Population Viability Analysis for the Southeastern Beach Mouse (*Peromyscus polionotus niveiventris*)

Photo: NASA

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ABBREVIATIONS

ABM	Alabama beach mouse
AFP	Avalon State Park to Fort Pierce Inlet State Park (habitat unit)
CC	Cape Canaveral
CCSFS	Cape Canaveral Space Force Station
CNS	Canaveral National Seashore (habitat unit)
CPSG	Conservation Planning Specialist Group
CV	Coefficient of variation
EV	Environmental variation
GD	Gene diversity (as proportion of starting level in 2020)
GIS	Geographic Information System
HC	Happy Creek (habitat unit)
HTF	High tide flooding
IUCN	International Union for Conservation of Nature
K	Carrying capacity (for beach mice)
KSC	Kennedy Space Center
LE	Lethal equivalents
MINWR	Merritt Island National Wildlife Refuge
N	Population size (number of individuals)
NASA	National Aeronautics and Space Administration
PE	Probability of extinction
PINWR	Pelican Island National Wildlife Refuge
PVA	Population Viability Analysis
SSC	Species Survival Commission
SD	Standard deviation
SDP	Smyrna Dunes Park area (habitat unit)
SEBM	Southeastern beach mouse
SIPI	Sebastian Inlet to Pelican Island (habitat unit)
SLR	Sea level rise
SQ	Status quo
ST	Sensitivity testing
TS/ts	Timestep
USFWS	United States Fish and Wildlife Service

EXECUTIVE SUMMARY

In collaboration with project partners from the US Fish and Wildlife Service, the National Aeronautics and Space Administration Kennedy Space Center, the Florida Fish and Wildlife Conservation Commission, and others, the IUCN SSC Conservation Planning Specialist Group conducted a Population Viability Analysis (PVA) for the Southeastern beach mouse (SEBM), *Peromyscus polionotus niveiventris*, using the *VORTEX* individual-based stochastic simulation program. The objectives of this PVA were to project future short-term and long-term viability of wild SEBM populations under current and estimated future conditions; identify priority data gaps and their potential impacts on viability projections; estimate the relative impact of various threats on SEBM population viability; and evaluate potential management actions to improve viability, including potential reintroduction into former habitat.

The best available data and expert knowledge were used to develop the SEBM PVA model and scenarios. Project team members identified key threats, conservation goals, and management options to be examined via PVA modeling; provided data on species biology, habitats, and distribution; and assisted in estimating parameters needed for a robust model of population dynamics. Model development relied heavily on the 2020 Species Status Assessment (SSA) Report for the Southeastern Beach Mouse, in concert with published and unpublished data. Sensitivity testing of the base model indicated that stochastic processes, including inbreeding depression, may be important factors for small, isolated beach mouse populations, and that, in general, the base model was a reasonable representation of beach mouse population dynamics.

Four existing SEBM subpopulations were identified, and subpopulation-specific inputs were added to the base model to develop projections of future viability under the best estimates of current and future conditions. **Given model inputs and assumptions, the projected long-term viability of SEBM on Cape Canaveral (dune and inland subpopulations) is good**, with high gene diversity and no risk of extinction over 80 years, but with projected decline in numbers as habitat is lost due to projected sea level rise. In contrast, **projected long-term viability is poor for the smaller Smyrna Dunes (SDP) and Canaveral National Seashore (CNS) subpopulations inhabiting habitat more vulnerable to sea level rise and erosion due to major storms (hurricanes and Nor'Easters). These smaller subpopulations are projected to decline and eventually go extinct without management intervention.** The key driving factors affecting these projections are habitat loss (especially loss due to storms that can lower the carrying capacity for mice beyond sea level rise and high tide flooding impacts) and inbreeding depression. Inbreeding threatens SDP due to its small size and genetic isolation, while CNS is threatened both demographically and genetically by the risk of fragmentation and projected loss of connectivity with the Cape in the future due to sea level rise.

The greatest uncertainty surrounding these PVA projections is around the impact of storms on SEBM habitat and the post-storm recovery of that habitat, followed by uncertainty in the sensitivity of SEBM to inbreeding depression. Assumptions regarding connectivity and demographic rates across habitat types are also in need of verification, along with survival and reproductive rates of translocated mice.

Modeling of alternative management scenarios suggest that effective management may need to be two-fold: prolonged periodic genetic augmentation, in concert with efforts to restore, maintain and/or increase SEBM habitat and its carrying capacity (K) to support beach mice. While all potential habitat management actions were not explicitly modeled, any efforts to increase K for beach mice are likely to be beneficial, including post-storm habitat restoration as well as other potential habitat management strategies. Smaller subpopulations are more likely to require periodic genetic augmentation, especially if

isolated. **In general, the larger and more stable the habitat for a SEBM subpopulation, and the better the connectivity with other SEBM subpopulations, the less management effort will be needed to promote population viability**.

Model results indicate that the SEBM subpopulation in dune habitat on the Cape is likely of sufficient size, demographic health, and genetic diversity to serve as a source population for periodic translocations to reinforce existing SEBM subpopulations and/or in reintroduction efforts to re-establish extirpated SEBM populations in former habitat. Of the three potential reintroduction sites explored in this PVA, the Cape inland habitat known as Happy Creek within Kennedy Space Center offers the best projected viability and may be a good option for developing and refining SEBM reintroduction techniques. The other potential reintroduction sites on Orchid and North Hutchinson Islands are smaller and more vulnerable to habitat loss, and may pose to be more challenging and require more intensive management.

These PVA model results are influenced by the model inputs and assumptions surrounding them. Thus, rather than focusing on precise estimates of future population projections found in this report, it is suggested that **this model be viewed as a tool to better understand the primary driving factors influencing SEBM viability and to identify the most vulnerable subpopulations and situations**. This, in turn, can help to guide species and habitat managers in addressing important data gaps through relevant research and to prioritize management actions that are most likely to be effective to meet program goals across the taxon's distribution and former range.

SOUTHEASTERN BEACH MOUSE PVA AND BASE MODEL

PROJECT OVERVIEW

The Southeastern beach mouse (*Peromyscus polionotus niveiventris*) is one of 16 subspecies of *Peromyscus polionotus*, which consists of 8 subspecies of "beach mouse", all of which are of various levels of conservation concern (up to and including extinction) and 8 additional subspecies of "oldfield mouse". Beach mice are vulnerable to loss and alteration of their dune habitat as well as severe weather events, habitat alteration, feral cats and other threats.

In collaboration with project partners, the IUCN SSC Conservation Planning Specialist Group (CPSG) conducted a Population Viability Analysis (PVA) for the Southeastern beach mouse (SEBM) using the *VORTEX* individual-based stochastic simulation program. CPSG met with project partners in virtual and in-person meetings from March 2022 to October 2024 to identify key threats, conservation goals, and management options to be examined via PVA modeling; provide data on species biology, habitats, and distribution; and to derive estimates of parameters needed for a robust model of population dynamics. Model results will help guide revision of recovery criteria for this taxon and identify management strategies to support recovery.

The general objectives of this PVA are to:

1) Project future short-term and long-term viability of wild SEBM populations under current and estimated future conditions;
2) Identify priority data gaps and their impacts on viability projections;
3) Estimate the relative impact of various threats (e.g., sea level rise, major storms); and
4) Evaluate potential management actions, including their robustness across data uncertainty.

Early in the process, the SEBM PVA team identified their vision for the taxon as "to have a viable metapopulation across the taxon's former range comparable to that at the time of Federal listing in 1989". Re-establishment of SEBM populations in some areas of former occupancy would be needed to achieve this vision. This vision provided a framework for interpreting model results and to guide the development of alternative model scenarios.

As the initial phase of this PVA, a demographic population model was developed for the Southeastern beach mouse (SEBM) using the *VORTEX* 10.6 software program (Lacy and Pollak 2023). This model is designed to simulate the seasonal and annual life history characteristics of this taxon living in free-ranging conditions. Model inputs are based upon published information, including previous beach mouse PVAs, as well as data and expert opinion provided by members of the Southeastern Beach Mouse PVA team (see Appendix A).

The second phase of the project was to use this base model to develop and parameterize scenarios to assess the short- and long-term viability of wild SEBM subpopulations under current and estimated future conditions, explore the impact of uncertainty in model inputs, determine factors driving viability, and explore management options to improve viability. A list of desired model outputs was developed by the PVA team that include the number of mice, gene diversity (as a proportion of the starting level in 2020), and risk of extinction for each subpopulation and for the metapopulation, over time (in years 2050, 2070 and 2100, to match scenarios in the draft Species Status Assessment).

MODEL STRUCTURE AND INPUTS

VORTEX is a Monte Carlo simulation of the effects of deterministic forces as well as demographic, environmental, and genetic stochastic events on wild or captive small populations. *VORTEX* models population dynamics as discrete sequential events that occur according to defined probabilities. The program begins by either creating individuals to form the starting population or importing individuals from a studbook database and then stepping through life cycle events (e.g., births, deaths, dispersal, catastrophic events), typically on an annual basis. Events such as breeding success, litter size, sex at birth, and annual survival are determined based upon designated probabilities that incorporate both demographic stochasticity and annual environmental variation. Consequently, each run (iteration) of the model gives a different result. By running the model hundreds of times, it is possible to examine the probable outcome and range of possibilities. For a more detailed explanation of *VORTEX* and its use in population viability analysis, see Lacy (1993, 2000) and Lacy *et al*. (2023).

The SEBM *VORTEX* model uses a 28-day timestep, with 13 *VORTEX* timesteps in one calendar year. This provides sufficient resolution to simulate juvenile dependency, sexual maturity, parturition, and individual lifespan. Individuals in the initial (starting) population were allocated to age-sex classes (timesteps) to represent a stable age distribution based upon the vital rates used in the model.

Detailed descriptions of base model inputs are provided below. A summary table of inputs can be found in Appendix B.

