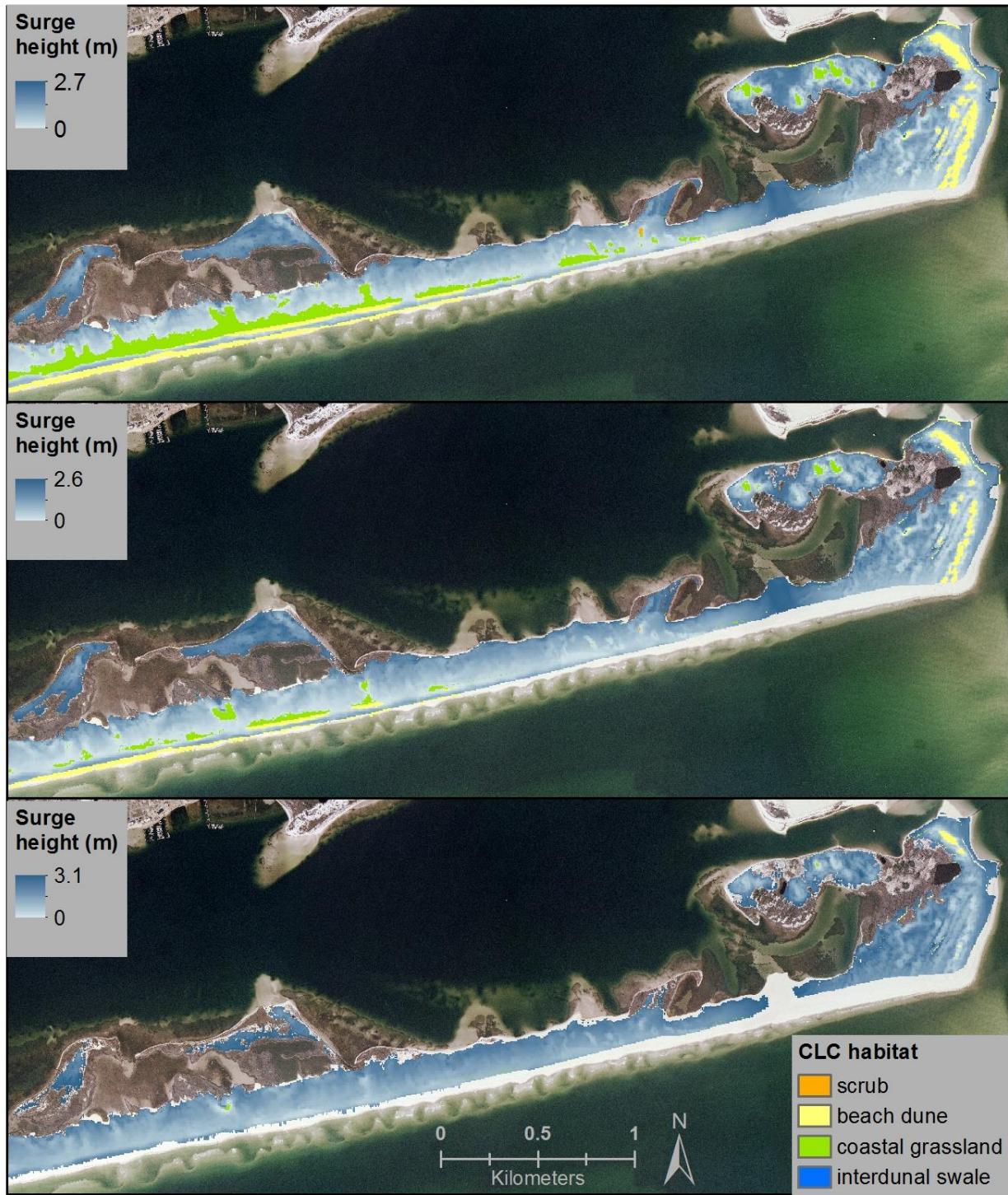


Figure 3-22. Beach mouse habitat remaining in **2025 (top)**, **2050 (middle)**, and **2100 (bottom)** after sea level rise at a **rate of 1 m** (by 2100) and a **Category 2 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

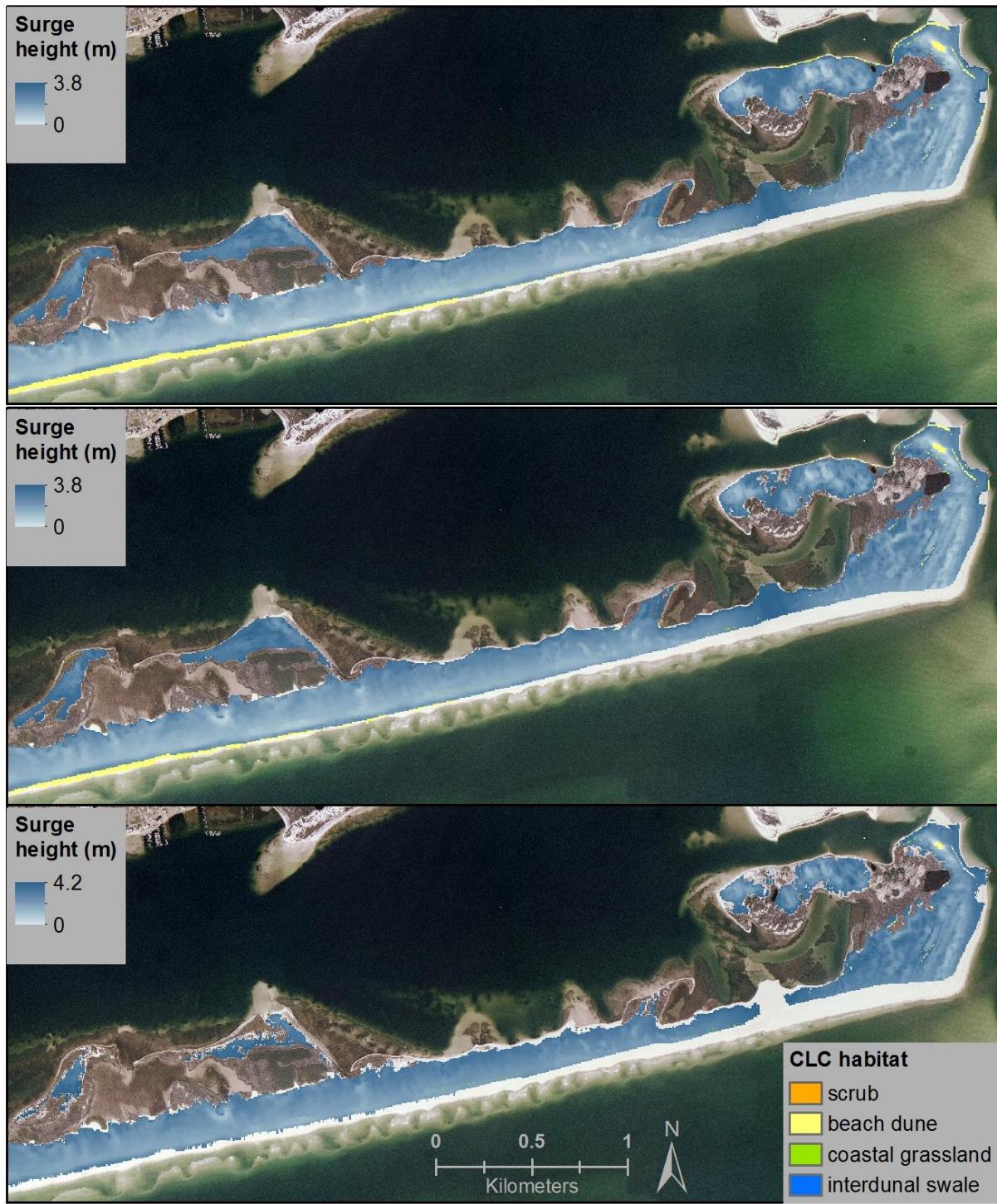


Figure 3-23. Beach mouse habitat remaining in **2025 (top)**, **2050 (middle)**, and **2100 (bottom)** after sea level rise at a **rate of 1 m** (by 2100) and a **Category 3 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~5.3 km, Figure 3-2C). Areas where the background image (NOAA 2017b) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

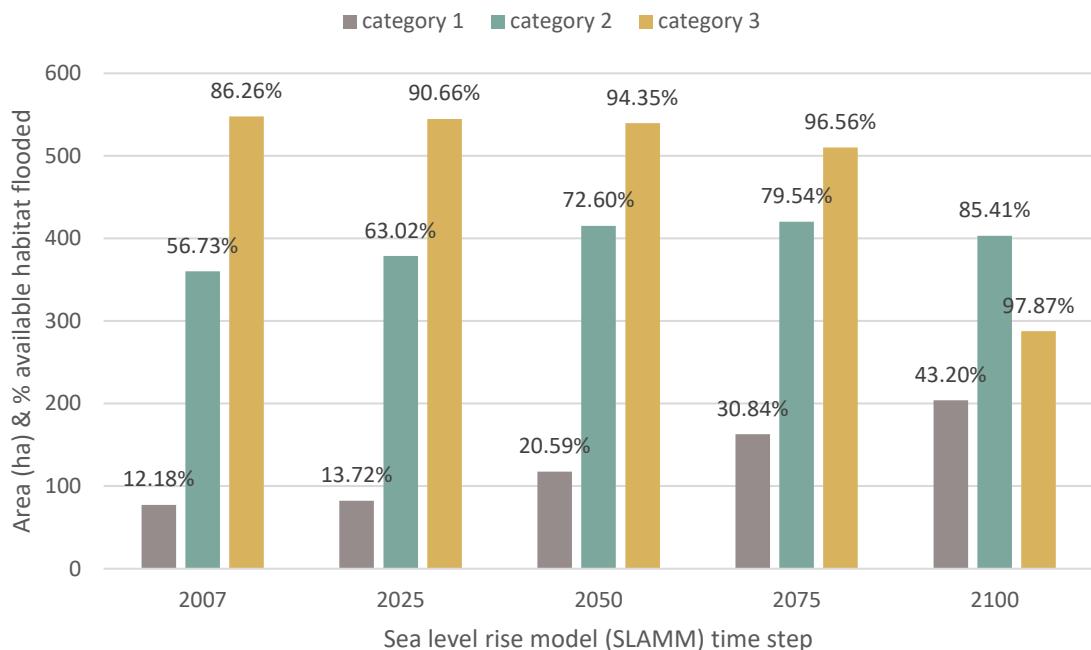


Figure 3-24. Area (ha) of potential beach mouse habitat on Perdido Key flooded by Category 1, 2, and 3 hurricanes after taking into account projected sea level rise (1-m rate) at different time steps. The number above each bar shows the percentage of potential beach mouse habitat expected to be flooded under each hurricane category and year combination. Potential habitat was calculated using the SLAMM sea level rise model; flooding was estimated using the SLOSH hurricane surge model. Results are based on Cooperative Land Cover (CLC) habitat categories (Table 3-1). See Appendix F, Figures F-1 and F-2 for changes in individual CLC and SLAMM habitat categories.

CHAPTER 4

IF YOU BUILD IT, WILL THEY WILL COME? USE OF RESTORED DUNES BY BEACH MICE

Introduction

Ecological systems around the world are degraded, fragmented, and destroyed by development and climate-related disturbances, leading to biodiversity loss (Wilson et al. 2016). Habitat restoration efforts in these compromised systems often focus on restoring the structure and composition of the plant community with the expectation that organisms that depend on this habitat will eventually return (Lithgow et al. 2013, McAlpine et al. 2016). Consequently, the progress and success of most restoration projects is measured using plant responses more than those of other species groups. This is the case in coastal areas where restoration is often necessary because of the intensity human impacts, e.g., from development and recreation, and the damaging effects of severe storms, e.g., erosion, wind damage, and flooding from storm surge (Schlacher et al. 2007, Defeo et al. 2009, Sims et al. 2013). Numerous studies document the efficacy of dune re-vegetation and rebuilding (Miller et al. 2001, Grafals-Soto 2012, Miller et al. 2014, Hesp and Hilton 2016), but few have measured wildlife responses to restoration in degraded dunes (Kutiel et al. 2000, Russell et al. 2009, Rolo et al. 2017). Likewise, no studies have measured wildlife responses in areas where dunes have been rebuilt after having been removed by storms or other processes. As sea level rise progresses and the intensity of tropical storms increases (Knutson et al. 2010, Rahmstorf and Coumou 2011, Grinsted et al. 2013, Vitousek et al. 2017), rebuilding habitat and restoring degraded habitat may become necessary to maintain plant and animal diversity and dune function in vulnerable coastal systems.

Tropical storms and hurricanes damage beach dune systems by flattening dunes and denuding them of vegetation, and creating or widening open-sand gaps between dunes (Morton and Sallenger 2003, Claudino-Sales et al. 2010). These low-lying gaps can become scoured and

widened as wind and storm surge push water into or across them, reducing the connectedness for species that rely on dunes and vegetation for movement and cover (Houser et al. 2008, Houser and Hamilton 2009). Beach dune restoration efforts that involve planting of native dune plants not only re-vegetate disturbed areas but can facilitate the re-growth of dunes after storms by helping to establish a root and shoot matrix on which dunes can grow. This matrix strengthens as plants become established, grow in height, spread laterally above and below ground, and catch sand that is moved by wind or water inland. As dunes develop, the physical connectedness of dunes increases, and presumably this increases connectivity for animal species that use this habitat.

Beach mice (*Peromyscus polionotus* spp) are one coastal species group whose populations are at risk due to habitat loss and fragmentation caused by storms and other disturbances. They occur only on narrow barrier islands and coastal peninsulas along the Florida and Alabama coasts, and all but one of the seven extant sub-species is listed as threatened or endangered (USFWS 1985). In these systems, vegetation, dune height, and slope of dunes are known to be important characteristics in selection of beach mouse burrow locations (Lynn 2000, Scheckenberger 2001) and of habitat in general (Branch et al. 2011). Also, increasing the width or number of sand gaps between dunes or vegetation patches is known to affect movement behavior of beach mice by increasing the risk perceived to move across the landscape (Wilkinson et al. 2013). Management actions such as dune plant restoration that create or re-establish dune habitat in disturbed areas or minimize the number of “risky” gaps mice must travel across are often hypothesized to benefit beach mice (USFWS 2007), but whether or not mice actually use and benefit from such efforts has not been evaluated. I addressed this knowledge gap by comparing the use of planted dune restoration plots by Perdido Key beach mice (*P. p.*

trissyllepsis) to their use of open sand gaps and natural dunes. Because light and cover can affect the amount of predation risk animals perceive (Brown 1999, Kotler et al. 2010) and influence movement decisions, I also evaluated whether risk associated with moonlight affects use of different plots. This study clarifies the utility in the short-term of dune plant restoration plots for creating and connecting beach mouse habitat in storm-impacted areas. Understanding the function of these plots may help managers implement restoration efforts that benefit beach mice as well as other dune-dependent species.

Methods

I assessed use of natural, open sand, and planted dune restoration sites by beach mice at four sites on land administered by Gulf Islands National Seashore on Perdido Key, FL. Sites were originally established in 2010 to assess sand accumulation and the influence of planting configuration on survival and growth of different plant species in planted restoration plots (Miller et al. 2016). The plot dimensions were approximately 5 x 65 m and included plantings of beach elder (*Iva imbricata*), bitter panicum (*Panicum amarum*), maritime bluestem (*Schizachyrium maritimum*), and sea oats (*Uniola paniculata*). The seeds of these species are known or potential food sources for beach mice (Sneckenberger 2001). I selected only four of six original restoration plots to reduce dependence among my sites, based on mean nightly movement distances of beach mice (87.6 m: Sneckenberger 2001; 75.4 m: Stoddard 2010). The shortest distance between sample points of different sites in this study was 115 m. Near each restoration plot, I identified an open sand gap and a naturally vegetated dune 60-70 m across or in length, and considered each 3-treatment area to represent a block. I installed one linear six-tube transect (9- or 10- m spacing between tubes) in each of the three treatments within each of the four blocks. Five-night trapping sessions were centered on the full moon and new moon and were conducted once early in summer and once in mid-summer during each moon phase (i.e.,

four total sessions per year for three years). Presence of mouse tracks in tubes was recorded each morning of a trapping session. Trapping sessions were completed between May 17 and July 5, 2012; June 6 and July 24, 2013; and June 11 and July 30, 2014. I visually estimated percent vegetation cover and average vegetation height in 2-m radius plots around each tube once in July of each year the study was conducted.