SEASONALITY

Seasonal fluctuation in population numbers has been observed in several *P.p.* subspecies even in the absence of seasonal hurricanes or storms. Alabama beach mouse (ABM) populations, *P.p. ammobates*, peak in late winter/early spring and are the lowest in late summer/early fall, possibly due to seasonal changes in food availability in dune habitat and other factors (Hill 1989; Rave and Holler 1992; Swilling and Wooten 2002; also see Traylor-Holzer *et al*. 2005). Similar trends have been observed in Perdido Key beach mice *P.p. trissyllepsis* (Gotteland *et al.* 2016) and SEBM populations (Breininger *et al*. 2018). Seasonal population fluctuations may be attributed to the availability of food resources, the ability to breed throughout the year, as well as dispersal and mortality (USFWS 2022). Observed seasonal trends are stronger for reproduction than for survival. Figure 1 provides a schematic illustration of this seasonality in food availability and beach mouse demography and population size.

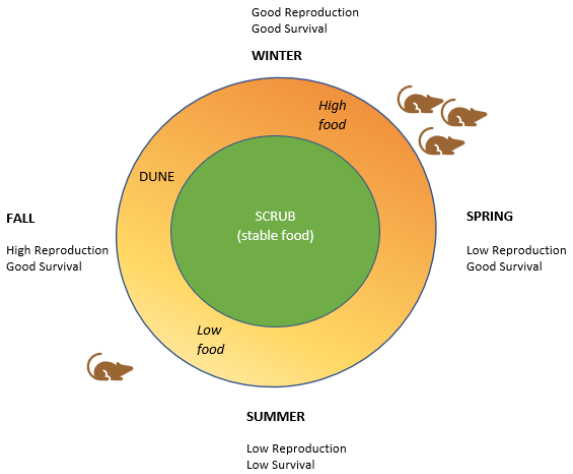


Fig. 1. Diagram of seasonality in food availability and beach mouse demography and numbers.

In order to simulate seasonal changes in *VORTEX*, a Global State Variable (GS1, labeled SEASON) was created in the model to delineate seasons, as follows:

1 = Winter	3 timesteps (84 days)	1 December to 22 Feb
2 = Spring	3 timesteps (84 days)	23 Feb to 17 May
3 = Summer	4 timesteps (112 days)	18 May to 6 September
4 = Fall	3 timesteps (84 days)	7 September to 30 November

This allows the option to include seasonality in demographic rates and other inputs (e.g., hurricane risk). As there are 13 timesteps per year, the 'extra' timestep was added to "summer", in part due to increasing temperatures related to climate change, and to err on the side of lower projected growth. Each model year begins in January, simulating the calendar year in 13 timesteps.

MORTALITY

VORTEX uses age-and sex-specific mortality rates, which can be calculated from survival rates estimated from field data. Below in Table 1 are the model inputs for age-specific mortality rates used in the SEBM model, followed by an explanation of the derivation and application of these rates.

Table 1. Preliminary mortality rates (per time step) for the SEBM preliminary model.

	Juveniles* (0-28 days)	**Sub-adult (29-56 days)**	**Adult (> 56 days)**	**Adult EV**
Winter	0.578	0.109	0.099	0.051
Spring	0.456	0.165	0.150	0.098
Summer	0.802	0.270	0.245	0.063
Fall	0.518	0.175	0.159	0.007
Average over year:	*0.605*	*0.186*	*0.170*	

**actual model inputs for juveniles were reduced to account for dam mortality – see below*

Adult and sub-adult mortality

Seasonal adult survival rates were estimated from mark-recapture data for wild SEBM (KSC's LC41 field site) from 2001-2009 (Stolen, pers. comm.). Data for Winter 2002 were omitted from analysis due to very small sample size, while data for Fall 2004 were omitted because two hurricanes occurred in this area during September of that year and sample size was small. The resulting estimates indicate that mortality is highest during summer, which is consistent with seasonal trends used in ABM PVA models (Traylor-Holzer *et al.* 2005).

The ABM PVA used higher rates for sub-adults (age 29-56 days), with subadult mortality estimated to be ~1.3x adult mortality. There are no data suggesting higher mortality in sub-adult vs adult SEBMs; however, data are sparse. It is not unreasonable for mortality to be higher during the weeks following dispersal from the natal burrow and establishment of a burrow and home range. For this model, subadult mortality was increased slightly, to 1.1x that of adults.

If these mean annual rates (0.186 for subadults; 0.170 for adults) are applied to mice that survive the first timestep, average longevity is about 130 days of age. This is not out of line with other measures of beach mice longevity: Keim 1979: majority lived $\leq$ 124 days; Gotteland *et al* 2016: mean longevity of 116 days, based on standard estimated ages such as adults were 40 days of age at first capture; Oddy 2000: mean longevity of SEBM on Cape Canaveral = 113 days, with maximum longevity = 596 days. Using the mean rates in Table 1, few mice (~3%) would be expected to survive to one year of age (Figure 2).

Expected longevity varies seasonally, with mice born in early winter likely to live longer than early summer births.

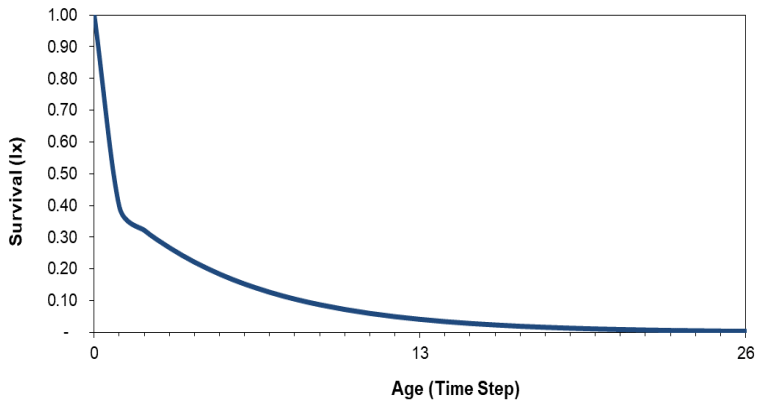


Fig. 2. Survivorship (Lx) for SEBM based on mean deterministic rates in Table 1.

These survival data (LC41, for 2001-2009) were analyzed by season-year (Stolen, pers. comm.) to allow for estimation of annual variation in mortality due to variation in environmental conditions from year to year. Annual environmental variation (EV) was high relative to the mean rate for winter and spring (coefficient of variation, CV=0.52-0.65), suggesting that conditions affecting adult SEBM survival are most variable during this time (in non-hurricane years). Little EV was observed in fall, which coincides with low EV for reproduction in fall (see *Reproduction* section below).

Juvenile mortality

Estimates of juvenile mortality could not be found for any wild beach mouse populations. Laboratory mortality rates at Auburn University suggest up to 55% mortality to age 21 days (see ABM PVA report in Traylor-Holzer *et al*. 2005). Kaufman and Kaufman (1987) report that mortality rates in the lab (to day 21) average 25% but vary greatly from 0 to 50%. Lacy *et al*. (1996) reported 14% pre-weaning (0-20 day) mortality for a laboratory population of *P. p. leucocephalus*, and 44% pre-weaning mortality from data in Brewer *et al*. (1990). Captive mortality rates are, in all likelihood, lower than those in natural populations because captive individuals do not have to forage for food or avoid predators.

Seasonal juvenile mortality (0-28 days of age) for wild SEBM was estimated using seasonal subadult and adult mortality rates above, seasonal reproductive rates and mean litter size (see below), and estimated recruitment rates (from KSC LC41 field site, from 2001-2009; Stolen, pers. comm.). As with adult mortality, juvenile mortality is estimated to be highest in summer (80.2%). Estimated mortality for the first 28 days is 45.6-57.8% for the rest of the year, which is in line with that reported in the laboratory.

Juvenile mice are dependent upon their dam for ~20 days (pre-weaning period). The *VORTEX* option was used to kill off dependent offspring of any females that die in the model. Thus, input values to be entered into the software for juvenile mortality were adjusted to prevent double counting this source of juvenile mortality. Actual values entered into the model were: 0.531 (winter); 0.360 (spring); 0.738 (summer); and 0.427 (fall); when combined with expected mortality of lactating females, this results in the juvenile mortality rates in Table 1.

Inbreeding depression is imposed in *VORTEX* as lower juvenile survival for inbred individuals, with the degree of impact determined by the number of lethal equivalents (LE). Brewer *et al*. (1990) calculated 5.52 LE in SEBM (founded from wild-caught mice from Canaveral National Seashore) under laboratory

conditions. This estimate was reanalyzed by Lacy *et al*. (1996) to account for probability of breeding, resulting in a new estimate of 4.72 LE. Jiménez *et al*. (1994) found that the impact of the same degree of inbreeding on survival in white-footed mice (*P. leucopus noveboracensis*) is more severe in the wild than in laboratory conditions, which are more benign than natural conditions. The default *VORTEX* setting is 6.29 LEs, which is a conservative estimate for wild vertebrate populations (O'Grady *et al.* 2006) and is not unreasonable for wild mouse populations. This default value of 6.29 LE was used for sensitivity testing. After some discussion, the PVA team chose the more conservative laboratory estimate of 4.72 LE for modeling the wild SEBM subpopulations. While wild populations are generally thought to show greater inbreeding effects than those of the same species under laboratory conditions, the stochastic nature of beach mouse population dynamics may have purged some of the effects in the wild. Two mechanisms were used to apply these inbreeding impacts: 50% of impacts were implemented via recessive lethal alleles and 50% were implemented by applying proportionately higher juvenile mortality to increasingly more inbred individuals (*VORTEX* default setting).

Maximum age

Maximum age was set to 26 timesteps (two years). However, in the model few mice live beyond 12 months with the deterministic application of the mortality rates above (~3% live to 12 months; ~1% live to 18 months) – see Figure 2.

REPRODUCTION

Mating system

Breeding was modeled using long-term monogamy, shifting to long-term polygyny if the adult sex ratio becomes female biased to allow extra-pair copulations for unpaired females (USFWS 2008). Polygyny is triggered in the model if the adult sex ratio (female/male) > 1.22 (i.e., >55% of adults are female), with a maximum of two female mates per male. All adult males were considered to be potential breeders (i.e., in the breeding pool and eligible for pairing with a female).

Age of first reproduction

In *VORTEX* this represents the average age of mice at the time of birth of their first offspring (not time of first mating) and was set for age 3 timesteps (~3 months) for both sexes. Beach mice reach sexual maturity at ~55 days (Ehrhart 1978), and average gestation is 29-30 days (Weston 2007), so most mice may produce their first offspring at ~3 timesteps (84 days). No reproductive senescence was included in the model, as mice have been observed to reproduce beyond age 2 years in laboratory conditions. Few mice are likely to live to one year of age given the mortality rates used in the model.

% adult females breeding (mean and EV)

In *VORTEX* this represents the average proportion of adult females that produce a litter per timestep, and is assumed to be independent of age. There is stronger evidence for seasonality in reproductive rates than in survival rates (ABM PVA, Traylor-Holzer *et al*. 2005). Table 2 provides unpublished estimates of reproductive females in wild SEBM (KSC's LC40-41-BH) (Provancha *et al*., pers. comm.). Rates are based on eight years of data (2001-2009) with $N_{females}$ = 163 to 374 per season, with 85% of data from 2006-2009. Reproductive females were those that exhibited mammae indicative of lactation or were noticeably pregnant. Given that lactation lasts ~20 days, the combination of these two classifications is reasonable for estimating the proportion of females reproducing during a single timestep.

Other unpublished data collected at KSC show a similar seasonal pattern as data in Table 2 but are lower in absolute values. These data were not used here, as they were pooled from three separate projects between 1990 and 2005 that had substantially smaller sample sizes for most seasons.

Table 2. Percent of adult females that are actively reproductive (per time step) for the SEBM base model.

	Winter	Spring	Summer	Fall
Mean	58.7	35.4	38.6	83.7
EV	21.9	25.3	19.6	3.6
CV	0.374	0.715	0.507	0.043
# years	8	6	7	7

These data demonstrate a seasonable pattern in reproduction that is moderate in winter, lower in spring and summer, and high in fall. Fall reproduction is consistently high, whereas there is substantial annual variation in reproductive rates during the rest of the year. These rates result in a mean of 54.5% adult females reproducing per timestep across the year.

Density-dependent effects

No density dependence in reproduction was included in the model. Evidence of long-distance movements by beach mice after reintroductions suggest there is no Allee effect (low reproductive rates at low density due to difficulty finding a mate). Beach mice can reach high densities when sufficient food resources are available, but there are no data on SEBM density-dependent impacts on vital rates.

The ABM PVA model did incorporate density-dependent reproduction, which helped to control population size as N approached K (carrying capacity of the habitat for beach mice), noting that density-density reproduction has been observed in *Peromyscus*, although the shape of the density-dependence curve differs among species. A literature search of 13 populations of *Peromyscus* found estimates of percent breeding at low and high density to be 63% and 28%, respectively. Unpublished data for Perdido Key and Choctawhatchee beach mice suggest a high reproductive rate of 90% and a low of 14% (Lynn, pers. comm.). Given the variability in reproductive rates using in this SEBM model due to both seasonal effects and EV, as well as the paucity of density-dependent impacts for SEBM, no density-dependent impacts were included in the SEBM model.

Litter size

Given gestation length, a maximum of one litter can be produced within one timestep by an adult female. Offspring are randomly assigned sex using a 50:50 sex ratio. In the model, offspring are dependent upon their dam for one timestep and die if she dies during their first timestep. Mean litter size was taken from Lacy *et al.* (1996) laboratory studies of SEBM. This study found that the first litters of females are smaller (mean=3.56) and subsequent litters are larger (mean = 4.33). These means were used in the model with a Poisson distribution. It is assumed that litter size will be similar in the wild but that neonatal mortality may be higher than in laboratory conditions (*see Juvenile Mortality*).

DEMOGRAPHIC MODEL POPULATION DYNAMICS

Seasonality

The input values described above were used to run a preliminary demographic model for SEBMs. The results in Figure 3 are based on a scenario with an initial population of 900 and habitat carrying capacity of 1000, sufficient to minimize small population effects. No inbreeding impacts were included. Two general catastrophe events were included, to simulate a major hurricane (~once every 25 years, in summer/fall) and a Nor'Easter storm (same frequency but in March-April). This preliminary model was run as a closed population for 20 years (260 timesteps) with 500 iterations. **The results are presented solely for the purpose of examining the resulting population dynamics, including seasonality, and do NOT represent actual SEBM wild population projections**. These results are consistent with the seasonal pattern expected for a large, healthy SEBM population with no loss of habitat or other major threats. The population peaks in late winter-early spring and reaches its low in late summer-early fall. Stochastic growth rate (r) is highest in late fall (~r=0.35), with negative growth during mid-summer to early fall. This approximates the seasonal population dynamics outlined in Figure 1.

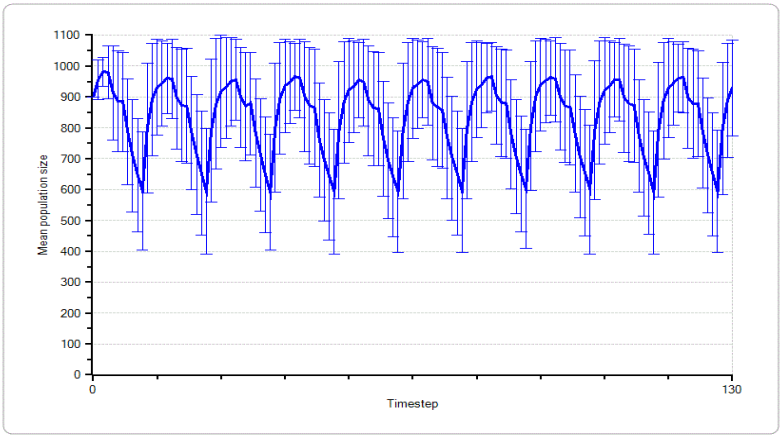


Fig. 3. Mean population size over 10 years for a hypothetical population using the base demographic model (K=1000) Error bars represent ± 1 SD.

Truncation to K

Since neither density-dependent reproduction nor density-dependent mortality are included in the model, probabilitistic truncation is used to limit population size during seasons of positive growth. This is implemented in the base model by applying the same probability of additonal mortality to each individual across all age and sex classes at the end of each timestep if population size (N) exceeded carrying capacity (K). This *VORTEX* default setting was used in sensitivity testing. This results in the mean population size of ~92-94% of K when growth is high and the population is at capacity (Figure 3).

After discussion of high density situations for beach mice, the PVA team decided to implement truncation to K in the actual SEBM wild population models by limiting truncation to juveniles only, as it is believed that higher mortality under high density conditions is mostly likely to impact young in the burrow. Truncating all age classes vs juveniles only does not affect base model results, but may impact the availability of juveniles in some of the PVA scenarios (i.e., trapping juveniles for translocation).

SENSITIVITY TESTING

General sensitivity testing (ST) of the base model was conducted by systematically varying inputs (demographic rates, presence/absence of threats) to observe the impact on population growth. This can provide insight on inputs that may drive viability projections, which in turn can identify important (and unimportant) areas of uncertainty, as well as help identify management actions that may most affect population viability. Stochastic growth rate (the mean exponential rate, r = ln(N+1)/N, across years in the population simulation) is an appropriate measure, as this is calculated by *VORTEX* prior to truncation in the model and provides a good measure of potential for demographic population growth.

ST scenarios were run for a population with K=1000 for 20 calendar years (260 timesteps), with an initial population of 850 starting in January. This population size should be large enough to minimize any impacts of demographic stochasticity. Stochastic r for the base model (r=0.0436) was compared to the resulting stochastic r from changing one input value at a time, as listed in Table 3. These represent model inputs with uncertainty.

Table 3. Preliminary mortality rates (per time step) for the SEBM preliminary model.

Scenario	Parameter	Base value	ST scenarios
M2Season	Seasonal mortality	4 seasons	Two seasons (summer vs rest of year)
Rep	% females breeding	Table 2 means	Mean ± 5%; mean ± 10%
AdMort	Adult mortality	Table 1 means	Mean survival (1 – mean mortality) ± 5% & ± 10%
SAdMort	Subadult mortality	Table 1 means	Mean survival (1 – mean mortality) ± 5% & ± 10%
JuvMort	Juvenile mortality	Table 1 means	Mean survival (1 – mean mortality) ± 5% & ± 10%
SAd=AdMort	Subadult mortality	1.1x adult mort.	Same as adult mortality
SAMEVAd	Subadult mortality EV	1.1x adult mort. EV	Same as adult mortality EV
JMEVAd	Juvenile mortality EV	50% AdMort EV CV	Same CV as adult mortality (AdMort) EV CV
Inbreed	Inbreeding depression	6.29 LE	No inbreeding depression
NoCats*	Major storms	50% survival	No storms (no catastrophes)
HurrOnly	Major storms	Both storm types	Only hurricanes (50% survival); no Nor'Easters
NEonly	Major storms	Both storm types	Only Nor'Easters (50% survival); no hurricanes
NoA3Trun	Truncation when N≥K	All ages	No truncation of prime age adults (A=3 ts)

**Note: represents no catastrophes, not the absence of cats* Felis catus

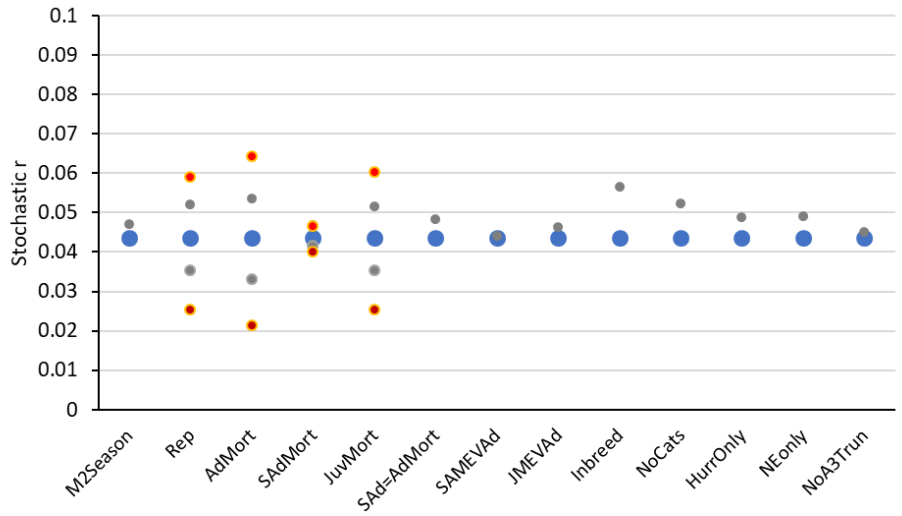


Fig. 4. Mean stochastic population growth rate for sensitivity testing scenarios. Blue = base model; gray = most ST scenarios; red = ST scenarios for ± 10%.