Tracking tubes were constructed from 1.5-inch diameter 30-40 cm PVC pipe sections (Pries et al. 2009)(Figure 4-1). A cap was attached to one end of the tube and a PVC elbow joint (through which mice enter) was attached to the other (Loggins et al. 2010). A paper liner with inkpad at one end was inserted into each tube, with the inkpad located at the tube entrance (i.e., the elbow). The ink solution is a non-toxic 1:2 mixture of carbon powder and food-grade mineral oil. When mice enter the tube and walk across the inkpad, tracks are registered on the paper liner and signal presence of mice at that point. A small quantity of whole oats was put at the closed end of each tube as bait. Tubes were elevated using rigid wire legs to prevent access by ghost crabs (*Ocypode quadrata*). Mice accessed elevated tubes using a dowel that was placed at the elbow entrance and inserted into the sand.

I used the GLIMMIX procedure in SAS (SAS Institute 2013) to model the influence of treatment, trapping season (early or mid-summer), moon, season*moon, season*treatment, and moon*treatment on the proportion of tracking tubes that had mouse tracks present over the total number of tubes deployed for each tracking session and treatment. So, the maximum number of tubes with tracks present was 30 (6 tubes in each treatment for each 5-night trapping session). Each tracking session comprised a 5-night trapping period during a particular year, moon phase, season, block, and treatment. I included year, year*season, year*season*moon, and year*season*moon*block as random effects. Differences in percent vegetation cover among

treatments were assessed using the MIXED procedure in SAS (SAS Institute 2013), with year and year*block included as random variables. For both analyses, the significance of main effects was assessed using Type III tests, and I used Tukey's tests to evaluate differences of least squares means within and among the fixed effects. I used $\alpha=0.05$ to assess whether or not results were significant in all analyses.

Results

In all trapping sessions, beach mice use of restored plots was intermediate between that in naturally vegetated dunes and open sand gaps (Figure 4-2). Differences among treatments was significant ($F(2,133)=388.53$, $p<0.001$), and this treatment effect was influenced by moon phase (full or new) ($F(2,133)=15.26$, $p<0.001$) and when data were collected—in early or mid- summer (i.e., season) ($F(2,133)=30.46$, $p<0.001$). Despite interactions, the main effect of treatment alone (i.e., open sand gap v. naturally vegetated dunes v. restoration plots) is meaningful because there is a consistent pattern in treatment responses, with the proportion of tracking tubes used by beach mice always occurring—from highest to lowest—in the naturally vegetated dunes, restoration plots, and open sand gaps. This pattern, in which mice use the most vegetated sites most and the least vegetated sites least exists regardless of the presence of interactions between treatment and moon (Figure 4-2) or treatment and season (Figure 4-3). Without considering moon and season, proportions of tubes visited were different between all combinations of treatment pairs (natural v. open: $t=27.28$, $df=133$, adj. $p<0.0001$; natural v. restoration: $t=11.82$, $df=133$, adj. $p<0.0001$; open v. restoration: $t=-18.26$, $df=133$, adj. $p<0.0001$).

Mice always used sites more during the new moon when ambient light was low than during the full moon when ambient light was high in all treatments (Figure 4-2), although this difference was significant only in the restoration plots ($t=-4.32$, $df=4.46$, adj. $p=0.01$). All possible comparisons but the following four were significantly different between moon and

treatment levels (Figure 4-2): 1. Naturally-vegetated dunes during the full v. new moon ($t=-0.84$, $df=4.60$, adj. $p=0.96$); 2. Naturally-vegetated dunes during the full moon v. restoration plots during the new moon ($t=-0.45$, $df=4.49$, adj. $p=1.0$); 3. Open sand gaps during the full v. new moon ($t=-2.63$, $df=4.43$, adj. $p=0.10$); and 4. Restoration plots during the full moon v. open sand gaps during the new moon ($t=-0.80$, $df=4.28$, adj. $p=0.97$).

In general, use of tubes by mice was higher in mid-summer than early in the summer in all treatments (Figure 4-3). This difference was significant in the restoration plots ($t=-5.06$, $df=2.24$, adj. $p<0.0001$), but not in the naturally vegetated dunes ($t=-2.17$, $df=2.30$, adj. $p=0.26$) nor in the open sand gaps ($t=-2.75$, $df=2.22$, adj. $p=0.07$). Use of naturally-vegetated dunes in early season was similar to use of restoration plots late in the season ($t=-1.32$, $df=2.23$, adj. $p=0.77$), and use of restoration plots in early season was similar to use of open sand gaps late in the season ($t=-0.48$, $df=2.16$, adj. $p=1.0$). Over time, the pattern of use in treatments remained the same, with use always highest in the naturally-vegetated dunes and lowest in the open sand gaps in all years. Use in 2014 was lower than in 2012 and 2013, particularly in the open sand gaps (Figure 4-4).

Vegetation cover and height remained relatively constant in the naturally-vegetated dune and open gap sites between 2012 and 2014 but increased in the restoration plots (Figures 4-5 and 4-6). Natural vegetation sites had the tallest and densest vegetation, followed by that in the restoration plots and open plots, which were mostly unvegetated (Figures 4-5 and 4-6).

Vegetation differed by treatment for mean % cover ($F(2,22)=118.39$, $p<0.0001$) and mean height ($F(2,22)=78.01$, $p<0.0001$). Pairwise comparisons of treatments (i.e., natural v. open, restoration v. natural, open v. restoration) also differed for % cover and height (adjusted p-value<0.0001 for all comparisons except height in natural v. restoration, adj. $p=0.10$). The lack of difference in

height between naturally-vegetated dunes and restoration plots is likely due to the similar heights in both treatments in 2014.

Discussion

Study results indicate that within two years of establishment dune restoration plots can provide habitat for beach mice. Mice used restoration plots consistently more than open sand gaps, although after three years, vegetation cover in restored plots and mouse activity remained lower than in natural dunes. Mice used restored habitat for foraging and movement but did not burrow in restored plots during this study. Vegetation and sand accumulation increased since plots were established in 2010 (Miller et al. 2016) and barring major disturbances, restoration plots are expected to eventually resemble natural dunes more closely in the cover, elevation, and slope characteristics associated with beach mouse burrows (Lynn 2000, Sneckenberger 2001), and be used as burrow habitat as well.

Barrier islands and coastal areas that beach mice inhabit comprise a patchwork of dunes and open areas. So, mice must move across open gaps, but previous field studies show that mice are less likely to do so when gaps become too wide (Wilkinson et al. 2013). Therefore, low use of open sand gaps relative to restored and natural dunes was expected, given gaps provide no cover and likely represent areas with high risk of perceived predation for mice (Brown 1999). Restoration plots planted in gaps that have been created or made larger by flooding from storms may initially serve as stepping stones for mice to move across gaps, and ultimately these dunes will provide nesting and foraging habitat as dunes build vegetation and become established. Also, the potential benefits of restoration plots to beach mice and other dune-dependent species extends to facilitating movement at broader spatial scales as well, e.g., for long-distance dispersal. Severe storms can flatten very wide areas, particularly across narrow parts of barrier islands, creating barriers to movement and recolonization following local extinction (Oli et al.

2001). Beach mice have been shown to move through and reside in relatively narrow strips of dune habitat parallel to shore (Austin et al. 2015). So, even long narrow restoration plots may enhance connectivity for mice and other dune-dependent species by providing food and cover as they disperse across the landscape, and, as a result, enhance reestablishment of populations and gene flow among existing sub-populations.

Our observation that moon phase influenced beach mouse activity has been observed in other studies on beach mice (Wilkinson et al. 2013) as well as many other taxa (Orrock et al. 2004, Kotler et al. 2010, Prugh and Golden 2014). Activity is believed to decrease as moonlight increases because moonlight is an indirect cue that elevates risk perceived by some animals (Abramsky et al. 2002, Orrock et al. 2004, Fanson 2010). My findings corroborate this idea; use of tubes by mice was always lower during the full than the new moon, although this difference was significant only in the restoration plots. Perceived risk associated with amount of vegetation cover may explain why some comparisons between treatment and moon phase did not differ. Cover likely mitigates some but not all of the additional perceived risk of movement by mice during the full moon. So, risk associated with use of an area with intermediate cover (restoration plot) under low risk conditions (new moon) may be perceived as similar as risk associated with use of an area with high cover (natural dune) under high risk conditions (full moon). Likewise, use of low cover areas (open sand gap) under low risk (new moon) would be similar to use of intermediate cover areas (restoration plot) under high risk (full moon). Perception of risk by beach mice also increases with artificial light (Bird et al. 2004).

The difference in use of tubes between the first and second trapping sessions during a summer is likely explained by the acclimation of mice to tubes. Over time, animals can learn the location of known food resources and may travel from food patch to food patch (Gill 1988). So,

once baited tracking tubes had been discovered early in the season, resident beach mice likely learned to associate tubes with food and therefore visited tubes more later than earlier in the summer. Seasonal differences were particularly large in the restoration plot, and early summer use in the restoration plots did not differ from that in the open sand gaps in mid-summer. While restoration plots were still novel (early summer), mice may have equated risk of using these areas with that in open gaps but once restoration plots were discovered, mice apparently perceived as little risk in using these plots as they would in naturally vegetated dunes. It is also possible that an increase in cover from early to mid-summer led to an increase in use by mice by reducing the predation risk mice may have perceived. Food resources for mice are generally low in the summer (Sneckenberger 2001), and I would not expect a change in resources from early to mid-summer that would influence behavior of mice. As dune height and vegetation increase in restoration plots and become even more similar to natural dunes, mice likely will use these areas in a manner similar to that of natural dunes regardless of season or moon phase.

Restoration of dunes through planting can significantly enhance use of restored areas by beach mice compared to areas that remain open after storms. Because foraging by beach mice is affected by artificial light and vegetation cover mitigates the impacts of lights (Bird et al. 2004), restoration in areas with artificial lights may be particularly important. Important next steps for restoration studies are identification of means to facilitate restoration of dunes that provide sufficient food (i.e., quantity and quality) for beach mice throughout the year and identification of the dimensions of restoration plots (e.g., width and dune height) where plots transition from only facilitating movement to providing year-around habitat for mice for foraging and reproduction (i.e., burrow sites).

Identifying how restoration can reduce fragmentation and enhance long-term connectivity across the landscape should also be a priority. In coastal systems, this will mean moving from a site- and plant-based approach to restoration to one that takes into account how the local (e.g., foraging) and broad-scale (e.g., dispersal) needs of different species will be met (McAlpine et al. 2016). A main challenge will be to find a balance between restoring and protecting vegetated dunes that provide habitat for species like beach mice while also preserving natural processes like overwash that shape open habitat used by other sensitive species like shorebirds (Seavey et al. 2010, Maslo et al. 2011, Schupp et al. 2013).



Figure 4-1. Tracking tube deployed in the field. Beach mouse tracks can be seen around the tube. Photo courtesy of the author.

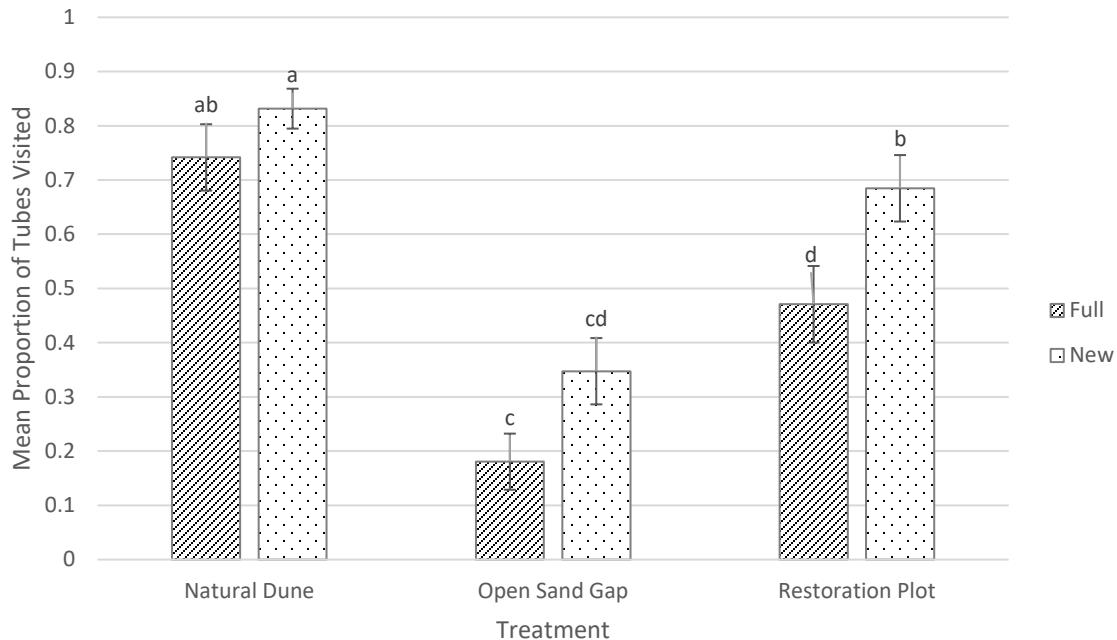


Figure 4-2. Mean proportion (+ 1 SE) of tracking tubes visited by Perdido Key beach mice in three treatments during new and full moon phases. The same letters above different bars signify means were the same for the variables represented by the bars (based on Tukey's test and $\alpha=0.05$).

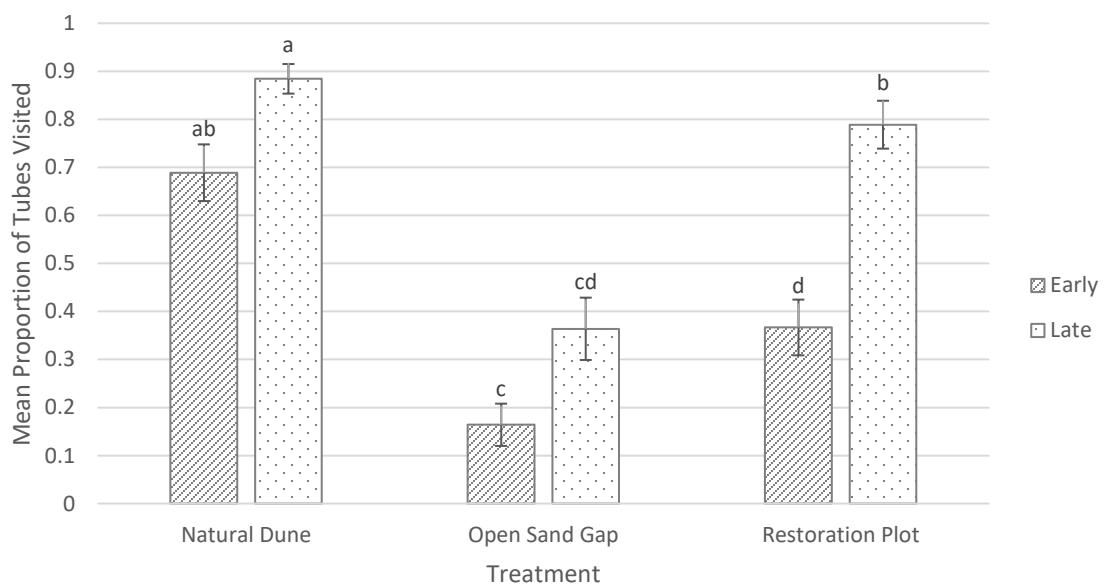


Figure 4-3. Mean proportion (+ 1 SE) of tracking tubes visited by Perdido Key beach mice in three treatments during trapping sessions in early vs. mid- summer. The same letters above different bars signify means were the same for the variables represented by the bars (based on Tukey's test and $\alpha=0.05$).