Figure 4 provides the sensitivity testing results for stochastic r. Across the ranges tested, population growth is most sensitive to adult survival, followed closely by reproduction and juvenile survival. Removing catastrophes increases growth slightly, as major storms occur, on average, 1.6x over 20 years. Removing inbreeding also increases growth; 20 years represents ~38 generations and results in about 7% inbreeding accumulation in a panmictic population of 1000 mice. Note that population growth remains positive across all values tested. These results suggest that factors affecting reproduction and survival might slow population growth, and that inbreeding depression can impact population growth.

Stochastic processes such as inbreeding, demographic stochasticity, environmental variation and catastrophes can have greater impacts in small populations. Additional ST scenarios were run to assess the influence of population size on population growth and extinction risk over 20 years, and the interaction between population size and inbreeding. Scenarios using the base model were developed with N=K=50 to 1000, with and without inbreeding depression. Orange bars on Figures 5 and 6 show that non-genetic stochastic processes begin to impact population growth rate and extinction risk for isolated populations of fewer than 200 mice. When inbreeding depression (i.e., 6.29 LE) is included, populations of 600 mice or fewer are impacted, with no growth and high chance of extinction in 20 years for populations of fewer than 200 mice. Precise viability projections based on size will be situation specific; however, these ST results suggest the importance of population size and potential inbreeding depression in SEBM populations. Larger populations are expected to become vulnerable with long timelines, as inbreeding accumulates with each successive generation.

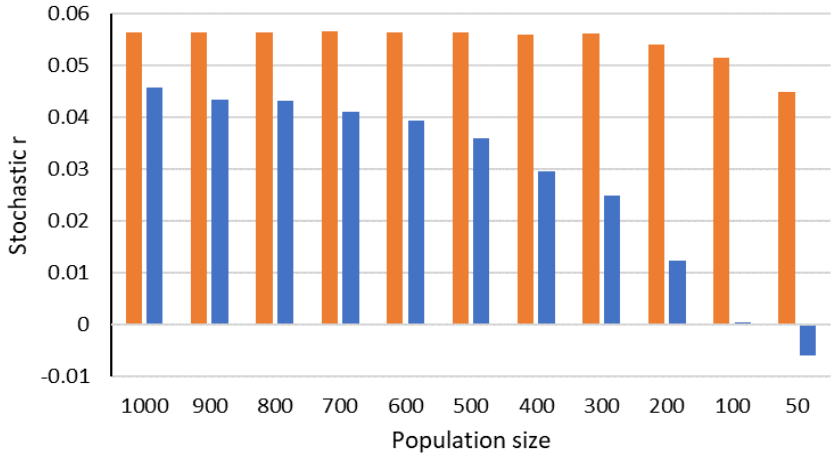


Fig. 5. Mean stochastic population growth rate over 20 years for populations ranging from 1000 to 50 mice, with and without inbreeding depression.

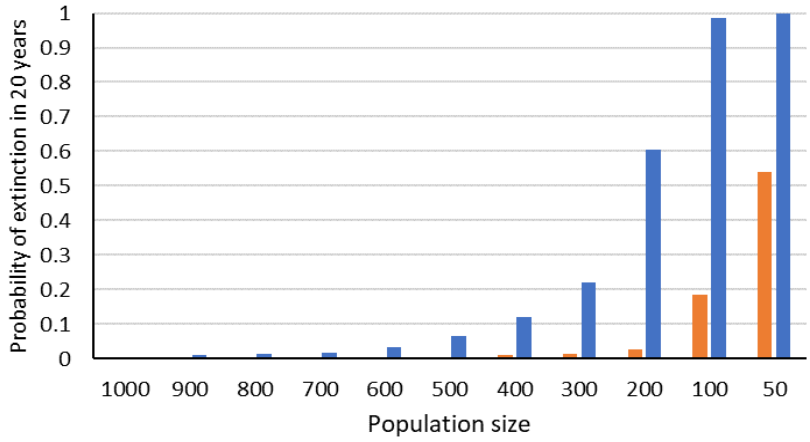


Fig. 6. Probability of extinction over 20 years for populations ranging from 1000 to 50 mice, with and without inbreeding depression.

CURRENT SEBM METAPOPULATION

Population distribution and structure

The historic range of SEBM once extended along the east coast of Florida from Ponce Inlet in the north to Hillsboro Inlet to the south. This range is divided into nine geographic segments determined by inlet and channel breaks within the barrier islands (USFWS 2022). SEBMs disappeared from much of the southern range by the time of federal listing in 1989. Currently, SEBMs are only confirmed to live in the Canaveral North Geographic Segment (Ponce Inlet to Port Canaveral Entrance Channel), at the northern end of the historic range (Figure 7). The status of SEBM populations in the Orchid Island/North Hutchinson Island Geographic Segment is listed as uncertain; SEBMs were last recorded there in 2012 despite continued efforts since then to detect their presence.

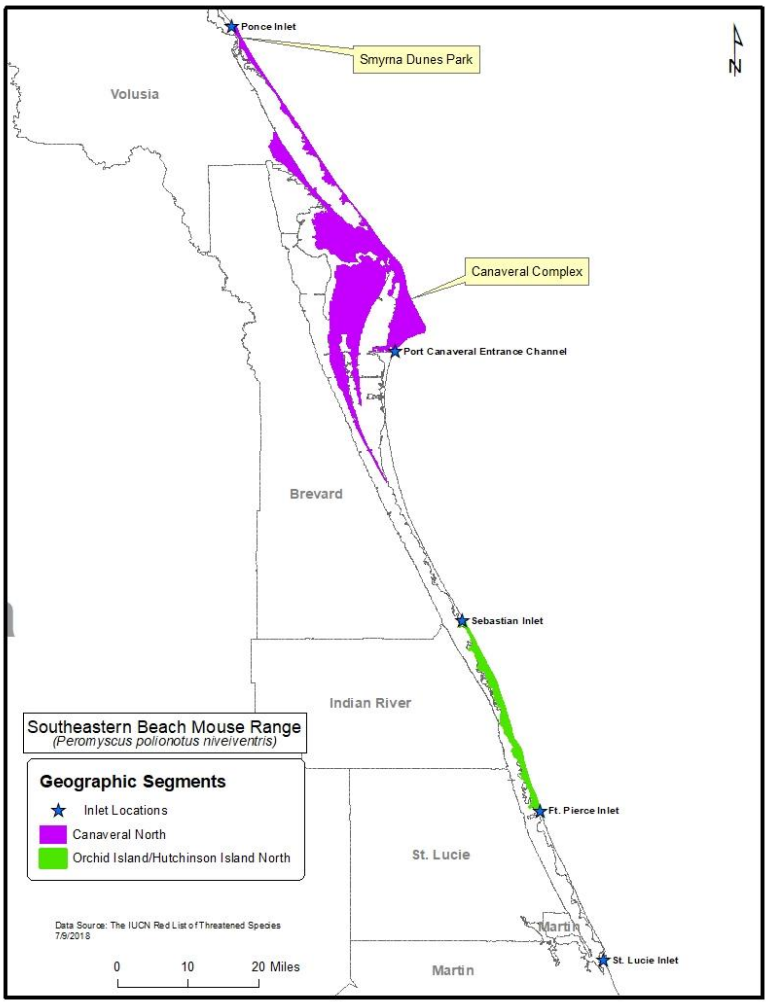


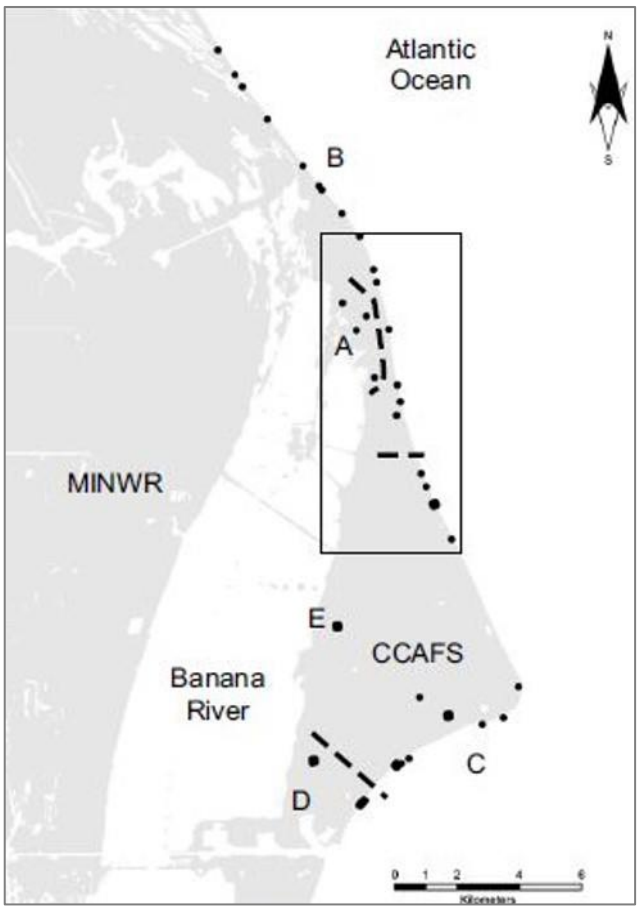
Fig. 7. Southeastern beach mouse distribution map from USFWS (2022). Call out boxes indicate extant populations.

There are two known isolated extant SEBM populations, both in the Canaveral North Geographic Segment. The northernmost Smyrna Dunes Park (SDP) population is a small, isolated population. The Canaveral Complex (CC) population is considered to be the core SEBM population and occupies habitat from the northern reaches of Canaveral National Seashore (CNS) south to the Port Canaveral Entrance Channel.

These two populations are separated by areas of heavy development that are assumed to prevent movement between them. Degner *et al*. (2007) concluded that the proportions of heterozygous individuals and allelic richness were higher in SEBM on CC as compared to SDP, indicating reduced genetic diversity within SDP, consistent with founder effects and/or genetic drift. Kalkvik *et al.* (2018) also found lower genetic diversity in SDP.

The CC population likely functions as a metapopulation. Zimmerman *et al*. (2015) identified at least four subpopulations (Figure 8). These genetic clusters are not completely isolated from each other but are thought to have limited dispersal among them. Divergence among these clusters is relatively recent and corresponds to the timing of the development of roads and structures within the CC.

Fig. 8. Genetic breaks (dashed lines) for the SEBM CC population. Dots = sampling areas. A-D = genetic clusters identified by Geneland (from Zimmerman *et al.* 2015).



Habitat

Beach mice utilize a heterogenous mix of coastal habitat types, including dune, coastal strand and scrub habitats. For PVA purposes, SEBM habitat was grouped into two large-scale habitat descriptors similar to those described by USFWS (2022): 1) dune/coastal strand (dune); and 2) scrub/inland areas (inland).

Habitat for the SDP population consists of ~81 ha of almost entirely dune and coastal strand. This habitat extends ~0.5 miles across the barrier island from the beach to the lagoon and can provide refugia from storm impacts.

The CC SEBM metapopulation ranges along 72 km and can be divided into three areas: 1) CNS section (northern 30 km), which is narrow and has little to no suitable inland habitat; 2) KSC (central 10 km), with some inland habitat; and 3) CCSFS section, labelled as CCAFS (southern ~30 km), with substantial connected suitable inland habitat.

There is considerable uncertainty regarding the optimal and range of habitat conditions for SEBM. SEBM habitat is generally more heavily vegetated than beach mouse habitat along the Gulf coast (USFWS 2022). Long-term occupancy of coastal scrub habitat has been documented for SEBM (Stout 1979; Suazo *et al.* 2009; Simmons 2008). The availability of food resources in the foredunes fluctuates (Sneckenberger 2001), while scrub habitat provides a more stable level of food resources. Simmons (2008) recorded higher SEBM captures in coastal scrub grids vs dune grids, five months post-hurricane. Scrub and other inland areas may provide critical refugia habitat and may support SEBM resource needs, provide movement corridors, and/or support an extension of a population (USFWS 2022).

Most occurrences of SEBM in scrub habitat are in coastal scrub. However, SEBM are known to occur in oak-saw palmetto scrub habitat within CC. Management regimes, including periodic prescribed fire, may influence the suitability of this habitat for SEBM (USFWS 2022).

Subpopulations for the SEBM PVA

The current distribution and population structure of SEBM was discussed by the SEBM PVA group, based on both data and expert knowledge. The resulting consensus identified four extant subpopulations, as follows (Figure 9):

- one isolated subpopulation in and near Smyrna Dunes Park (SDP);
- a metapopulation of three connected subpopulations in Cape Complex (CC), consisting of two beach subpopulations (CNS, aka Beach1, and Beach2 in MINWF/KSC/CCSFS on the Cape) and one subpopulation in inland scrub and ruderal habitat in CCSFS (Inland)

PVAs were conducted for each of these four areas, with SDP and CNS run as individual models, and Beach2 and Inland run as two connected subpopulations within a metapopulation model (see following PVA sections for details). Each PVA included a status quo (SQ) model as the best projection for future viability, exploratory scenarios to assess the primary factors driving viability and to assess the significance of uncertainty in model inputs, and alternative management scenarios to explore options for improving viability.

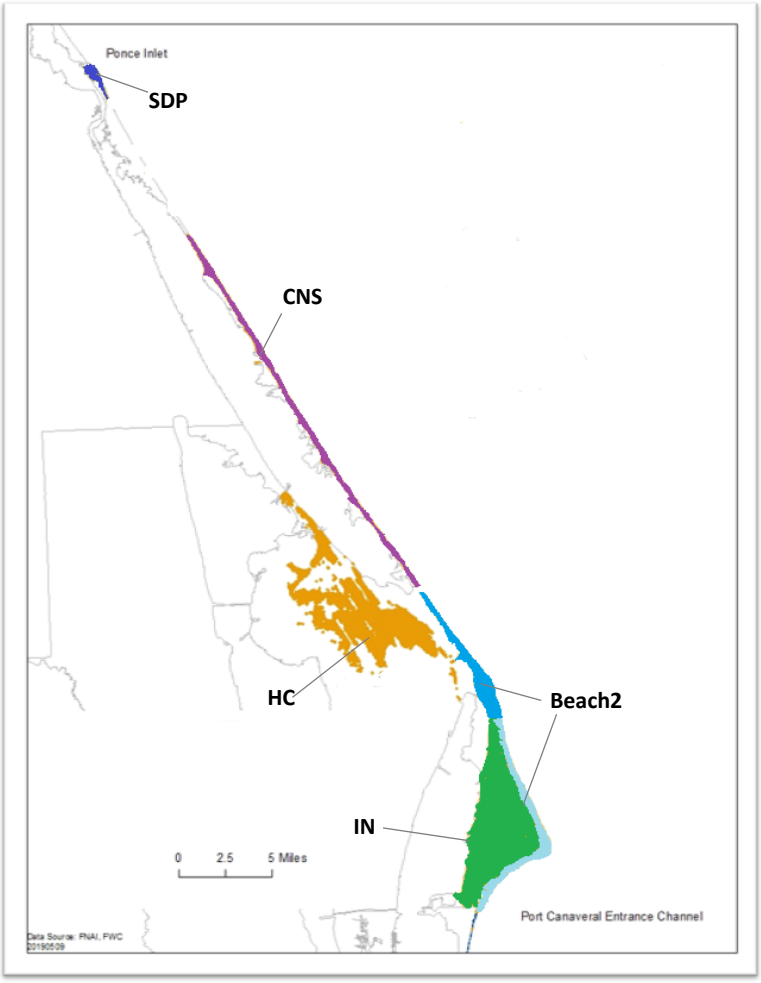


Fig. 9. Southeastern beach mouse subpopulations used in the PVA. Beach2 includes beach habitat on MINWR and KSC (medium blue) and on CCSFS (light blue). HC designates a potential reintroduction area on CC near extant SEBM subpopulations.

Additional areas were identified where SEBM are believed to be extirpated and which may be potential reintroduction sites. These include an inland area on the Cape known as Happy Creek (HC) (Figure 9), as well as former SEBM habitat to the south on the barrier islands of Orchid Island and North Hutchinson Island (green areas on Figure 7).

Estimating population size and carrying capacity

Beach mice can achieve high densities (due to high reproductive potential) and have significant fluctuations in abundance, including seasonally (Extine 1980). Blair (1951) recorded densities of 2 to 3.5 mice/ha for Santa Rosa beach mice (SRBM), *P.p. leucocephalus*, while Frank (1996) estimated 2 to 90 mice/ha for Anastasia Island beach mice (AIBM), *P.p. phasma*. Lynn (2000) observed seasonal variations in population densities of Alabama beach mice (ABM), *P.p. ammobates*, with increases noted during fall, winter, and spring months and decreases in summer months. Peaks during fall, summer, and winter months have been observed in SEBM (Oddy 2000).

Recent work (Stolen *et al*., pers. comm.) over ~500ha of prime SEBM habitat on CNS and KSC found a wide range of beach mouse densities that showed a seasonal pattern of higher in spring and winter, lower in summer and fall. High density in dune habitat was ~8-9 mice/ha, and estimated at ~3 mice/ha in inland habitat. There are limited FWS data, with estimates for inland (spring only 2004&2005: ~7.5-8/ha) and for dune (spring 2004: ~6 spring 2005 ~5.5). Spring estimates for mice around launch pads are: 1.75/ha (spring 1996), 2.5/ha (spring 1997), and 3.8 mice/ha (spring 1998).

Some studies found no detectable differences between scrub and frontal dunes in beach mouse body mass, home range size, dispersal, reproduction, survival, food quality, and burrow site availability (Swilling et al. 1998; Swilling 2000; Sneckenberger 2001).

Species experts on the SEBM PVA team consider all available information, including on-the-ground knowledge of each habitat area, and generated density estimates for each area. Density estimates ranged from 5-9 mice/ha and 4-5 mice/ha for inland habitat, depending upon habitat quality. SEBM habitat area was multiplied by the estimated density for each area to determine the projected carrying capacity (K) of the habitat for SEBM.

Existing beach mouse subpopulations were assumed to be near or at habitat carrying capacity. Since population size varies seasonally, the base model was used to determine the mean N/K ratio at the beginning of January, when all scenarios were initiated. Mean N/K in this timestep is 85%; thus, initial size of extant subpopulations was set to 85% of K. Available habitat estimates were current for 2020; thus, models were initiated as beginning in January 2020.