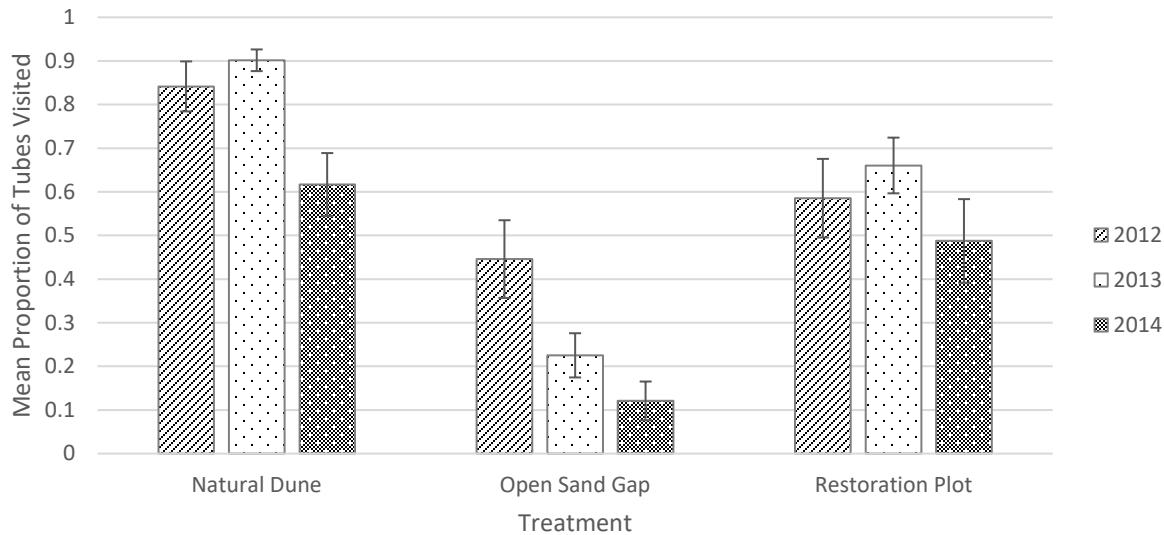


Figure 4-4. Mean proportion (± 1 SE) of tracking tubes visited by Perdido Key beach mice in three treatments across study years. All pairwise combinations at the treatment level (e.g., Restoration Plot versus Open Sand Gap) differed, based on Tukey's test and $\alpha=0.05$.

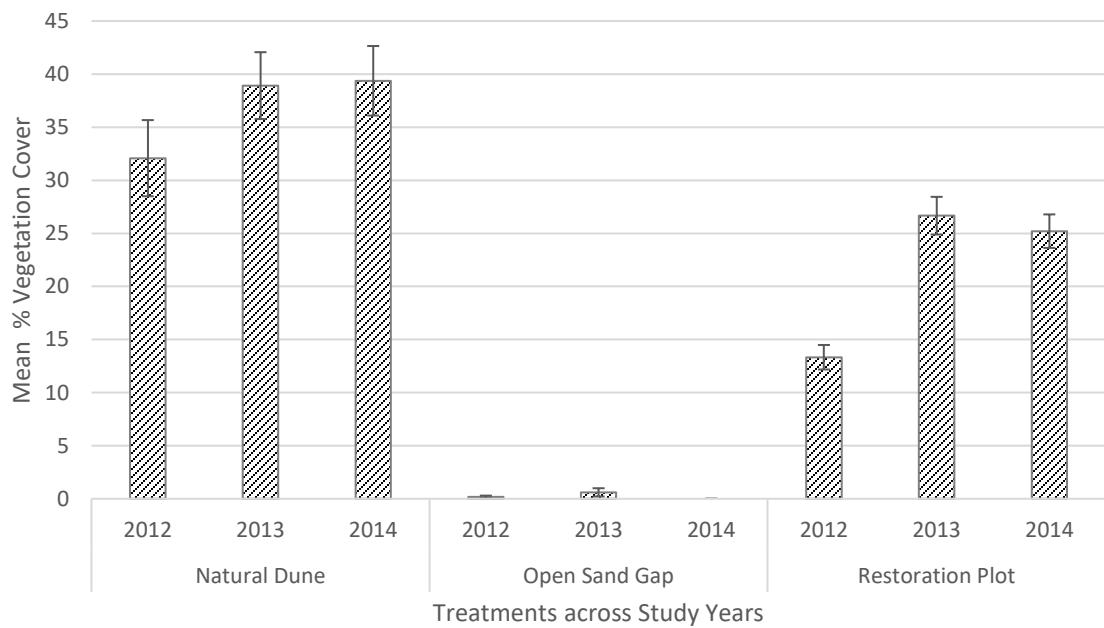


Figure 4-5. Mean percent vegetation cover (± 1 SE) in three treatments by year. Vegetation cover was estimated visually in a 2-m plot centered on each tracking tube. All pairwise combinations at the treatment level (e.g., Restoration Plot versus Open Sand Gap) differed, based on Tukey's test and $\alpha=0.05$.

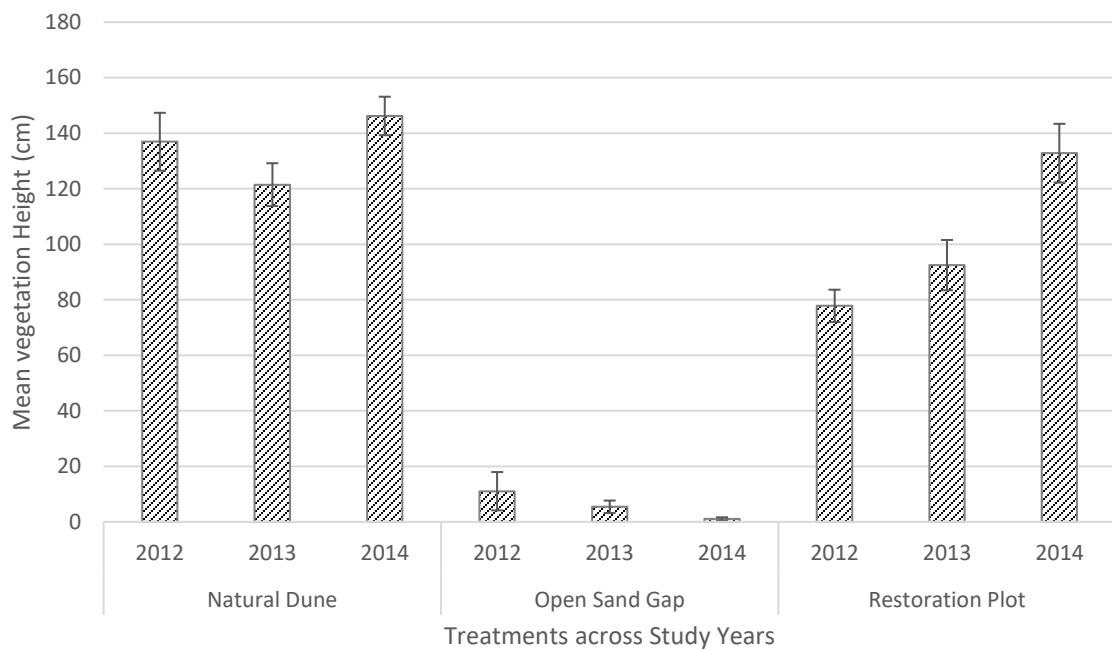


Figure 4-6. Mean vegetation height in cm (± 1 SE) in three treatments by year. Vegetation height was measured in a 2-m plot centered on each tracking tube. All pairwise combinations at the treatment level except Restoration Plot versus Natural Dune differed, based on Tukey's test and $\alpha=0.05$.

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

The main goal of this project was to provide information on the foraging behavior, distribution, and habitat of beach mice (*Peromyscus polionotus* spp) in development- and climate-impacted landscapes under current and future expected conditions following sea level rise and storm surge. I also assess how dune restoration might benefit beach mice. Results presented in this dissertation can be used to make informed decisions on how to best manage and restore coastal habitat to ensure the persistence of beach mouse populations.

In Chapter 2 I determined whether and how landscape structure influences perceived risk of predation under ambient and heightened risk by comparing foraging in natural habitat (microhabitat and landscape scales) and experimental patches (patch scale). Mice perceived risk with an owl cue, but not with cues of cats and foxes, and therefore may not avoid areas used by these known beach mouse predators. This behavior can make beach mice particularly vulnerable to predation, and control of non-native predators needs to be evaluated and applied carefully. Mice foraged less under new v. full moon conditions, and under artificial lights v. low ambient light, apparently avoiding lit areas. Lighting could imperil beach mouse populations by restricting the amount of habitat available to them to use, even when vegetation or other characteristics appear to be adequate as habitat. Although I did not observe a clear effect of corridors or patch size on foraging by mice in my experimental plots, this finding may be related to the small range of patch and corridor sizes in my experiments. These sizes may not have been sufficiently different to induce different behavior by mice. Given this, my results should not be interpreted to mean these factors should not be considered when designing management and conservation plans for beach mice. Overall, the results of this work highlight the importance of

measuring factors that can affect risk perceived by animals, including indirect and direct cues, in addition to physical habitat characteristics when evaluating wildlife habitat relationships.

In Chapter 3, I show both high quality burrowing habitat found primarily in dunes with relatively high elevation and woody vegetation, and habitat that may not support burrows but can be important for foraging and movement between burrow sites (e.g., lower elevation beach dunes and coastal grasslands) will be lost to sea level rise. The effects of storm surge on beach mouse habitat will worsen as sea level rise progresses because flooding from surge will be more extensive as the coast retreats with increasing water levels. Up to 98% of remaining beach mouse habitat will be flooded from a Category 3 storm under a 1-m rate of sea level rise. Protection of tall dunes from human impacts and dune restoration in gaps between existing dunes should be implemented if maintenance of good burrow habitat and connectivity between these patches is a priority. This work outlines an approach for identifying climate-related effects on beach mouse habitat and areas that should receive conservation priority as important burrow habitat or potential refugia for mice during storms.

In Chapter 4, I assessed use of restoration plots, naturally vegetated dunes, and open sand gaps by beach mice. I found restoration plots provide habitat for movement of beach mice within two years of establishment. If connectivity of habitat for beach mice is a concern, dune restoration plots should be established as soon as possible in areas where dunes have been flattened and denuded by weather-related disturbances. By doing so, plants can become well-rooted and dunes will be able to develop the characteristics necessary to facilitate movement by beach mice and burrowing (i.e., sufficient elevation, slope, and vegetation cover). Restoration plots can be established to facilitate long distance movements (i.e., dispersal) by beach mice

across wide areas without dunes as well. This will hedge against isolation of subpopulations in the future.

APPENDIX A COMPARISON OF SLAMM and CLC RESULTS

The SLAMM model is able to calculate how habitat types change dynamically as sea level rise progresses. For example, an area of “inland freshwater marsh,” a SLAMM category that includes “coastal interdunal swale” (a habitat known to be used by beach mice) could transition to “ocean beach” as current coastal habitats are inundated and dry beach habitat moves inland. This type of model differs from “bathtub” models of inundation in which water levels are raised resulting in inundations, but not shifts in habitat types. With simple bathtub models the area of particular habitats can constrict as land is flooded and habitats are covered by water, but habitats do not change from one type to another (Figure A-1).

SLAMM habitat categories, which are one of the required model input files to the SLAMM model (i.e., as a land cover file) are based on CLC categories in this study area, but SLAMM categories are much courser than CLC categories. I modeled SLAMM output and also generated output based on CLC categories by overlaying on, and clipping CLC habitat maps from SLAMM model output. Because results based on CLC cover types are calculated by overlaying the same static raster map of CLC cover types on top of remaining beach mouse habitat calculated from SLAMM model results for each rate of sea level rise and time step, CLC-habitat types constrict from the island perimeter as land areas are inundated with sea level rise but do not change dynamically (Figure A-1). For example, coastal grassland—a CLC habitat type that is part of the coarser SLAMM undeveloped dry habitat type—will remain coastal grassland across timesteps unless it is covered by water. The same area on the SLAMM map (undeveloped dry land) could be covered in water or change to another land cover classified as beach mouse habitat, such as ocean beach. This movement inland of beach dunes, displacing undeveloped dry land was observed in SLAMM results as were (few) other dynamic changes in

habitat. In some areas this new beach habitat could eventually transition to scrub dunes under the right conditions (i.e., with sand accumulation, woody plant establishment, etc.), which would likely provide better habitat than relatively flat areas of beach dune. In SLAMM however, this transition would be from “ocean beach” to the broad category “undeveloped dry land” and the relative size and thus importance to beach mice of more specific habitat types within undeveloped dry land—such as coastal grassland and scrub—would be unknown. Because of the coarseness of the SLAMM habitat categories, and because SLAMM does not account for processes such as growth of dunes over the time, the value to beach mice of some habitats is likely to be underestimated by this modelling approach. Furthermore, because SLAMM is originally based on CLC categories, SLAMM-based maps and CLC-based maps were for the most part very similar. CLC maps were superior in some cases because they provide more detailed habitat information, and through the modeling and overlay process (of CLC coverage over SLAMM output), habitat erroneously categorized as beach mouse habitat (by SLAMM) was re-classified correctly as non-habitat using CLC categories.

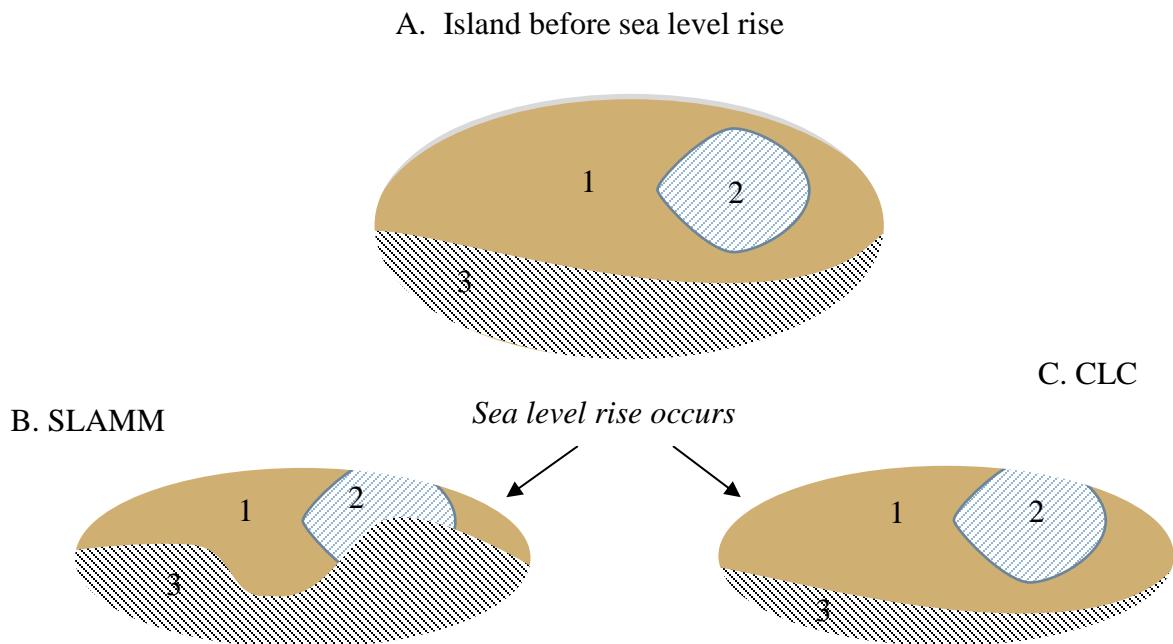


Figure A-1. Hypothetical island showing how the SLAMM sea level rise model allows habitat to change dynamically as sea level rises; as island area constricts (from A. to B.), some of habitats 1 and 2 become habitat 3 as habitat 3 moves inland (B.). The results based on the Cooperative Land Cover (CLC) land cover data incorporate constriction of island area as well (from A. to C.), but the areas of individual habitat patches change only as a result of inundation from sea level rise (C.) and not a redistribution of habitat types.