Estimating impacts of sea level rise

The 2020 SSA Report provides estimates of SEBM habitat loss (in acres) due to sea level rise (SLR) for each of the SEBM subpopulations. These estimates were based on the 2017 NOAA SLR estimates (in feet) for Trident Pier, coupled with GIS analysis of the impacts of these higher sea levels on each subpopulation habitat. Subsequently, NOAA provided updated SLR estimates in 2022, which were lower than 2017 estimates and especially so in the first few decades. The 2022 NOAA analysis also provided estimates of High Tide Flooding (HTF) that leads to the periodic flooding of additional habitat above SLR. HTF (Moderate) estimates for Trident Pier are 2.76 ft by 2050, with no additional HTF estimates calculated above that level. SLR and HTF are not considered to be additive (Brooks, pers. comm.). For

this PVA, the following general methodology was used to project remaining SEBM habitat for each subpopulation:

1) Projected SLR was calculated using point estimates from the 2022 NOAA projections for years 2050 (1.51ft), 2080 (4.37ft), and 2110 (8.22ft), and extrapolating SLR for all intervening timesteps (to Year 2100) assuming a linear increase between points.
2) Projected HTF was calculated by assuming a level of 2.76ft by year 2050 and beyond, and extrapolating a linear increase from 0 to 2.76ft from 2020 to 2050.
3) Final sea rise was calculated for each timestep by using the larger of the two values (SLR or HTF). This resulted in HTF driving sea rise until year 2063, when projected SLR surpasses projected HTF.
4) To convert the final sea rise to SEBM habitat for a specific subpopulation, the closest benchmark(s) were used from the SSA based on the 2017 projections and GIS analysis. For example, the SSA states that a SLR of 2.56ft would flood 17.16ac of SEBM habitat in SDP. Using the value of 6.70ac flooded per ft of rise (17.16/2.56), this would estimate that 18.50ac will be flooded with SLR=2.76.
5) Finally, remaining SEBM habitat after accounting for SRL was converted to carrying capacity (K) using the density for that subpopulation.

This method includes the following simplifying assumptions: 1) Linear loss of K between the point estimates; 2) no seasonal differences in loss of K due to SLR; 3) no additional impacts of HTF beyond 2.76ft rise; and 4) no average change in density due to disproportionate loss of one habitat type over another in mosaic habitats. In addition, modeling includes the following subpopulation-specific assumptions:

SDP: no additional impacts of fragmentation of habitat; succession of vegetation into the spoils area simultaneously with SLR such that the relative proportion of habitats remains the same.

CNS: Loss of connectivity with the Beach2 subpopulation on the Cape occurs in 2080, as a consequence of rising sea levels.

Cape: Loss of connectivity with the CNS subpopulation occurs in 2080 in the model, as a consequence of rising sea levels.

Estimating impacts of major storms

The PVA team identified major hurricanes and Nor'Easter storms as catastrophic events to be included in the model. The frequency of major hurricanes hitting the Cape Canaveral area was taken from historic records, with a major hurricane affecting the Cape once every 25.17 years (about 4% annual risk) (https://hurricanecity.com/city/capecanaveral.htm). Hurricanes occur seasonally during mid-June to November (6 of 13 timesteps in the model) and with the highest risk in September. In the model, five of these timesteps were each given 0.6% risk of major hurricane occurrence, with 1% risk for the timestep representing September, to total 4% annual risk. The model assumes that all populations are hit when a major hurricane occurs (e.g., Category 4 or 5). Smaller, more frequent storms are considered to be covered through environmental variation in reproduction and survival, with no impact on habitat. Nor'Easter storms were modeled as an independent, second catastrophe with the same 4% annual risk but occurring only during the 2 timesteps representing March-April. The frequency of major storms is assumed to stay the same as in the past and not become more frequent in the future; however, this may

be an optimistic assumption due to some projections that the frequency of severe storms may increase over time due to climate change impacts.

The PVA team estimated the following impacts for a subpopulation if a hurricane or Nor'Easter occurs:

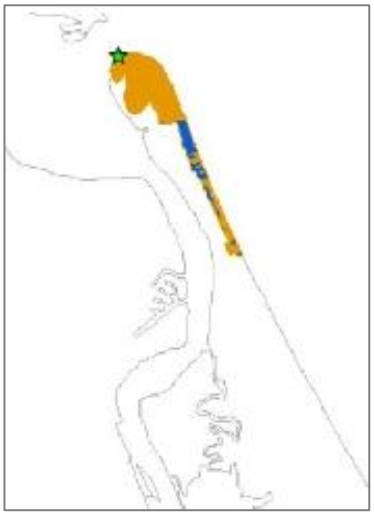
1) Direct mortality of SEBM across all age classes, varying from 10-30% depending upon the area and options for SEBM to reach higher ground away from the beach.
2) Potential impacts on successful reproduction (i.e., surviving births), varying from no successful reproduction during the timestep of the storm or the next timestep for vulnerable beach subpopulations, to only 50% reduction of reproduction during the storm timestep for mice in inland habitat. The logic is that any young in the dune burrows will be lost during the storm, and breeding behavior and burrows will be disrupted such there may be no successful litters produced in the subsequent time step in some areas.
3) Storms will result in habitat loss in narrow dune habitat (i.e., SDP, CNS, some potential reintroduction sites), reducing K for SEBM to 40% of pre-storm conditions (this is the amount of habitat estimated to be high enough to remain after a major storm).
   Note: Because there will be no successful reproduction during the storm and 20-30% direct mortality of all other age classes (plus 'normal' mortality) in beach subpopulations, the resulting mouse numbers should be far below K at the end of the storm timestep before truncation to the new post-storm K. Double-counting of mortality through these two model mechanisms should be minimal.
4) SEBM habitat (K) will recover in 5 years to 70% of pre-storm conditions (50% for CNS), with no further recovery after that point. Recovery is projected as being linear over these 5 years, which may necessitate some restoration efforts.
5) If another storm hits within 5 years (i.e., before 5-year habitat recovery is complete), then recovery from the second storm does not exceed 70% of K when the second storm occurs.
6) The model does not allow the habitat (K) to recover above habitat (K) available after SLR. Thus, K becomes the lower value of the two forces: recovered K after a storm vs K available after SLR.
7) This parameterization of habitat loss and recovery means that each storm will successively lower K, even in the absence of sea level rise. Successive storms within 5 years of each other are especially destructive.
8) No storm-related reduction in K is included for the Beach2 and Inland subpopulations on the Cape, or for inland habitat in reintroduction areas (i.e., Happy Creek, Pelican Island).
9) Each major hurricane also results in a 50% probability of permanent fragmentation of CNS into two fragments with equal number of surviving mice and equal K. The northern fragment remains isolated, while the southern fragment remains connected to the Beach2 subpopulation on the Cape until 2080. There is no fragmentation risk with Nor'Easters.
10) For the CNS subpopulation, recent storm erosion events in 2022 and 2024 were forced to occur in the model, with a loss of habitat down to 40% of pre-storm conditions and continued recovery at the same linear rate as described above (i.e., 10% of pre-storm K recovered every 5 years). No demographic impacts on survival or reproduction were included, and there was no risk of fragmentation for these recent events.

Details on subpopulation-specific model inputs are provided in each subpopulation PVA sections, as well as in Appendix B.

SMYRNA DUNES PARK SOUTHEASTERN BEACH MOUSE SUBPOPULATION

The northernmost subpopulation of the Southeastern beach mouse (SEBM) inhabits Smyrna Dunes Park and nearby lands (SDP). This beach mouse habitat is surrounded by water on three sides and is geographically isolated from other SEBM subpopulations by heavy development to the south (Figure 10, adapted from 2020 SSA). Habitat for the SDP beach mouse subpopulation consists of 201 acres (81.34 ha); most of this area (~63 ha) is dunes and coastal strand habitat, plus ~18ha of spoils area with little vegetation.

Fig. 10. General area for SDP subpopulation; yellow=public; blue=private land; star=inlet.



A Status Quo (SQ) *VORTEX* model was developed for the SDP SEBM subpopulation to represent the best available estimates of current and future conditions, assuming there are no changes in current management. The SQ model enabled projections of future viability under these conditions, sensitivity testing regarding the relative contribution of various threats to future viability, and exploration of alternative management scenarios and their impact on viability. All model scenarios began in January 2020 and were run for 80 years (i.e., to 2100). Each scenario was run for 500 iterations.

Model Inputs

The SEBM demographic base model was used, which includes seasonal fluctuations in sex- and age-specific reproduction and survival (see earlier descriptions of this model). The following information summarizes the subpopulation-specific inputs and revisions (see Appendix B for summary table of model inputs and details on the derivation of habitat- and catastrophe-related inputs).

Initial carrying capacity (K)

Expert opinion based on PVA team discussions estimated SEBM density in dune habitat at 7.5 mice/ha x 63 ha, which would be 473. It was recognized that the "spoils" area supports a few mice where the hardpan has been broken up (e.g., by tortoises), but that this area is primarily used as routes and that mice generally do not reside there. After further discussion, the team decided to apply a density of 5.86 mice/ha across the entire 81.34 ha (including the 18 ha of spoils) for the SQ model. This results in **initial K = 477**. Dune and spoils habitat were not tracked separately in the model.

Initial population size

The SQ model begins in early January, when the mouse population is projected to average about 85% of carrying capacity. This is based on stochastic runs of the base model with seasonal demographic rates, resulting in a mean N/K of 85%. The resulting **initial population size of 405** (i.e., 477 x 0.85) was distributed across age and sex classes based on a stable distribution using the base model winter demographic rates. Full juvenile mortality rates (including mortality due to death of dam) were applied to generate this distribution across age classes starting at A=1 (4 weeks old).

Initial genetic status

Degner *et al*. (2007) concluded that the proportions of heterozygous individuals and allelic richness were higher in SEBM on the Cape Canaveral Complex (CC) as compared to SDP, indicating reduced genetic diversity within SDP, consistent with founder effects and/or genetic drift. Kalkvik *et al.* (2018) also found lower genetic diversity in SDP. Recent analyses by Austin (2021; unpublished data) confirmed that there is less genetic variability in SDP (i.e., higher kinships among SDP mice than among CC mice) and more inbreeding relative to CC (but no more than would be expected with random mating). The base demographic rates were taken from the slightly inbred CC subpopulation. In order for *VORTEX* to generate the correct demographic rates for SDP with its higher level of inbreeding, we rescaled current inbreeding in SDP relative to that in CC. Specifically, we used the mean kinship among SDP mice – mean inbreeding among CC mice to estimate the increase in inbreeding in SDP relative to CC (i.e., 0.179 – 0.092 = 0.087). The value of 0.087 was used to represent initial kinships and initial inbreeding (F) in the SDP population. This method may slightly overestimate gene diversity in SDP but should work well to estimate the relative inbreeding impacts in SDP vs CC.