APPENDIX B

SLAMM MODEL INPUT DETAILS

Table B-1. Main LiDAR datasets used in SLAMM model.

Area	Dataset	Source
Perdido Key & Santa Rosa Island	2006 NFWFMD LiDAR: Escambia, Santa Rosa, and Walton Counties ^{1, 2}	NOAA Office for Coastal Management Digital Coast
Santa Rosa Island	2007 FDEM LiDAR: Coastal Okaloosa County ^{1, 2}	https://coast.noaa.gov/digitalcoast/
Perdido Key (AL section)	2010 USACE NCMP Topobathy LiDAR: Gulf Coast (AL, FL) ²	

¹ Primary dataset covering Perdido Key and Santa Rosa Island included in the DEM mosaic of Florida from the UF GeoPlan Center. The mosaic dataset is available through the Florida Geographic Data Library (<http://www.fgdl.org/metadataexplorer/explorer.jsp>).

² Metadata can be accessed by searching on the dataset name at <http://coast.noaa.gov/dataregistry/search/collection>.

Table B-2. Site parameters used in SLAMM model.

Parameter	Site		
	PK	SRI	Global
Description			
Land cover photo date ¹	2007	2007	2007
DEM Date ²	2006	2006	2006
Direction Offshore [N, S, E, W]	South	South	South
Historic trend in sea level rise (mm/year) ³	2.25	2.25	2.25
NAVD correction (MTL-NAVD88, m) ⁴	n/a	n/a	n/a
Great diurnal tide range (m) ⁵	0.307	0.343	0.328
Salt elevation (m above MTL) ⁶	0.382	0.426	0.408
Marsh erosion rate (horizontal m/year) ⁷	2.0	2.0	2.0
Swamp erosion rate (horizontal m/year) ⁷	1.0	1.0	1.0
Tidal flat erosion rate (horizontal m/year) ⁷	0.5	0.5	0.5
Saltmarsh (regularly flooded) accretion rate (mm/year) ⁸	2.25	2.25	2.25
Brackish (irregularly flooded) marsh accretion rate (mm/year) ⁸	3.75	3.75	3.75
Tidal freshwater marsh accretion rate (mm/year) ⁸	4.0	4.0	4.0
Beach sedimentation rate (mm/year) ⁸	0.5	0.5	0.5
Frequency of overwash (years) ⁹	n/a	n/a	n/a
Used elevation pre-processor [True, False]	FALSE	FALSE	FALSE

¹ CLC land cover data; year represents approximate mean year of collection of data from different sources that contributed to the landcover dataset.

² DEM date based on information in LiDAR dataset metadata.

³ From summary information on Pensacola, FL tide station (# 8729840) at <https://tidesandcurrents.noaa.gov/>.

⁴ A raster of correction values created using the Vdatum program (vdatum.noaa.gov) was used instead of a constant correction factor.

⁵ Mean of diurnal range values for tidal stations in each subsite or for all sites together (global value), Values obtained from <https://tidesandcurrents.noaa.gov/>. See Table B-3.

⁶ Mean of salt elevation values for tidal stations in each subsite or for all sites together (global value). See Table B-3 for description of approach to estimate salt elevation.

⁷ SLAMM default values, following Geselbracht et al. (2015).

⁸ From Clough (2006), following Geselbracht et al. (2015).

⁹ Overwash function was turned off due to the relatively small cell size and inconsistencies in output observed with modeling sea level rise around barrier islands, per recommendations in Clough (2012) and comments in the SLAMM forum (<http://warrenpinnacle.com/SLAMMFORUM/>).

Table B-3. NOAA tide stations from which data were collected to calculate SLAMM parameters.
The diurnal range value for each station is shown. Global gauge parameters comprised data from PK and SRI stations.

Subsite	Station Name	Station ID	Great tide range, Gt (m) ¹	Estimated salt elevation (m above MTL) ²
PK	Big Lagoon, FL	8729909	0.311	0.387
PK	Perdido Key, Old River, FL1	8729974	0.247	0.307
PK	Alabama Point, AL	8730667	0.262	0.326
PK & SRI	Pensacola, Naval Air Station, FL	8729868	0.366	0.455
PK & SRI	Fort Pickens, Pensacola Bay, FL	8729882	0.347	0.432
SRI	Navarre Beach, FL	8729678	0.384	0.478
SRI	Santa Rosa Sound, East End, FL	8729679	0.421	0.523
SRI	Woodlawn Beach, FL	8729736	0.393	0.489
SRI	Fishing Bend, Santa Rosa Sound, FL	8729806	0.390	0.485
SRI	Harris, Santa Rosa Sound, FL	8729613	0.381	0.474
SRI	Destin, East Pass, FL	8729511	0.186	0.231
SRI	Little Sabine Bay, FL	8729808	0.393	0.489
SRI	Hurlburt Field, FL	8729598	0.354	0.440
SRI	Ft. Walton Beach, Santa Rosa Sound, FL	8729554	0.155	0.193

¹ From summary information on each tidal station at <https://tidesandcurrents.noaa.gov/>; diurnal range values were averaged by subsite to input in the SLAMM model (see Table B-2). The value for the global value is the mean of the subsite values.

² To estimate salt elevations, I used the inundation analysis function at <https://tidesandcurrents.noaa.gov/> to plot a frequency table of MHHW (above MTL) for each of five years (2007-2011) for the Pensacola, FL tide station (8729840); identified the elevation (i.e., midpoint of bin value) at which inundation occurred ~1x/mo (i.e., frequency=9 to 11 per year); and calculated the ratio of this elevation to the diurnal range of the Pensacola tidal station (0.384 m) to get the percentage of the SLAMM tide range that represents the salt boundary. This value (1.24) was used to calculate salt elevations for each station using the formula: [1.24 x Gt]. The mean of salt elevations by subsite or of all sites (global value) were used as input parameters in the SLAMM model (see Table B-1). The Pensacola station data was used because it is the closest active station to the study area for which historic tide data is available. MHHW was assumed to approximate the threshold at which inundation can begin to occur.

Table B-4. Change in SLAMM wetland categories present on Santa Rosa Island between initial condition (2007, vegetation cover date) and Time 0 (1990). For wetland categories comprising >2% of the area modeled, I considered a change of <5% between the initial condition and Time 0 to suggest that the SLAMM model performed well. Inland freshwater marsh fell outside this range, but because it comprised a relatively small area, I presumed the model performed well overall. Italicized categories are those designated as potential beach mouse habitat in this report (see Table 3-1).

SLAMM Wetland Category	Initial Condition (IC, ha)	% of Study Area	Time 0 (T0, ha)	% Change	% change, IC-T0
<i>Estuarine Open Water</i>	<i>4131.51</i>	<i>37.47</i>	<i>4103.86</i>	-27.65	-0.67
<i>Open Ocean</i>	<i>2895.54</i>	<i>26.26</i>	<i>2894.49</i>	-1.05	-0.04
<i>Undeveloped Dry Land</i>	<i>1767.68</i>	<i>16.03</i>	<i>1795.86</i>	28.18	1.59
<i>Ocean Beach</i>	<i>875.98</i>	<i>7.94</i>	<i>875.18</i>	-0.80	-0.09
<i>Developed Dry Land</i>	<i>829.48</i>	<i>7.52</i>	<i>843.14</i>	13.66	1.65
<i>Inland-Freshwater Marsh</i>	<i>269.47</i>	<i>2.44</i>	<i>310.82</i>	41.35	15.34
Trans. Salt Marsh	68.06	0.62	0.00	-68.06	-100.00
Swamp	66.24	0.60	71.57	5.32	8.04
Tidal Flat	49.83	0.45	40.34	-9.49	-19.04
Regularly-Flooded Marsh	45.66	0.41	55.77	10.11	22.15
Estuarine Beach	18.60	0.17	0.00	-18.60	-100.00
Inland Open Water	8.60	0.08	35.62	27.02	314.35

APPENDIX C

ISLAND-WIDE CHANGES IN HABITAT TYPES

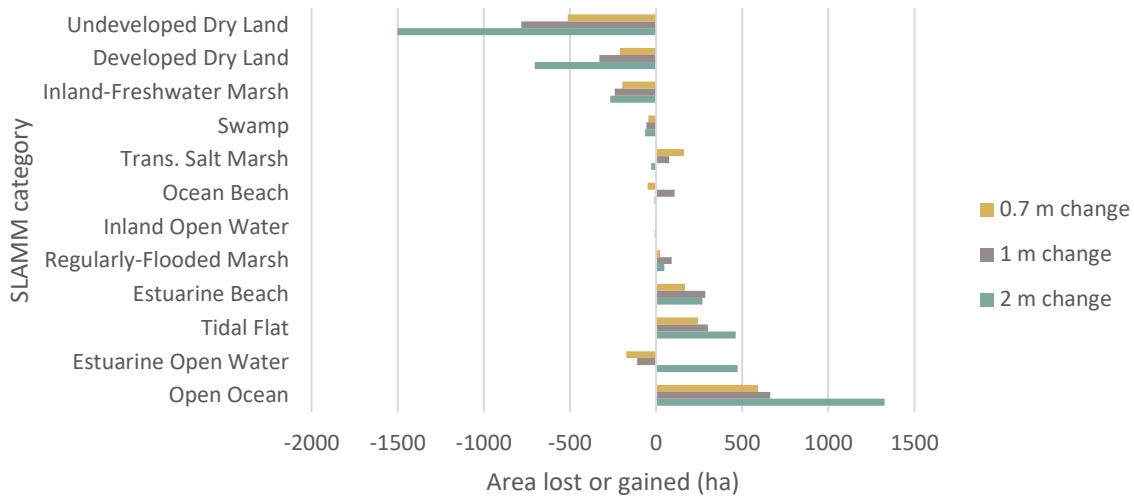


Figure C-1. Change in dry land or wetland area (SLAMM categories) on and surrounding Santa Rosa Island between 2007 & 2100 for three sea level rise rates. NOTE: Actual change in Developed dry land actual was larger than depicted (>8K% in area) for the 2-m SLR rate; and only Undeveloped Dry Land, Inland Freshwater Marsh, and Ocean Beach were considered to include potential beach mouse habitat (see Table 3-1).

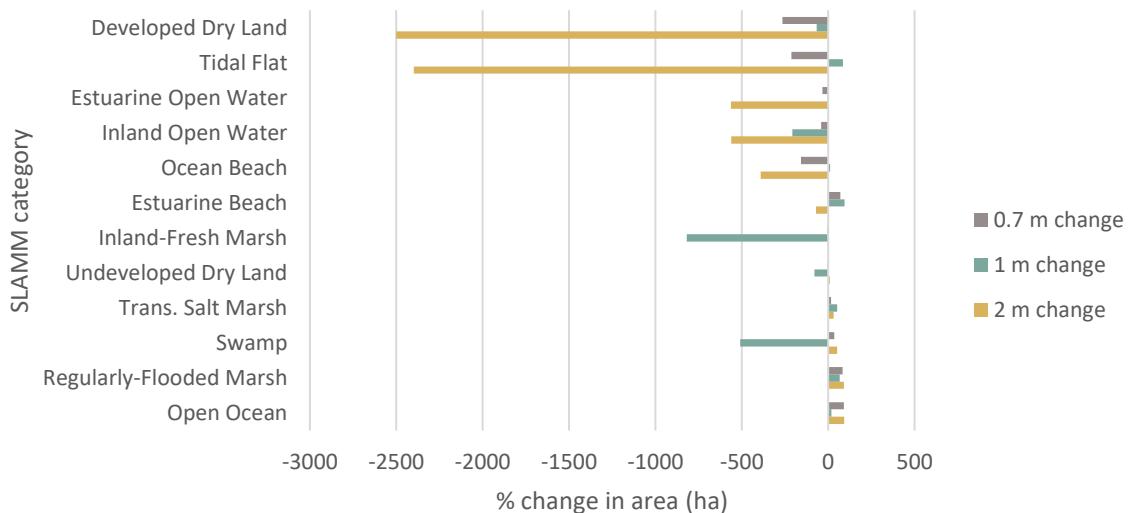


Figure C-2. Percentage change in dry land or wetland area (SLAMM categories) on and surrounding Santa Rosa Island between 2007 & 2100 for three sea level rise rates. NOTE: Actual change in Developed dry land was >8K% for the 2-m SLR rate; and only Undeveloped Dry Land, Inland Freshwater Marsh, and Ocean Beach were considered to include potential beach mouse habitat (see Table 3-1).

APPENDIX D
SAFFIR-SIMPSON HURRICANE WIND SCALE

Category	Sustained Winds	Types of Damage Due to Hurricane Winds
1	74-95 mph 64-82 kt 119-153 km/h	Very dangerous winds will produce some damage: Well-constructed frame homes could have damage to roof, shingles, vinyl siding and gutters. Large branches of trees will snap and shallowly rooted trees may be toppled. Extensive damage to power lines and poles likely will result in power outages that could last a few to several days.
2	96-110 mph 83-95 kt 154-177 km/h	Extremely dangerous winds will cause extensive damage: Well-constructed frame homes could sustain major roof and siding damage. Many shallowly rooted trees will be snapped or uprooted and block numerous roads. Near-total power loss is expected with outages that could last from several days to weeks.
3 (major)	111-129 mph 96-112 kt 178-208 km/h	Devastating damage will occur: Well-built framed homes may incur major damage or removal of roof decking and gable ends. Many trees will be snapped or uprooted, blocking numerous roads. Electricity and water will be unavailable for several days to weeks after the storm passes.
4 (major)	130-156 mph 113-136 kt 209-251 km/h	Catastrophic damage will occur: Well-built framed homes can sustain severe damage with loss of most of the roof structure and/or some exterior walls. Most trees will be snapped or uprooted and power poles downed. Fallen trees and power poles will isolate residential areas. Power outages will last weeks to possibly months. Most of the area will be uninhabitable for weeks or months.
5 (major)	157 mph or higher 137 kt or higher 252 km/h or higher	Catastrophic damage will occur: A high percentage of framed homes will be destroyed, with total roof failure and wall collapse. Fallen trees and power poles will isolate residential areas. Power outages will last for weeks to possibly months. Most of the area will be uninhabitable for weeks or months.

Figure D-1. Saffir-Simpson Hurricane Wind Scale; source:
<http://www.nhc.noaa.gov/aboutsshws.php>.

APPENDIX E
PERDIDO KEY STATE PARK AND WEST GINS HURRICANE SURGE MAPS

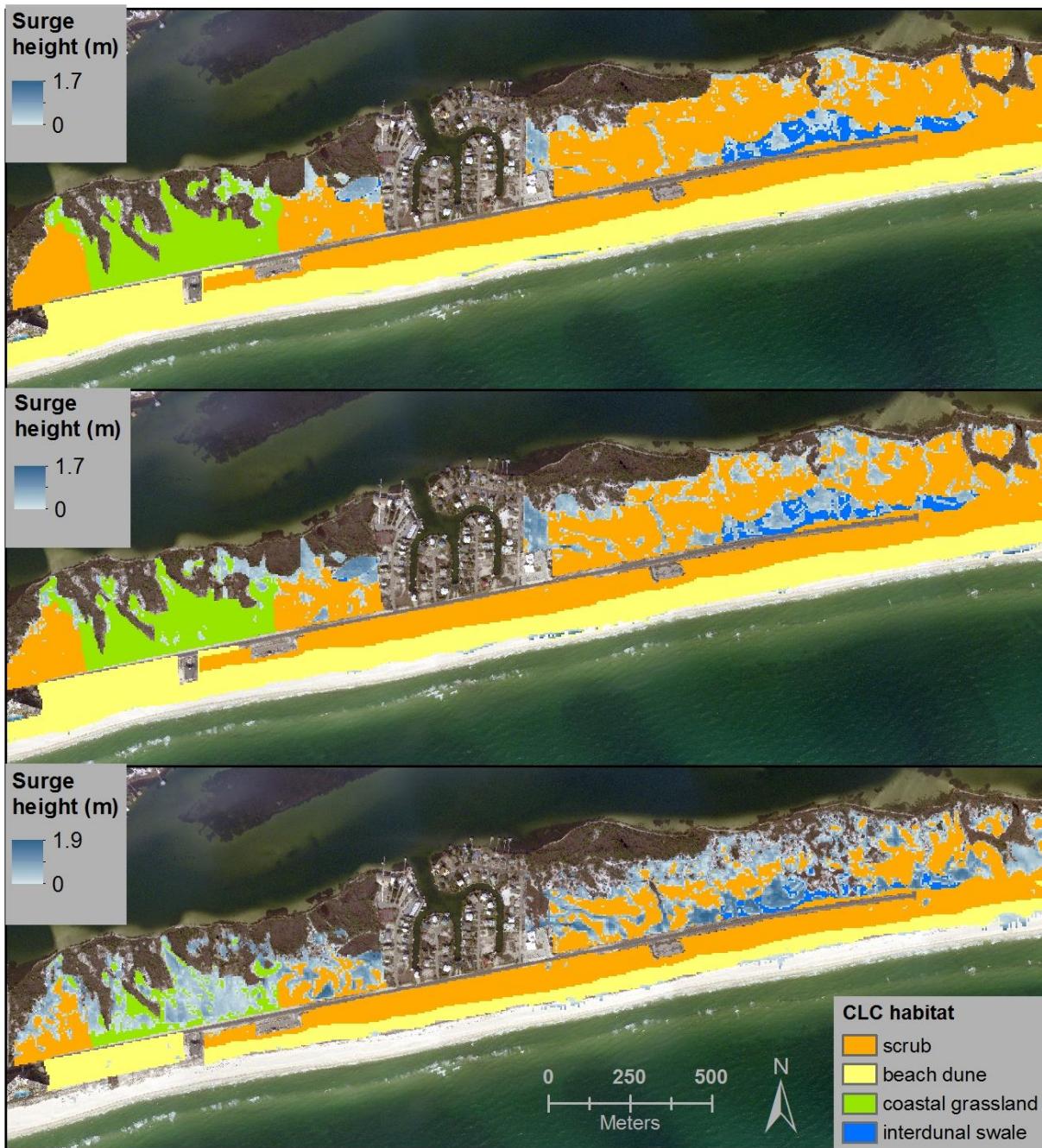


Figure E-1. Beach mouse habitat remaining in 2025 (top), 2050 (middle), and 2100 (bottom) after sea level rise at a **rate of 1 m** (by 2100) and a **Category 1 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.3 km, Figure 3-2B). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

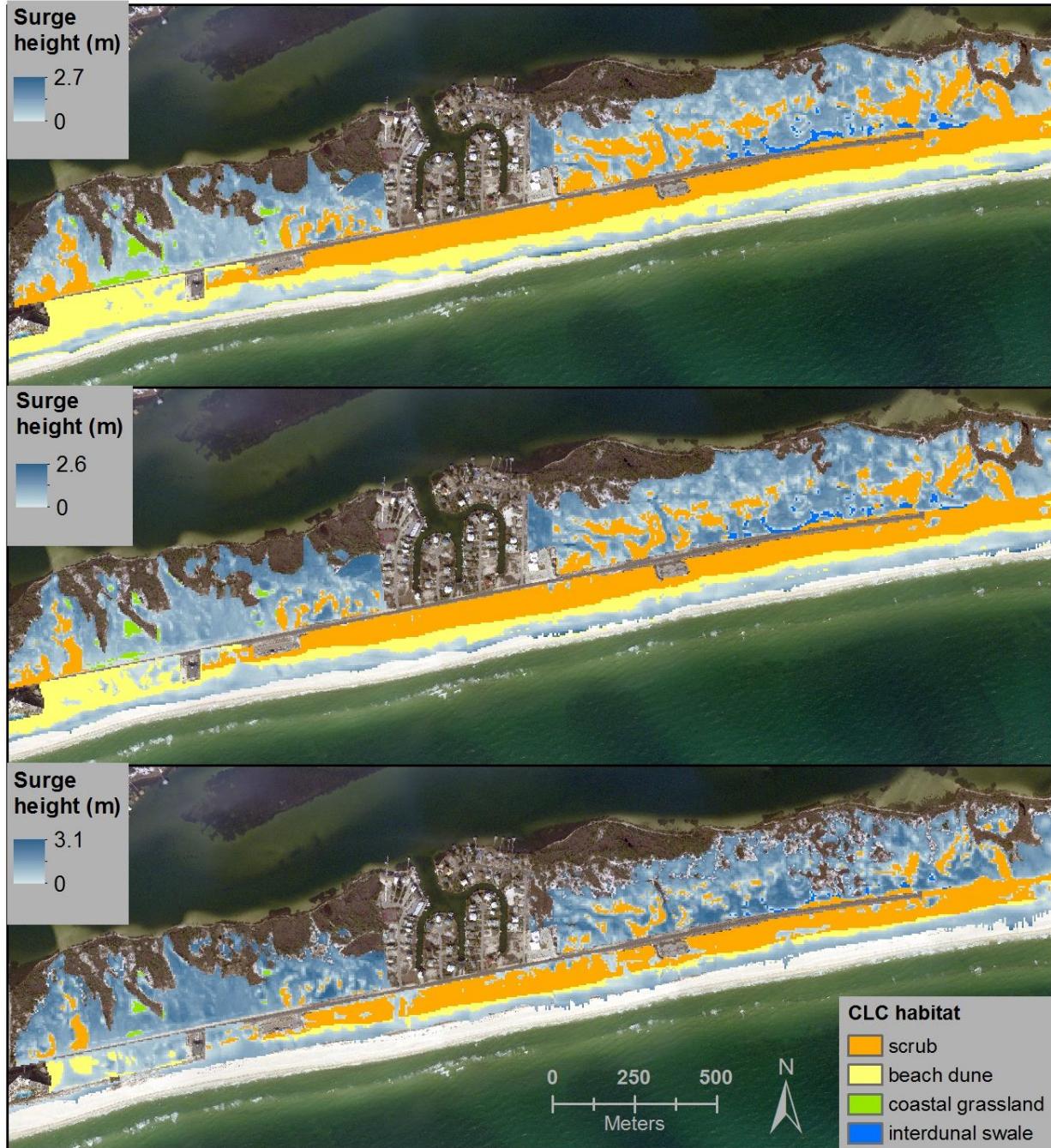


Figure E-2. Beach mouse habitat remaining in **2025** (top), **2050** (middle), and **2100** (bottom) after sea level rise at a **rate of 1 m** (by 2100) and a **Category 2 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.3 km, Figure 3-2B). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

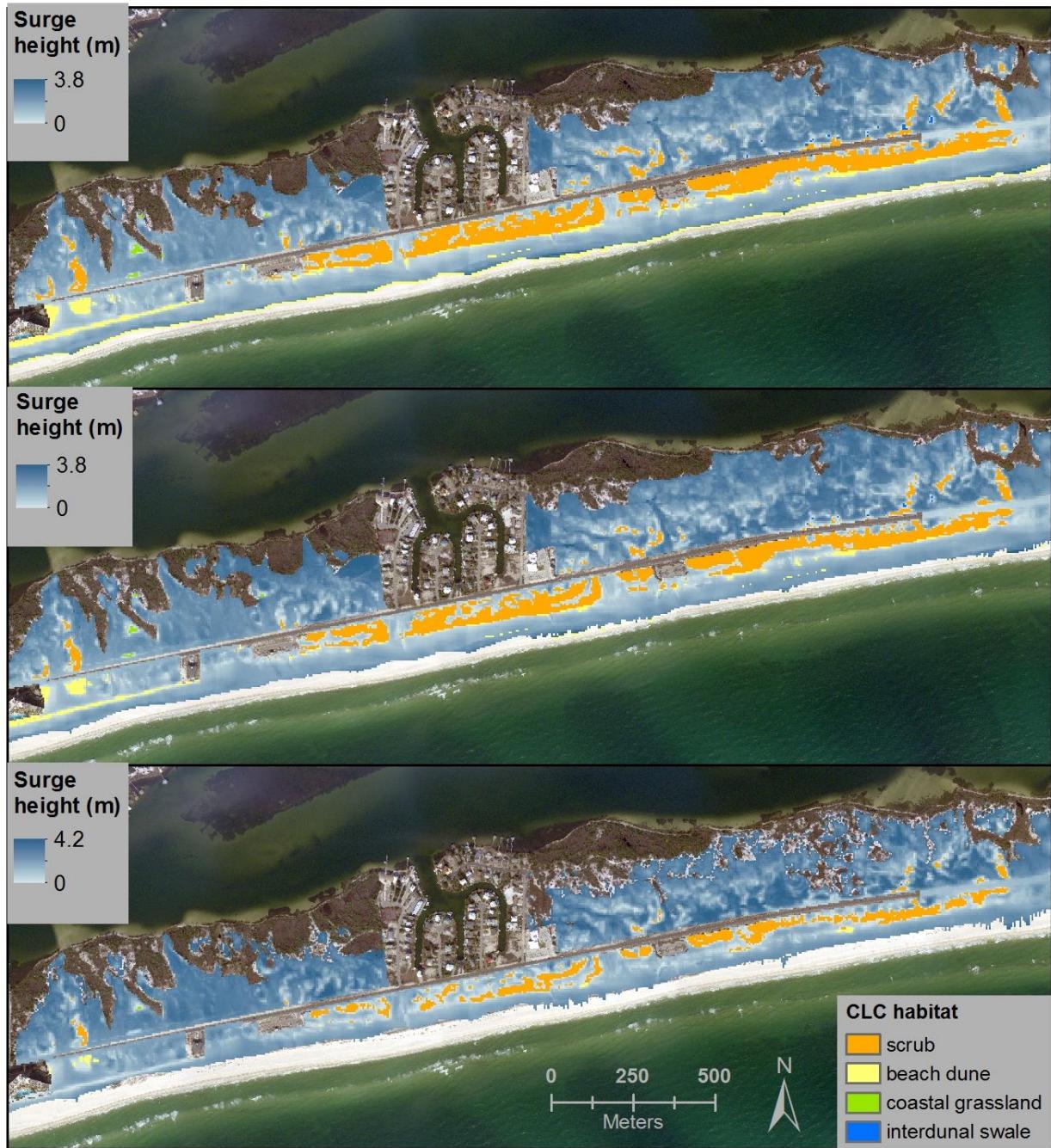


Figure E-3. Beach mouse habitat remaining in 2025 (top), 2050 (middle), and 2100 (bottom) after sea level rise at a **rate of 1 m** (by 2100) and a **Category 3 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.3 km, Figure 3-2B). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

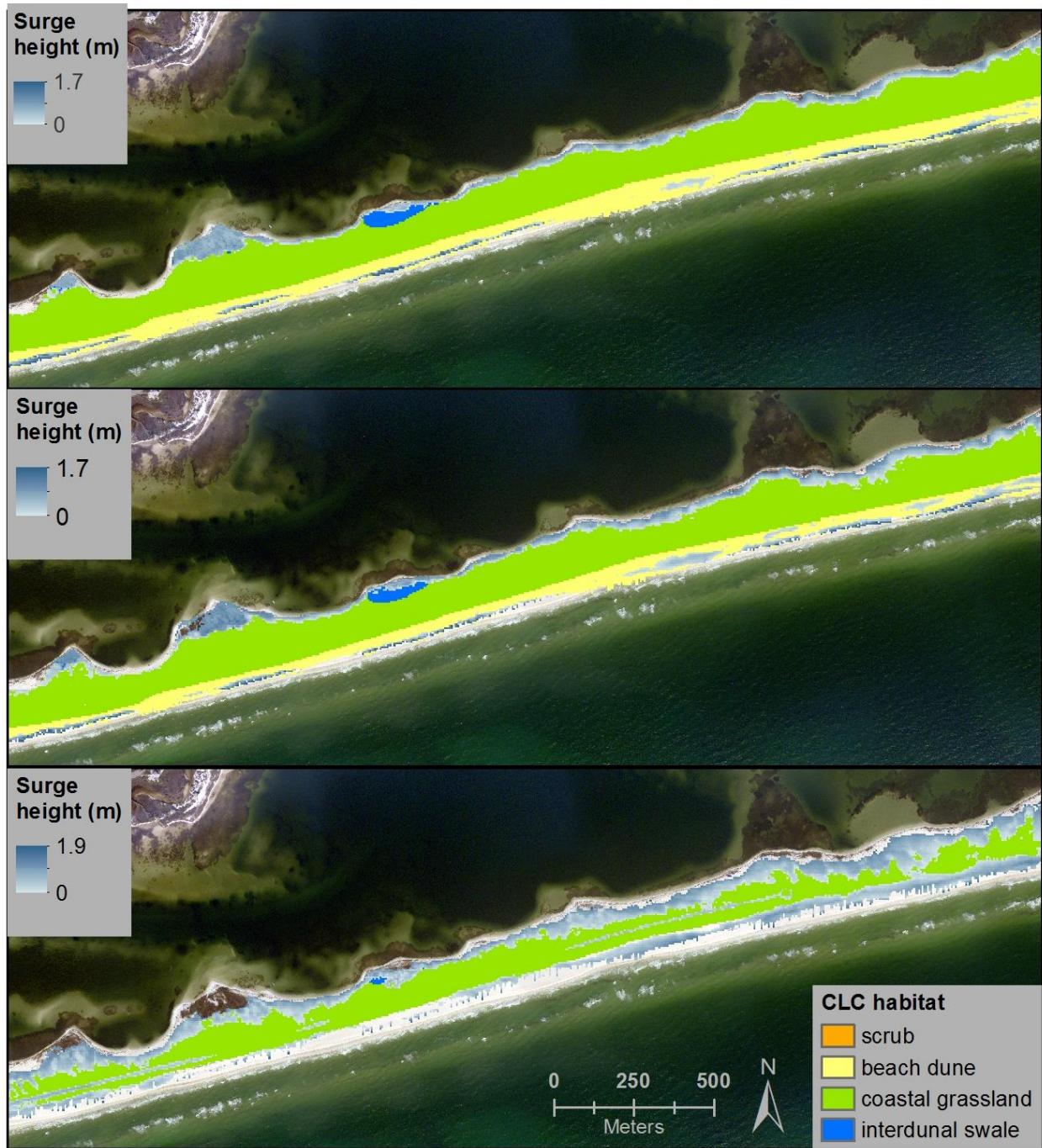


Figure E-4. Beach mouse habitat remaining in **2025 (top)**, **2050 (middle)**, and **2100 (bottom)** after sea level rise at a **rate of 1 m** (by 2100) and a **Category 1 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.4 km, Figure 3-2C). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