Inbreeding impacts

Inbreeding effects are applied in *VORTEX* as lower juvenile survival, parameterized as lethal equivalents (LE). Brewer *et al*. (1990) calculated 5.52 LE in SEBM under laboratory conditions; this estimate was reanalyzed by Lacy *et al*. (1996) to account for probability of breeding, resulting in a new estimate of 4.72 LE. Laboratory conditions are relatively benign compared to challenges faced by species in their natural environment, resulting in greater inbreeding effects in the wild (Frankham *et al*. 2010). O'Grady *et al*. (2006) estimate a combined mean effect of inbreeding on fecundity and first year survival for wild vertebrate populations to be 6.29 LE (default value in *VORTEX*). This value was used in sensitivity testing of the base model and was found to have an impact on all population sizes tested (K = 50 to 1000), with smaller SEBM populations impacted more quickly and leading to population decline and higher probability of extinction.

The PVA team chose to use the more conservative laboratory estimate of 4.72 LE for modeling the wild SEBM subpopulations. While wild populations are generally thought to show greater inbreeding effects than those of the same species under laboratory conditions, the stochastic nature of beach mouse population dynamics may have purged some of the effects in the wild. Given the historical isolation of the SDP subpopulation and subsequent inbreeding, the model will assume that all lethal alleles have been purged from this small, closed population. Thus, the SQ model for SDP was set to 2.36 LE (with 0% due to lethal alleles). This is believed to be a conservative estimate of anticipated inbreeding effects. No additional inbreeding impacts were added to survival of other age classes or to reproductive rates.

Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in SDP was estimated using a combination of projections for sea level rise (SLR) and high tide flooding (HTF) from NOAA in 2017 (as reported in the 2020 draft SSA) and in 2022 (provided by PVA team), GIS analysis, and extrapolation. Remaining SEBM habitat in Smyrna Dunes was modeled based on the High SLR and Moderate HTF scenarios for Trident Pier for the years 2050, 2080 and 2110. Loss of SEBM habitat in years (and time steps) between these point estimates was extrapolated, assuming that the loss will be linear between point estimates (see Appendix for calculation details and assumptions). K was calculated based on the density of 5.86 mice/ha x remaining

ha of habitat (Figure 10). Note that the *VORTEX* model is not spatially explicit. The subpopulation is assumed to be panmictic and not fragmented. **By 2100, about 31% of SEBM habitat is estimated to be lost, reducing K to 329 mice**. Figure 11 illustrates how this projection of habitat loss is driven by HTF (estimated at 2.76ft, leaving ~74ha of SEBM habitat, with K~=433) until about 2063, when SLR overtakes HTF estimates and drives further habitat loss.

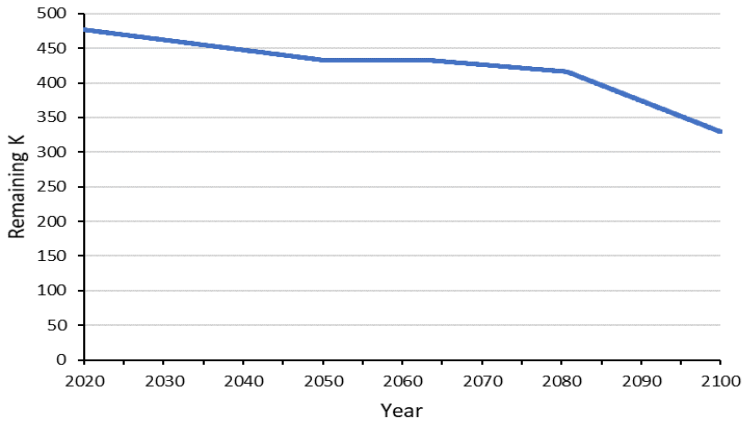


Fig. 11. Estimated K for SEBM SDP subpopulation over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110.

Impacts of storms

SDP is estimated to have the same risk of experiencing a hurricane (annual risk of 4%, occurring in summer and fall) or Nor'Easter storm (annual risk of 4%, occurring in March-April) as estimated for other SEBM dune habitat. The PVA team estimated the following impacts if a hurricane or Nor'Easter occurs: 1) 20% mortality of mice during the initial timestep; 2) no successful reproduction (i.e., births) during the initial or next timestep; 3) loss of habitat (K) for SEBM to 40% of pre-storm conditions; and 4) recovery of the habitat (K for SEBM) to 70% of pre-storm conditions in 5 years, with no further recovery. The SQ model does not allow the habitat (K) to recover above that available due to SLR and HTF.

Other threats/management

Beach erosion is not included in the SQ model for SDP, as it is assumed to be offset or repaired by beach renourishment in this area. Invasive species impacts and management (e.g., cogongrass, free-ranging cats) are assumed to remain the same as in 2020 at the onset of the projection.

Status Quo Model Results

Like the base demographic model, the Status Quo model for the SDP beach mouse subpopulation displays seasonal fluctuations in population size. In addition, demographic variation, environmental variation and other stochastic processes contribute to fluctuations in demographic rates and population size that are evident in this relatively small population. Figure 12 illustrates the seasonal dynamics of mean population size, as well as the standard deviation around that mean, for the first five years of the model (January 2020-December 2024). For this scenario, no catastrophe (major storm) was added. This seasonal pattern is a reasonable representation of beach mouse population biology.

The Status Quo model combines the impacts of demographic and genetic stochastic processes on the small, isolated SDP subpopulation with loss of the habitat's carrying capacity for SEBM due to progressive sea level rise (High SLR and Moderate HTF scenarios; see Figure 11) and major storms. These factors have been parameterized as accurately as possible based on best available data and expert

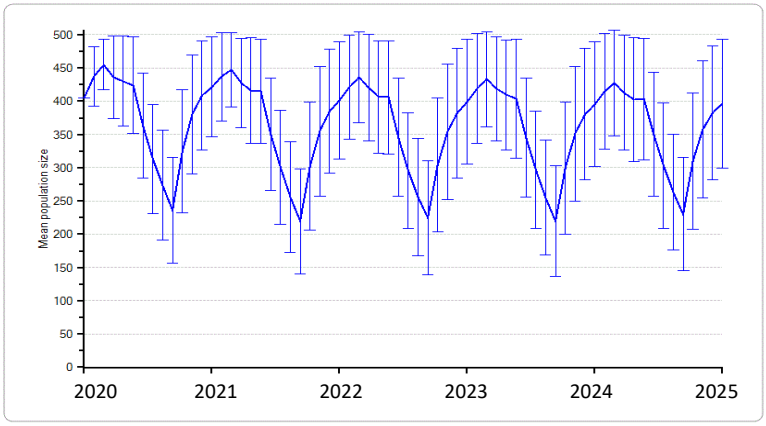


Fig. 12. Mean population size for the first five years of the Status Quo model. Bars represent ± 1 SD.

opinion; however, many of these parameters are not well understood. The model results are influenced by the model inputs and assumptions. **Thus, rather than focusing on precise estimates of future population projections, this model should be viewed as a tool to investigate the primary driving factors influencing viability. This can help us to understand the degree of uncertainty in model projections, identify the most important data gaps (and potential research needs), and help to prioritize management actions that are most likely to be effective.**

Given the model assumptions and inputs, the resulting projection over 80 years for SDP is overall population decline and eventual extinction. Figure 13 shows the probability of population persistence over time. There is 6.5% chance of extinction by 2040. Average time to extinction is 35 years (447-455 timesteps, ts), and extinction is essentially certain within 65 years.

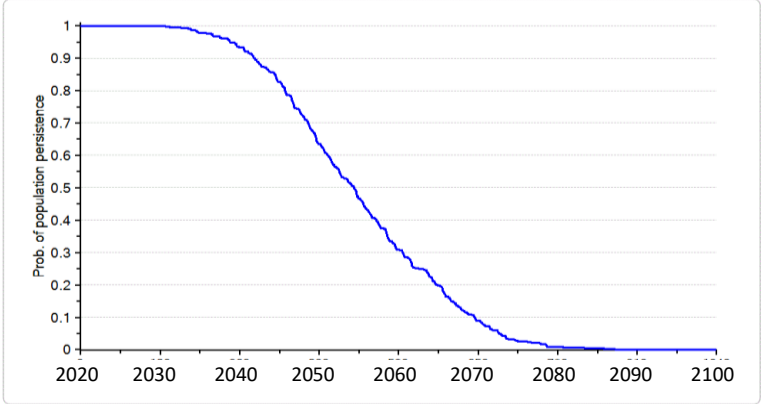


Fig. 13. Probability of persistence (i.e., at least one mouse of each sex) for the SDP subpopulation over 80 years.

Driving extinction is population decline. Mean population size decreases over time. Figure 14 demonstrates seasonal variation in mean population size, while Figure 15 show the mean population size for December (each 13th timestep) and the standard deviation around these means. One yearly census (in December) is plotted for illustrative purposes. A similar trend (decline and variations around the mean) is seen for other timesteps, with lower mean population size in summer. Both seasonal variation and stochastic variation among iterations contribute to the significant variation in population size at any one timestep or calendar year, as illustrated in Figure 16.

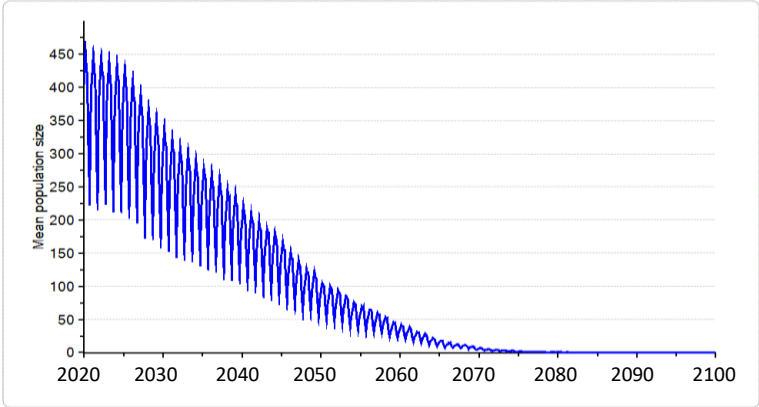


Fig. 14. Mean population size for the SDP subpopulation over 80 years. Each timestep (ts) is plotted, demonstrating seasonal variation in mean population size.