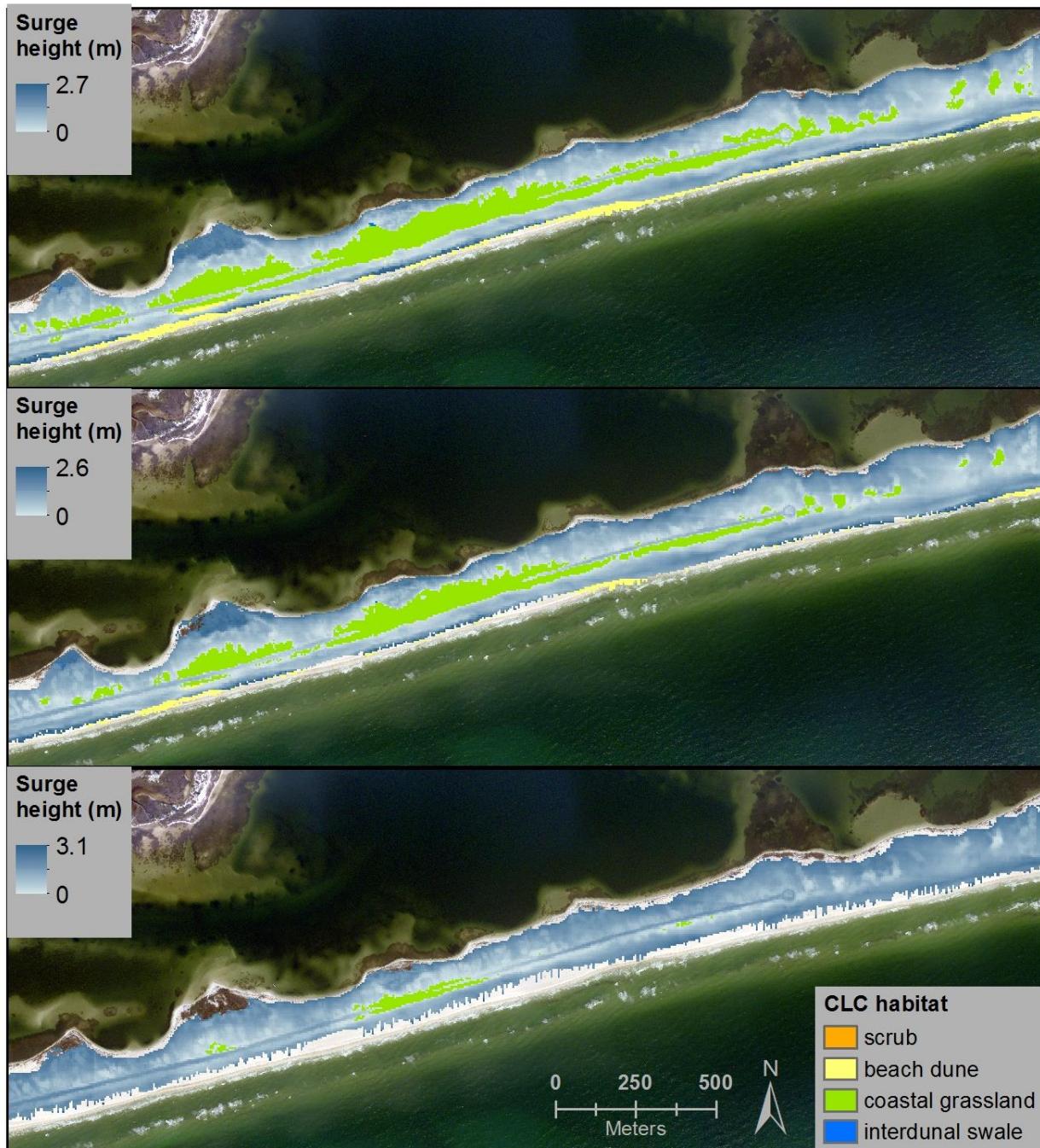


Figure E-5. Beach mouse habitat remaining in **2025 (top), 2050 (middle), and 2100 (bottom)** after sea level rise at a **rate of 1 m** (by 2100) and a **Category 2 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.4 km, Figure 3-2C). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

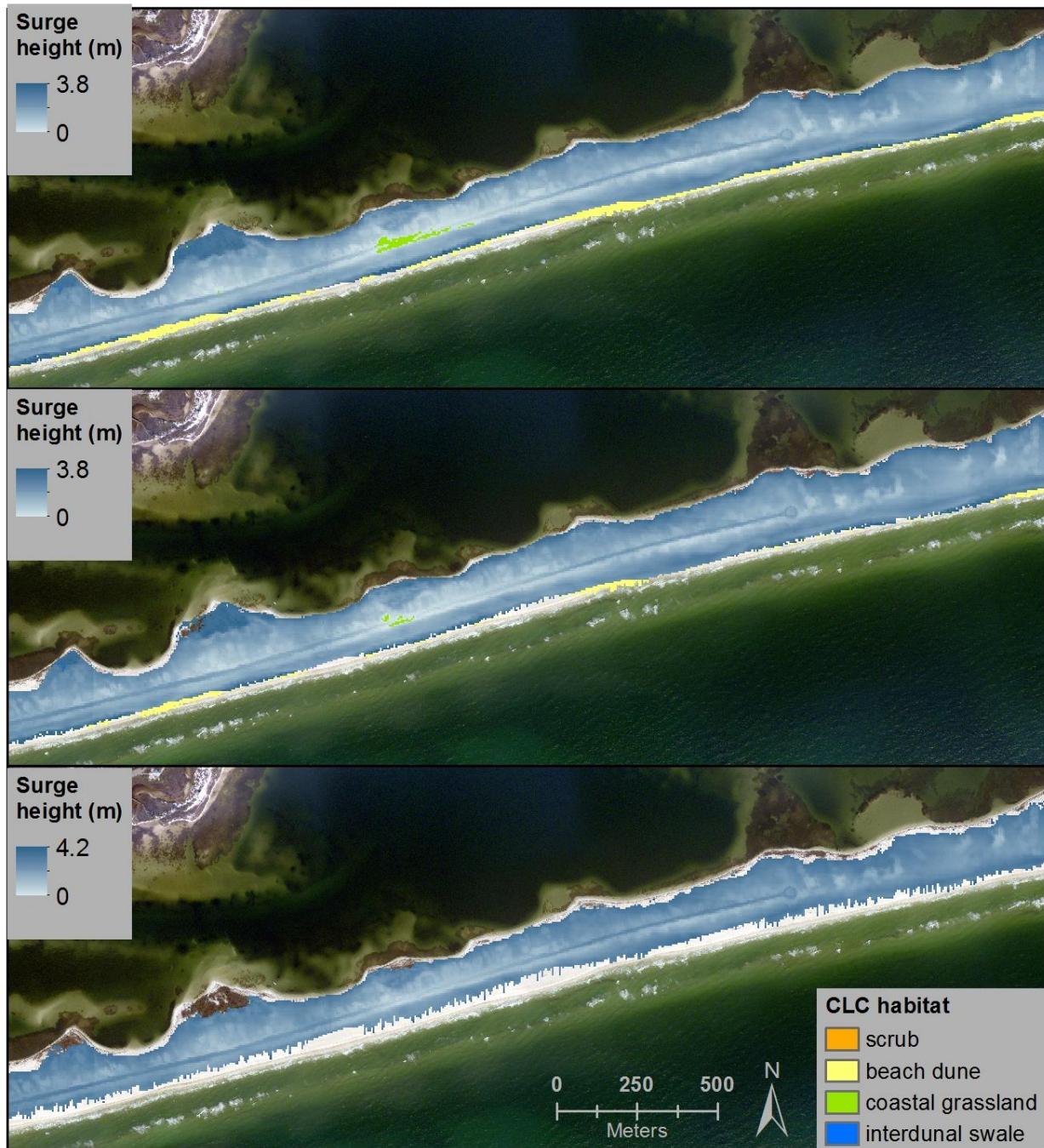


Figure E-6. Beach mouse habitat remaining in **2025 (top)**, **2050 (middle)**, and **2100 (bottom)** after sea level rise at a **rate of 1 m** (by 2100) and a **Category 3 hurricane**. Habitat is represented by CLC cover classes (Table 3-1) in part of Gulf Islands National Seashore (~3.4 km, Figure 3-2C). Areas where the background image (NOAA 2017c) shows through are not classified as beach mouse habitat; have changed to non-habitat, or have been lost to sea level rise (inundation).

APPENDIX F
AREA OF EACH BEACH MOUSE HABITAT TYPE AFFECTED BY HURRICANE SURGE

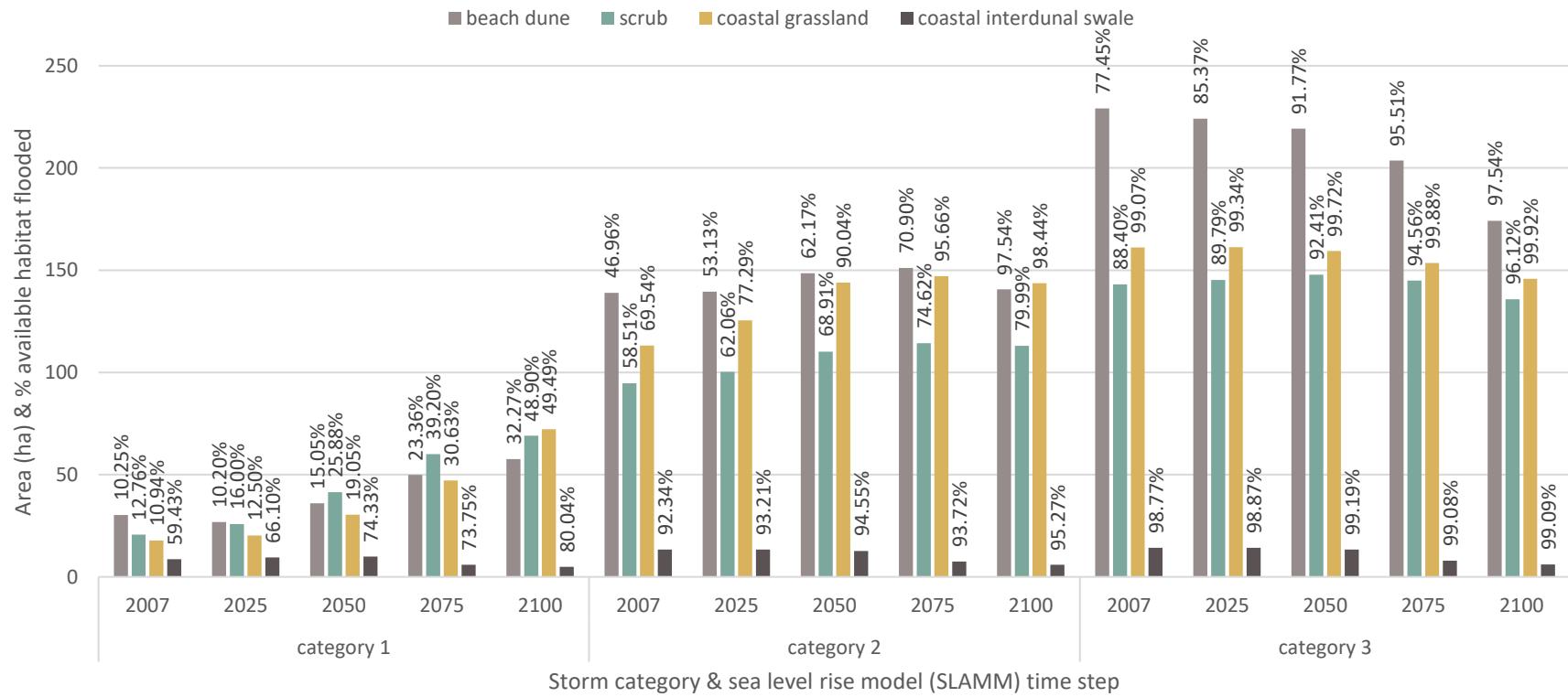


Figure F-1. Area (ha) of different beach mouse habitat types (CLC) on Perdido Key flooded by Category 1, 2, and 3 hurricanes after taking into account projected sea level rise (1-m rate). The number above each bar shows the percentage of habitat expected to be flooded under each hurricane category and sea level rise model (SLAMM) year time step. Flooding was estimated using the SLOSH hurricane surge model. Results are based on an overlay of **Cooperative Land Cover (CLC) habitat categories** (Table 3-1) on SLAMM results.

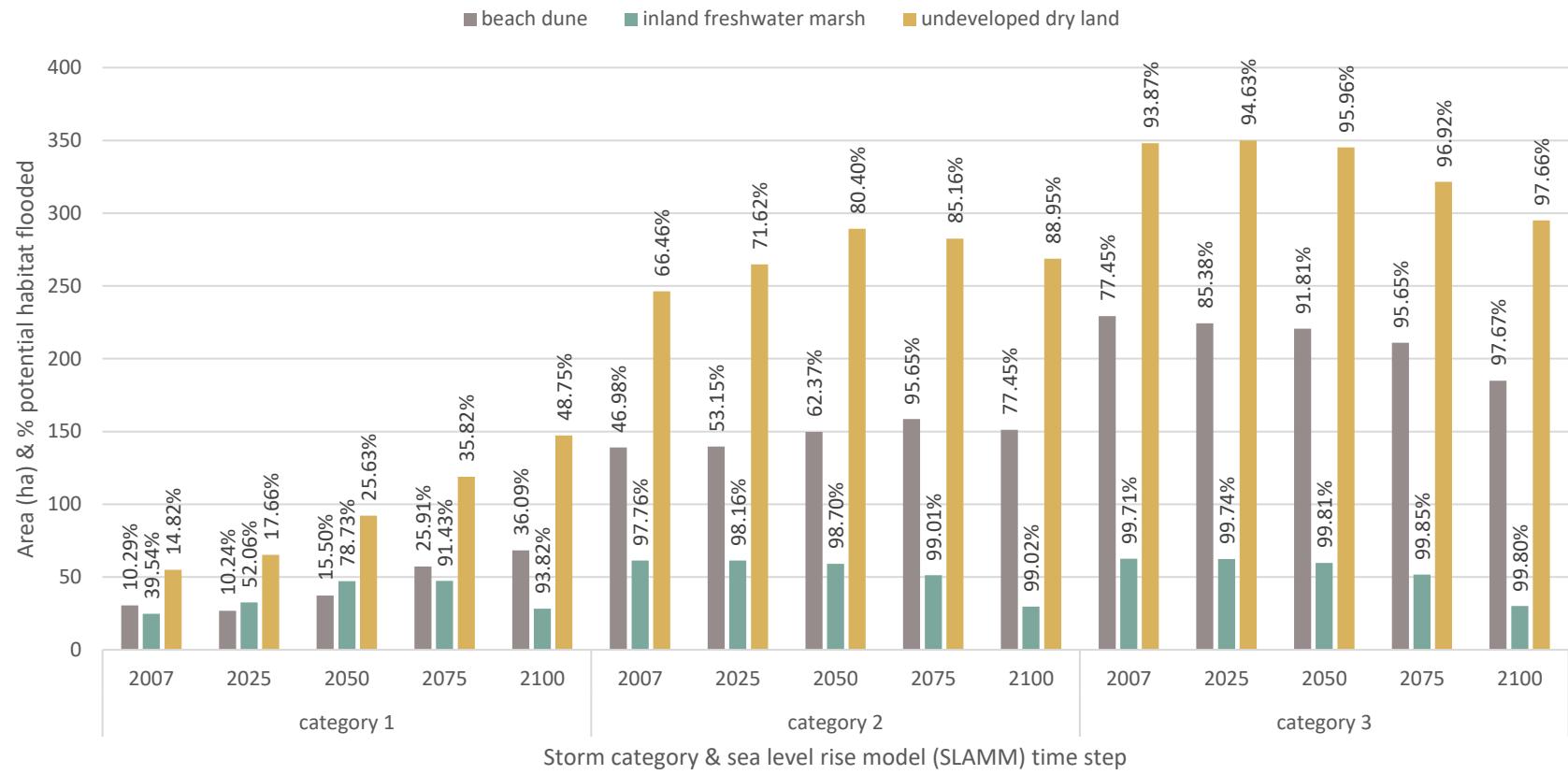


Figure F-2. Area (ha) of different beach mouse habitat types (SLAMM) on Perdido Key flooded by Category 1, 2, and 3 hurricanes after taking into account projected sea level rise (1-m rate). The number above each bar shows the percentage of habitat expected to be flooded under each hurricane category and sea level rise model (SLAMM) year time step. Flooding was estimated using the SLOSH hurricane surge model. Results are based on **SLAMM habitat categories** (Table 3-1).

LIST OF REFERENCES

- Abramsky, Z., M. L. Rosenzweig, and A. Subach. 1997. Gerbils under threat of owl predation: isoclines and isodars. *Oikos* **78**:81-90.
- Abramsky, Z., M. L. Rosenzweig, and A. Subach. 2002. The costs of apprehensive foraging. *Ecology* **83**:1330-1340.
- Aiello-Lammens, M. E., M. L. Chu-Agor, M. Convertino, R. A. Fischer, I. Linkov, and H. Resit Akçakaya. 2011. The impact of sea-level rise on Snowy Plovers in Florida: integrating geomorphological, habitat, and metapopulation models. *Global Change Biology* **17**:3644-3654.
- Arthur, A. D., R. P. Pech, and C. R. Dickman. 2005. Effects of predation and habitat structure on the population dynamics of house mice in large outdoor enclosures. *Oikos* **108**:562-572.
- Austin, J. D., J. A. Gore, D. U. Greene, and C. Gotteland. 2015. Conspicuous genetic structure belies recent dispersal in an endangered beach mouse (*Peromyscus polionotus trissyllepsis*). *Conservation Genetics* **16**:915-928.
- Bakker, V. J. and D. H. Van Vuren. 2004. Gap-crossing decisions by the red squirrel, a forest-dependent small mammal. *Conservation Biology* **18**:689-697.
- Baskaran, L. M., V. H. Dale, R. A. Efroymson, and W. Birkhead. 2006. Habitat modeling within a regional context: An example using gopher tortoise. *The American Midland Naturalist* **155**:335-351.
- Belisle, M. 2005. Measuring landscape connectivity: The challenge of behavioral landscape ecology. *Ecology* **86**:1988-1995.
- Bender, M. A., T. R. Knutson, R. E. Tuleya, J. J. Sirutis, G. A. Vecchi, S. T. Garner, and I. M. Held. 2010. Modeled impact of anthropogenic warming on the frequency of intense Atlantic hurricanes. *Science* **327**:454-458.
- Benscoter, A. M., J. S. Reece, R. F. Noss, L. A. Brandt, F. J. Mazzotti, S. S. Romanach, and J. I. Watling. 2013. Threatened and endangered subspecies with vulnerable ecological traits also have high susceptibility to sea level rise and habitat fragmentation. *PLoS ONE* **8**, :1-10, <https://doi.org/10.1371/journal.pone.0070647>.
- Berger-Tal, O., S. Mukherjee, B. P. Kotler, and J. S. Brown. 2010. Complex state-dependent games between owls and gerbils. *Ecology Letters* **13**:302-310.
- Beyer, H. L. 2004. Hawth's Analysis Tools for ArcGIS. <http://www.spatialecology.com/htool>.
- Bird, B. L., L. C. Branch, and D. L. Miller. 2004. Effects of coastal lighting on foraging behavior of beach mice. *Conservation Biology* **18**:1435-1439.

- Branch, L., D. Miller, and E. Wilkinson. 2011. Habitat restoration for beach mice: landscape-level population studies and dune restoration. Final Report submitted to the U.S. Fish and Wildlife Service, Panama City, FL.
- Brinkerhoff, R. J., N. M. Haddad, and J. L. Orrock. 2005. Corridors and olfactory predator cues affect small mammal behavior. *Journal of Mammalogy* **86**:662-669.
- Brown, J. S. 1988. Patch use as an indicator of habitat preference, predation risk, and competition. *Behavioral Ecology and Sociobiology* **22**:37-47.
- Brown, J. S. 1999. Vigilance, patch use and habitat selection: Foraging under predation risk. *Evolutionary Ecology Research* **1**:49-71.
- Brown, J. S. and B. P. Kotler. 2004. Hazardous duty pay and the foraging cost of predation. *Ecology Letters* **7**:999-1014.
- Chen, Q., H. Wang, L. Wang, R. Tawes, and D. Rollman. 2014. Predicting impacts of tropical cyclones and sea-level rise on beach mouse habitat. *Journal of Coastal Research Special Issue*:12-19.
- Chu-Agor, M. L., R. Muñoz-Carpena, G. Kiker, A. Emanuelsson, and I. Linkov. 2011. Exploring vulnerability of coastal habitats to sea level rise through global sensitivity and uncertainty analyses. *Environmental Modelling & Software* **26**:593-604.
- Chu-Agor, M. L., R. Muñoz-Carpena, G. A. Kiker, M. E. Aiello-Lammens, H. R. Akçakaya, M. Convertino, and I. Linkov. 2012. Simulating the fate of Florida Snowy Plovers with sea-level rise: Exploring research and management priorities with a global uncertainty and sensitivity analysis perspective. *Ecological Modelling* **224**:33-47.
- Claudino-Sales, V., P. Wang, and M. H. Horwitz. 2010. Effect of Hurricane Ivan on coastal dunes of Santa Rosa barrier Island, Florida: Characterized on the basis of pre- and poststorm LIDAR surveys. *Journal of Coastal Research* **26**:470-484.
- Clough, J., R. Park, M. Propato, A. Polaczyk, and R. Fuller. 2012. SLAMM 6.2 Technical Documentation. Warren Pinnacle Consulting, Inc.
- Convertino, M., P. Welle, R. Muñoz-Carpena, G. A. Kiker, M. L. Chu-Agor, R. A. Fischer, and I. Linkov. 2012. Epistemic uncertainty in predicting shorebird biogeography affected by sea-level rise. *Ecological Modelling* **240**:1-15.
- Dahl, K. A., M. F. Fitzpatrick, and E. Spanger-Siegfried. 2017. Sea level rise drives increased tidal flooding frequency at tide gauges along the U.S. East and Gulf Coasts: Projections for 2030 and 2045. *PLoS ONE* **12**:1-23, <https://doi.org/10.1371/journal.pone.0170949>.
- Dai, H., M. Ye, and A. W. Niedoroda. 2015. A model for simulating barrier island geomorphologic responses to future storm and sea-level rise impacts. *Journal of Coastal Research* **31**:1091-1102.

- Defeo, O., A. McLachlan, D. S. Schoeman, T. A. Schlacher, J. Dugan, A. Jones, M. Lastra, and F. Scapini. 2009. Threats to sandy beach ecosystems: A review. *Estuarine Coastal and Shelf Science* **81**:1-12.
- Dugan, J. E., O. Defeo, E. Jaramillo, A. R. Jones, M. Lastra, R. Nel, C. H. Peterson, F. Scapini, T. Schlacher, and D. S. Schoeman. 2010. Give beach ecosystems their day in the sun. *Science* **329**:1146-1146.
- Ewers, R. M. and R. K. Didham. 2006. Confounding factors in the detection of species responses to habitat fragmentation. *Biological Reviews* **81**:117-142.
- Falcy, M. R. and B. J. Danielson. 2014. Post-hurricane recovery and long-term viability of the Alabama beach mouse. *Biological Conservation* **178**:28-36.
- Fanson, B. G. 2010. Effect of direct and indirect cues of predation risk on the foraging behavior of the white-footed mouse (*Peromyscus leucopus*). *Northeastern Naturalist* **17**:19-28.
- Farbotko, C. 2010. Wishful sinking: Disappearing islands, climate refugees and cosmopolitan experimentation. *Asia Pacific Viewpoint* **51**:47-60.
- Feagin, R. A., D. J. Sherman, and W. E. Grant. 2005. Coastal erosion, global sea-level rise, and the loss of sand dune plant habitats. *Frontiers in Ecology and the Environment* **3**:359-364.
- Fielding, A. H. and J. F. Bell 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* **24**:38-49.
- Fish, M. R., I. M. Cote, J. A. Gill, A. P. Jones, S. Renshoff, and A. R. Watkinson. 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. *Conservation Biology* **19**:482-491.
- Fitzgerald, M., R. Shine, and F. Lemckert. 2002. Spatial ecology of arboreal snakes (*Hoplocephalus stephensi*, Elapidae) in an eastern Australian forest. *Austral Ecology* **27**:537-545.
- FNAI. 2010. Guide to the natural communities of Florida: 2010 edition. Florida Natural Areas Inventory, Tallahassee, FL.
- Forbes, C., R. A. L. Jr., C. A. Mattocks, and J. J. Westerink. 2010. A Retrospective Evaluation of the Storm Surge Produced by Hurricane Gustav (2008): Forecast and Hindcast Results. *Weather and Forecasting* **25**:1577-1602.
- Forbes, C., J. Rhome, C. Mattocks, and A. Taylor. 2014. Predicting the storm surge threat of Hurricane Sandy with the National Weather Service SLOSH model. *Journal of Marine Science and Engineering* **2**:437.
- Geselbracht, L., K. Freeman, E. Kelly, D. R. Gordon, and F. E. Putz. 2011. Retrospective and prospective model simulations of sea level rise impacts on Gulf of Mexico coastal marshes and forests in Waccasassa Bay, Florida. *Climatic Change* **107**:35-57.

- Geselbracht, L., K. Freeman, A. Birch, D. Gordon, A. Knight, M. O'Brien, and J. Oetting. 2013. Modeling and Abating the Impacts of Sea Level Rise on Five Significant Estuarine Systems in the Gulf of Mexico -- Final Report to the U.S. Environmental Protection Agency – Gulf of Mexico Program, Project # MX-95463410-2 The Nature Conservancy.
- Geselbracht, L. L., K. Freeman, A. P. Birch, J. Brenner, and D. R. Gordon. 2015. Modeled sea level rise impacts on coastal ecosystems at six major estuaries on Florida's Gulf Coast: Implications for adaptation planning. *PLoS ONE* **10**:1-28, <https://doi.org/10.1371/journal.pone.0132079>.
- Gill, F. B. 1988. Trapline foraging by hermit hummingbirds: Competition for an undefended, renewable resource. *Ecology* **69**:1933-1942.
- Glahn, R., A. Taylor, N. Kurkowski, and W. A. Shaffer. 2009. The role of the SLOSH model in National Weather Service storm surge forecasting. *National Weather Digest* **33**:3-14.
- Goldenberg, S. B., C. W. Landsea, A. M. Mestas-Nunez, and W. M. Gray. 2001. The recent increase in Atlantic hurricane activity: Causes and implications. *Science* **293**:474-479.
- Gomez, N., J. X. Mitrovica, M. E. Tamisiea, and P. U. Clark. 2010. A new projection of sea level change in response to collapse of marine sectors of the Antarctic Ice Sheet. *Geophysical Journal International* **180**:623-634.
- Google Maps. 2017a. Map of Perdido Key. Accessed 6-15-2017 at <https://www.google.com/maps/place/Perdido+Key,+FL+32507/@30.3042587,-87.4856279,12z/data=!3m1!4b1!4m5!3m4!1s0x8890af9b7b75d4e9:0x7dd9d6b7296d17d6!8m2!3d30.3035037!4d-87.4273172>.
- Google Maps. 2017b. Map of Santa Rosa Island. Accessed 6-15-2017 at <https://www.google.com/maps/place/Santa+Rosa+Island/@30.359542,-87.1877582,10z/data=!3m1!4b1!4m5!3m4!1s0x889126dbc25240e5:0x5f7e5a93c7167a3e!8m2!3d30.3978561!4d-86.7290634>.
- Grafals-Soto, R. 2012. Effects of sand fences on coastal dune vegetation distribution. *Geomorphology* **145–146**:45-55.
- Greene, D. U., J. A. Gore, and J. D. Austin. 2017. Reintroduction of captive-born beach mice: the importance of demographic and genetic monitoring. *Journal of Mammalogy* **98**:513-522.
- Greene, D. U., J. A. Gore, and M. A. Stoddard. 2016. Reintroduction of the endangered Perdido Key beach mouse (*Peromyscus polionotus trissyllepsis*): fate and movements of captive-born animals. *Florida Scientist* **79**:1-13.
- Grinsted, A., J. C. Moore, and S. Jevrejeva. 2013. Projected Atlantic hurricane surge threat from rising temperatures. *Proceedings of the National Academy of Sciences* **110**:5369-5373.

- Haddad, N. M., D. R. Bowne, A. Cunningham, B. J. Danielson, D. J. Levey, S. Sargent, and T. Spira. 2003. Corridor use by diverse taxa. *Ecology* **84**:609-615.
- Harewood, A. and J. Horrocks. 2008. Impacts of coastal development on hawksbill hatchling survival and swimming success during the initial offshore migration. *Biological Conservation* **141**:394-401.
- Herman, C. S. and T. J. Valone. 2000. The effect of mammalian predator scent on the foraging behavior of *Dipodomys merriami*. *Oikos* **91**:139-145.
- Hesp, P. A. 1991. Ecological processes and plant adaptations on coastal dunes. *Journal of Arid Environments* **21**:165-191.
- Hesp, P. A. and M. J. Hilton. 2016. Restoration of foredunes and transgressive dunefields: Case studies from New Zealand. Pages 67-92 in M. L. Martinez, J. B. Gallego-Fernández, and P. A. Hesp, editors. *Restoration of Coastal Dunes*. Springer-Verlag, New York.
- Houser, C. and S. Hamilton. 2009. Sensitivity of post-hurricane beach and dune recovery to event frequency. *Earth Surface Processes and Landforms* **34**:613-628.
- Houser, C., C. Hapke, and S. Hamilton. 2008. Controls on coastal dune morphology, shoreline erosion and barrier island response to extreme storms. *Geomorphology* **100**:223-240.
- Houser, C., P. Wernet, E. Rentschlar, H. Jones, B. Hammond, and S. Trimble. 2015. Post-storm beach and dune recovery: Implications for barrier island resilience. *Geomorphology* **234**:54-63.
- Houston, S. H., W. A. Shaffer, M. D. Powell, and J. Chen. 1999. Comparisons of HRD and SLOSH surface wind fields in hurricanes: Implications for storm surge modeling. *Weather and Forecasting* **14**:671-686.
- Hufbauer, R. A., M. Szűcs, E. Kasyon, C. Youngberg, M. J. Koontz, C. Richards, T. Tuff, and B. A. Melbourne. 2015. Three types of rescue can avert extinction in a changing environment. *Proceedings of the National Academy of Sciences* **112**:10557-10562.
- Humphrey, S. R. and D. B. Barbour. 1981. Status and habitat of three subspecies of *Peromyscus polionotus* in Florida. *Journal of Mammalogy* **62**:840-844.
- IPCC. 2007. Summary for Policymakers. In: *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* in S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, editors. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- IPCC. 2013. *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.,

- IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.
- Jacob, J. and J. S. Brown. 2000. Microhabitat use, giving-up densities and temporal activity as short- and long-term anti-predator behaviors in common voles. *Oikos* **91**:131-138.
- Jelesnianski, C. P., J. Chen, and W. A. Shaffer. 1992. SLOSH: Sea, lake, and overland surges from hurricanes NOAA Technical Report NWS 48, National Oceanic and Atmospheric Administration, U. S. Department of Commerce, 71 pp.
- Jevrejeva, S., J. C. Moore, and A. Grinsted. 2010. How will sea level respond to changes in natural and anthropogenic forcings by 2100? *Geophys. Res. Lett.* **37**:L07703.
- Knight, G. R. and A. Knight. 2010. Development of a Cooperative Land Cover Map: Final Report. Florida's Wildlife Legacy Initiative Project 08009.
- Knutson, T. R., J. L. McBride, J. Chan, K. Emanuel, G. Holland, C. Landsea, I. Held, J. P. Kossin, A. K. Srivastava, and M. Sugi. 2010. Tropical cyclones and climate change. *Nature Geoscience* **3**:157-163.
- Kotler, B. P. and L. Blaustein. 1995. Titrating food and safety in a heterogeneous environment: When are the risky and safe patches of equal value? *Oikos* **74**:251-258.
- Kotler, B. P., J. Brown, S. Mukherjee, O. Berger-Tal, and A. Bouskila. 2010. Moonlight avoidance in gerbils reveals a sophisticated interplay among time allocation, vigilance and state-dependent foraging. *Proceedings of the Royal Society B-Biological Sciences* **277**:1469-1474.
- Kutiel, P., Y. Peled, and E. Geffen. 2000. The effect of removing shrub cover on annual plants and small mammals in a coastal sand dune ecosystem. *Biological Conservation* **94**:235-242.
- LaFever, D. H., R. R. Lopez, R. A. Feagin, and N. J. Silvy. 2007. Predicting the impacts of future sea-level rise on an endangered Lagomorph. *Environmental Management* **40**:430-437.
- Landis, J. R. and G. G. Koch. 1977. The measurement of observer agreement for categorical data. *Biometrics* **33**:159-174.
- Lau, A. and C. K. Dodd. 2013. Multiscale burrow site selection of gopher tortoises (*Gopherus polyphemus*) in coastal sand dune habitat. *Journal of Coastal Research*:305-314.
- Lentz, E. E., C. J. Hapke, H. F. Stockdon, and R. E. Hehre. 2013. Improving understanding of near-term barrier island evolution through multi-decadal assessment of morphologic change. *Marine Geology* **337**:125-139.

- Lima, S. L. and L. M. Dill. 1990. Behavioral decisions made under the risk of predation--A review and prospectus. Canadian Journal of Zoology-Revue Canadienne De Zoologie **68**:619-640.
- Lindemer, C. A., N. G. Plant, J. A. Puleo, D. M. Thompson, and T. V. Wamsley. 2010. Numerical simulation of a low-lying barrier island's morphological response to Hurricane Katrina. Coastal Engineering **57**:985-995.
- Lithgow, D., M. L. Martínez, J. B. Gallego-Fernández, P. A. Hesp, P. Flores, S. Gachuz, N. Rodríguez-Revelo, O. Jiménez-Orocio, G. Mendoza-González, and L. L. Álvarez-Molina. 2013. Linking restoration ecology with coastal dune restoration. Geomorphology **199**:214-224.
- Loggins, R. E., J. A. Gore, L. L. Brown, L. A. Slaby, and E. H. Leone. 2010. A modified track tube for detecting beach mice. Journal of Wildlife Management **74**:1154-1159.
- Longcore, T. and C. Rich. 2004. Ecological light pollution. Frontiers in Ecology and the Environment **2**:191-198.
- Lynn, W. J. 2000. Social organization and burrow site selection of the Alabama beach mouse (*Peromyscus polionotus ammobates*). MS Thesis. Auburn University, Auburn, AL.
- Mandelik, Y., M. Jones, and T. Dayan. 2003. Structurally complex habitat and sensory adaptations mediate the behavioural responses of a desert rodent to an indirect cue for increased predation risk. Evolutionary Ecology Research **5**:501-515.
- Masetti, R., S. Fagherazzi, and A. Montanari. 2008. Application of a barrier island translation model to the millennial-scale evolution of Sand Key, Florida. Continental Shelf Research **28**:1116-1126.
- Maslo, B., S. N. Handel, and T. Pover. 2011. Restoring beaches for Atlantic Coast piping plovers (*Charadrius melanotos*): A classification and regression tree analysis of nest-site selection. Restoration Ecology **19**:194-203.
- McAlpine, C., C. P. Catterall, R. Mac Nally, D. Lindenmayer, J. L. Reid, K. D. Holl, A. F. Bennett, R. K. Runting, K. Wilson, R. J. Hobbs, L. Seabrook, S. Cunningham, A. Moilanen, M. Maron, L. Shoo, I. Lunt, P. Veski, L. Rumpff, T. G. Martin, J. Thomson, and H. Possingham. 2016. Integrating plant- and animal-based perspectives for more effective restoration of biodiversity. Frontiers in Ecology and the Environment **14**:37-45.
- Miller, D. L., L. C. Branch, M. Thetford, and M. A. Stoddard. 2016. Endangered beach mouse: Linking population studies/habitat restoration to predicted sea level rise. Final report submitted to the National Park Service. National Park Service.
- Miller, D. L., M. Thetford, J. Dupree, and L. Atwood. 2014. Influence of seasonal changes and shifting substrate on survival of restoration plantings of *Schizachyrium maritimum* (gulf bluestem) on Santa Rosa Island, Florida. Journal of Coastal Research **30**:237-247.

- Miller, D. L., M. Thetford, and L. Yager. 2001. Evaluation of sand fence and vegetation for dune building following overwash by Hurricane Opal on Santa Rosa Island, Florida. *Journal of Coastal Research* **17**:936-948.
- Moore, L. J., J. H. List, S. J. Williams, and D. Stolper. 2010. Complexities in barrier island response to sea level rise: Insights from numerical model experiments, North Carolina Outer Banks. *Journal of Geophysical Research-Earth Surface* **115**:F03004, doi:03010.01029/02009JF001299.
- Morris, D. W. 1997. Optimally foraging deer mice in prairie mosaics: a test of habitat theory and absence of landscape effects. *Oikos* **80**:31-42.
- Morton, R. A. and A. H. Sallenger. 2003. Morphological impacts of extreme storms on sandy beaches and barriers. *Journal of Coastal Research* **19**:560-573.
- Nicholls, R. J. 2011. Planning for the impacts of sea level rise. *Oceanography* **24**:144-157.
- Nicholls, R. J., N. Marinova, J. A. Lowe, S. Brown, P. Vellinga, D. de Gusmão, J. Hinkel, and R. S. J. Tol. 2011. Sea-level rise and its possible impacts given a ‘beyond 4°C world’ in the twenty-first century. *Phil. Trans. R. Soc. A* **369**:161-181.
- NOAA. 2017a. 2006 Southeast Pre-Baseline Natural Color 8 Bit Imagery. NOAA Data Access Viewer. Accessed 06-15-2017 at <https://coast.noaa.gov/dataviewer/#/imagery/search/-9747020.53584082,3537047.0468682456,-9630377.63067764,3557226.422335532/details/349>.
- NOAA. 2017b. 2010 Pensacola NOAA NGS DSS Natural Color 8 Bit Imagery. NOAA Data Access Viewer. Accessed 06-15-2017 at <https://coast.noaa.gov/dataviewer/#/imagery/search/-9747020.53584082,3537047.0468682456,-9630377.63067764,3557226.422335532/details/415>.
- NOAA. 2017c. 2014 Perdido Bay AL MLLW NOAA NGS DSS Infrared 8 Bit Imagery. NOAA Data Access Viewer. Accessed 06-15-2017 at <https://coast.noaa.gov/dataviewer/#/imagery/search/-9747020.53584082,3537047.0468682456,-9630377.63067764,3557226.422335532/details/4761>.
- NOAA. 2017d. Historical Hurricane Tracks Tool. National Oceanic and Atmospheric Administration. Accessed 06-01-2017 at <https://coast.noaa.gov/hurricanes/>.
- NOAA. 2017e. National Weather Service SLOSH Pages. Accessed 04-01-2017 at <http://140.90.8.37/sloshPub/>.
- Oli, M. K., N. R. Holler, and M. C. Wooten. 2001. Viability analysis of endangered Gulf Coast beach mice (*Peromyscus polionotus*) populations. *Biological Conservation* **97**:107-118.

- Orrock, J. L. 2010. When the ghost of predation has passed: Do rodents from islands with and without fox predators exhibit aversion to fox cues? *Ethology* **116**:338-345.
- Orrock, J. L. and B. J. Danielson. 2005. Patch shape, connectivity, and foraging by oldfield mice (*Peromyscus polionotus*). *Journal of Mammalogy* **86**:569-575.
- Orrock, J. L., B. J. Danielson, and R. J. Brinkerhoff. 2004. Rodent foraging is affected by indirect, but not by direct, cues of predation risk. *Behavioral Ecology* **15**:433-437.
- Orrock, J. L. and R. J. Fletcher. 2014. An island-wide predator manipulation reveals immediate and long-lasting matching of risk by prey. *Proceedings of the Royal Society B-Biological Sciences* **281**:20140391.
- Otvos, E. G. and G. A. Carter. 2008. Hurricane degradation-barrier development cycles, northeastern Gulf of Mexico: Landform evolution and island chain history. *Journal of Coastal Research* **24**:463-478.
- Park, R., T. Armentano, and C. Cloonan. 1986. Predicting the impact of sea level rise on coastal systems. Pages 149-153 in Supplementary proceedings for the 1986 Eastern simulation conference, Norfolk, Virginia.
- Pearce, J. L. and S. Ferrier. 2000. Evaluating the predictive performance of habitat models developed using logistic regression. *Ecological Modelling* **133**:225-245.
- Pfeffer, W. T., J. T. Harper, and S. O'Neil. 2008. Kinematic constraints on glacier contributions to 21st-century sea-level rise. *Science* **321**:1340-1343.
- Pontee, N. 2013. Defining coastal squeeze: A discussion. *Ocean & Coastal Management* **84**:204-207.
- Pries, A. J., L. C. Branch, and D. L. Miller. 2009. Impact of hurricanes on habitat occupancy and spatial distribution of beach mice. *Journal of Mammalogy* **90**:841-850.
- Pries, A. J., D. L. Miller, and L. C. Branch. 2008. Identification of structural and spatial features that influence storm-related dune erosion along a barrier-island ecosystem in the Gulf of Mexico. *Journal of Coastal Research* **24**:168.
- Priestas, A. M. and S. Fagherazzi. 2010. Morphological barrier island changes and recovery of dunes after Hurricane Dennis, St. George Island, Florida. *Geomorphology* **114**:614-626.
- Prugh, L. R. and C. D. Golden. 2014. Does moonlight increase predation risk? Meta-analysis reveals divergent responses of nocturnal mammals to lunar cycles. *Journal of Animal Ecology* **83**:504-514.
- Rahmstorf, S. and D. Coumou. 2011. Increase of extreme events in a warming world. *Proceedings of the National Academy of Sciences of the United States of America* **108**:17905-17909.

- Roger, E., S. W. Laffan, and D. Ramp. 2007. Habitat selection by the common wombat (*Vombatus ursinus*) in disturbed environments: Implications for the conservation of a 'common' species. *Biological Conservation* **137**:437-449.
- Rolo, V., P. I. Olivier, and R. van Aarde. 2017. Tree and bird functional groups as indicators of recovery of regenerating subtropical coastal dune forests. *Restoration Ecology*:n/a-n/a.
- Russell, W., J. Shulzitski, and A. Setty. 2009. Evaluating wildlife response to coastal dune habitat restoration in San Francisco, California. *Ecological Restoration* **27**:439-448.
- SAS Institute, I. 2013. SAS® 9.4 Copyright © 2013. SAS Institute Inc., Cary, NC.
- Saunders, M. A. and A. S. Lea. 2008. Large contribution of sea surface warming to recent increase in Atlantic hurricane activity. *Nature* **451**:557-560.
- Schlacher, T. A., J. Dugan, D. S. Schoeman, M. Lastra, A. Jones, F. Scapini, A. McLachlan, and O. Defeo. 2007. Sandy beaches at the brink. *Diversity and Distributions* **13**:556-560.
- Schmidt, J. A., R. McCleery, J. R. Seavey, S. E. Cameron Devitt, and P. M. Schmidt. 2012. Impacts of a half century of sea-level rise and development on an endangered mammal. *Global Change Biology* **18**:3536-3542.
- Schrope, M. 2010. Unarrested development. *Nature Reports: Climate Change* **4**:36-38.
- Schupp, C. A., N. T. Winn, T. L. Pearl, J. P. Kumer, T. J. B. Carruthers, and C. S. Zimmerman. 2013. Restoration of overwash processes creates piping plover (*Charadrius melanotos*) habitat on a barrier island (Assateague Island, Maryland). *Estuarine, Coastal and Shelf Science* **116**:11-20.
- Seavey, J. R., B. Gilmer, and K. M. McGarigal. 2010. Effect of sea-level rise on piping plover (*Charadrius melanotos*) breeding habitat. *Biological Conservation* **144**:393-401.
- Sims, S. A., J. R. Seavey, and C. G. Curtin. 2013. Room to move? Threatened shorebird habitat in the path of sea level rise-dynamic beaches, multiple users, and mixed ownership: a case study from Rhode Island, USA. *Journal of Coastal Conservation* **17**:339-350.
- Sneckenberger, S. 2001. Factors influencing habitat use by the Alabama beach mouse (*Peromyscus polionotus ammobates*). MS Thesis. Auburn University, Auburn, Alabama.
- Thomsen, S. K. and D. J. Green. 2016. Cascading effects of predation risk determine how marine predators become terrestrial prey on an oceanic island. *Ecology* **97**:3530-3537.
- Tischendorf, L., A. Grez, T. Zaviezo, and L. Fahrig. 2005. Mechanisms affecting population density in fragmented habitat. *Ecology and Society* **10**:7 (online), <https://www.ecologyandsociety.org/vol10/iss11/art17/>.
- USFWS. 1985. Determination of endangered status of three beach mice. Pages 23872-23889. *Federal Register*, Atlanta.

- USFWS. 2007. Perdido Key Beach Mouse (*Peromyscus polionotus trissyllepsis*), 5 Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service, Panama City, FL.
- Van Zant, J. L. and M. C. Wooten. 2003. Translocation of Choctawhatchee beach mice (*Peromyscus polionotus allophrys*): hard lessons learned. *Biological Conservation* **112**:405-413.
- Vermeer, M. and S. Rahmstorf. 2009. Global sea level linked to global temperature. *Proceedings of the National Academy of Sciences* **106**:21527-21532.
- Vitousek, S., P. L. Barnard, C. H. Fletcher, N. Frazer, L. Erikson, and C. D. Storlazzi. 2017. Doubling of coastal flooding frequency within decades due to sea-level rise. *Scientific Reports* **7**:1399 (online), <https://www.nature.com/articles/s41598-41017-01362-41597>.
- Whittaker, R. J. 1995. Disturbed island ecology. *Trends in Ecology & Evolution* **10**:421-425.
- Wilkinson, E. B., L. C. Branch, and D. L. Miller. 2013. Functional habitat connectivity for beach mice depends on perceived predation risk. *Landscape Ecology* **28**:547-558.
- Wilson, M. C., X.-Y. Chen, R. T. Corlett, R. K. Didham, P. Ding, R. D. Holt, M. Holyoak, G. Hu, A. C. Hughes, L. Jiang, W. F. Laurance, J. Liu, S. L. Pimm, S. K. Robinson, S. E. Russo, X. Si, D. S. Wilcove, J. Wu, and M. Yu. 2016. Habitat fragmentation and biodiversity conservation: key findings and future challenges. *Landscape Ecology* **31**:219-227.
- Winchester, C., S. B. Castleberry, and M. T. Mengak. 2009. Evaluation of factors restricting distribution of the endangered Key Largo woodrat. *Journal of Wildlife Management* **73**:374-379.
- Zollner, P. A. and S. L. Lima. 2005. Behavioral tradeoffs when dispersing across a patchy landscape. *Oikos* **108**:219-230.

BIOGRAPHICAL SKETCH

After earning a B.A. in International Relations and French from Colgate University, Margo Stoddard worked on community gardening and ecotourism projects in the Central African Republic (CAR) as a Peace Corps volunteer. She was inspired to pursue her interest and education in ecology while working with a WWF rural development and conservation project in CAR. She earned her M.S. in Forest Science from Oregon State University and worked at OSU as a faculty research assistant on projects examining the influence of forest management on wildlife before beginning a doctoral program in the Department of Wildlife Ecology and Conservation at the University of Florida. She received her Ph.D. in August 2017.