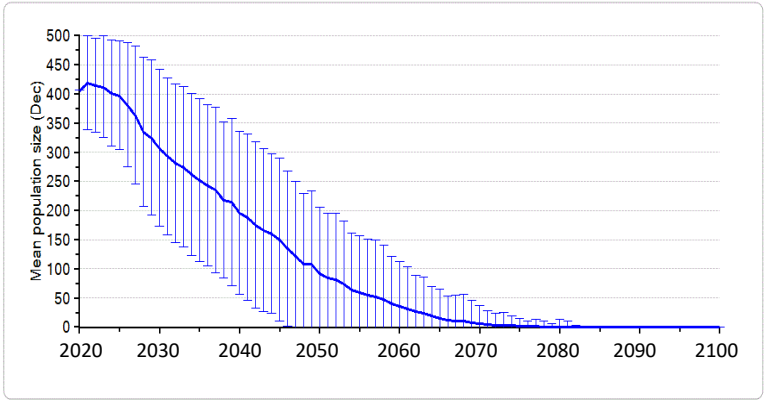


Fig. 15. Mean population size in December for the SDP subpopulation over 80 years. Each 13th ts is plotted, demonstrating variation in mean population size in December. Bars represent ± 1 SD.

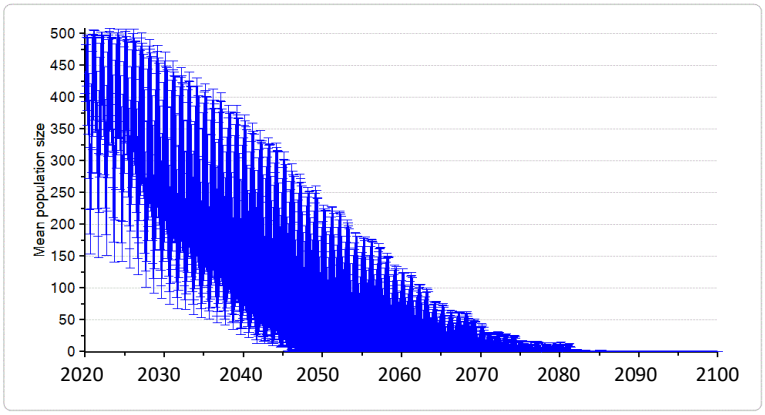


Fig. 16. Mean and standard deviation (error bars) in population size for the SDP subpopulation over 80 years.

In order to improve visual clarity in graphing SD and for comparisons across different scenarios, subsequent graphs for mean population size, gene diversity and inbreeding, starting with Figure 17, plot every 13th timestep (i.e., December means).

Smaller population size increases the rate of loss of gene diversity (Figure 17) and the accumulation of inbreeding (Figure 18), such that mean inbreeding reaches 25% in 20 years. The odd patterns for these means in later years are artifacts of only a few persisting simulations, usually from the less inbred iterations, and containing only a small number of mice. Means are only calculated from persisting iterations and thus go to zero once all iterations have gone extinct.

Taken together, the **Status Quo results demonstrate a declining population that loses gene diversity and accumulates inbreeding quickly** due to short generation time, increasingly smaller population size, and lack of connectivity with other SEBM populations.

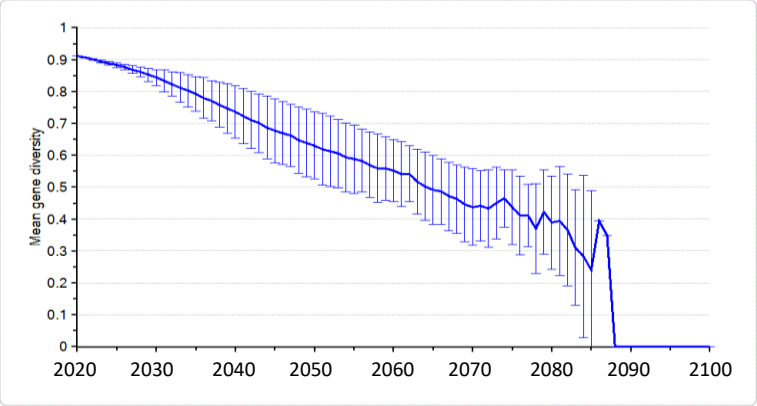


Fig. 17. Mean gene diversity for the SDP subpopulation over 80 years (each 13th ts is plotted, representing means for December). Bars represent ± 1 SD.

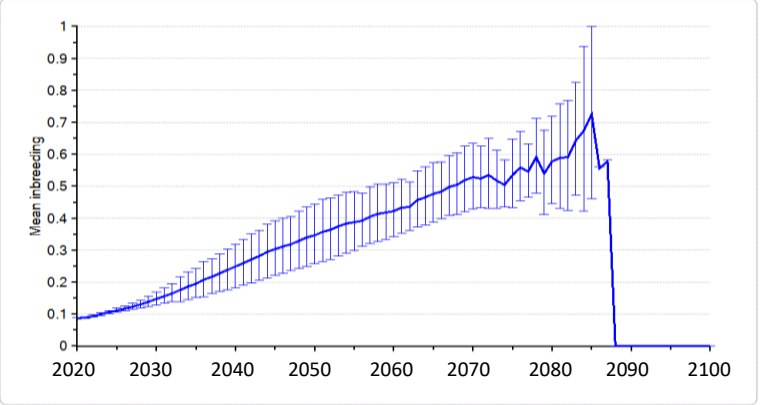


Fig. 18. Mean inbreeding coefficient for the SDP subpopulation over 80 years (each 13th ts is plotted, representing means for December). Bars represent ± 1 SD. F=0.25 is equivalent to full-sibling mating, while F=0.5 is equivalent to self-fertilization.

Driving Factors in SQ Projections

Several concurrent factors impact the SEBM population, both in the model and in the wild. Several exploratory scenarios were developed to isolate the effects of some of these factors in order to better understand how they are impacting the SQ projections. The following alternative scenarios were run:

1) All threats (SQ): Includes inbreeding depression; loss of K due to SLR and HTF; all storm impacts
2) SLR removed: includes storms and inbreeding depression only
3) Storms removed: includes SLR & HTF and inbreeding depression only
4) Inbreeding removed: includes SLR & HTF and storms only
5) All 3 threats removed: includes only effects of demographic and environmental variation

These five scenarios isolate the impacts of sea level rise, major storms and inbreeding. All scenarios include demographic variation (inherent to the model and a product of population size) and environmental variation (parameterized by EV for reproduction and mortality).

Figure 19 shows the probability of population persistence over time for each scenario, while Figure 20 provides the mean population size (SD is not provided on the graph to improve visual clarity for comparisons). These results show good long-term population viability if the impacts of major storms, SLR&HTF and inbreeding are removed (black line). In this case, the population maintains its size with no risk of extinction over 80 years. This indicates that model results are not significantly impacted by either demographic variation or annual environmental variation. Poor viability seen in the model results is a result of some combination of the three threats of inbreeding depression, SLR&HTF and major storms.

Removing the loss of habitat due to SLR&HTF has very little effect on the model results. Probability of persistence and mean population size are very similar between the SQ scenario (blue line) and the same scenario but with SLR&HTF impacts removed (red line). This leaves **major storms and inbreeding depression as the primary drivers of SDP subpopulation viability in the Status Quo model**.

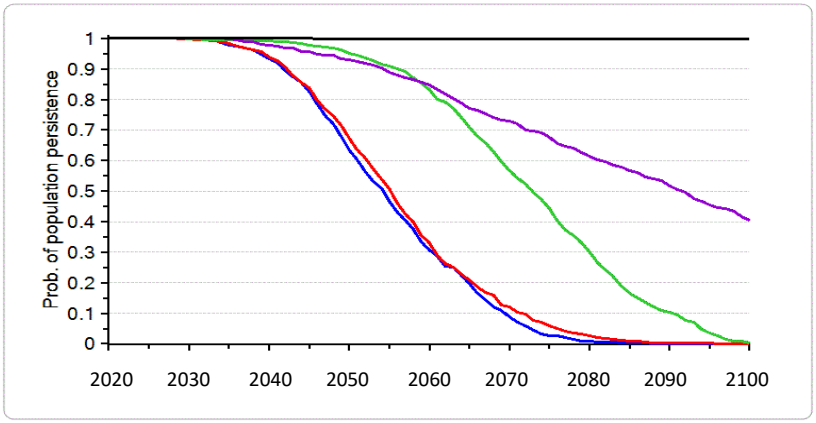


Fig. 19. Probability of persistence for the SDP subpopulation over 80 years for five scenarios that include or exclude different threats (SLR, major storms, inbreeding depression).

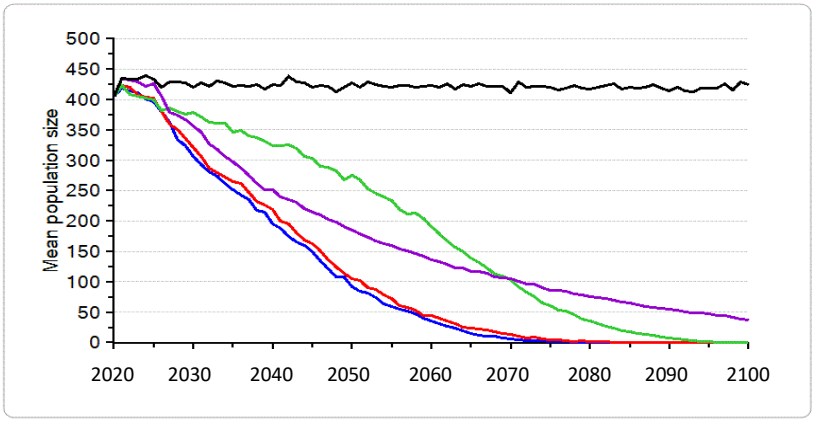


Fig. 20. Mean population size in December for the SDP subpopulation over 80 years for five scenarios that include or exclude different threats (SLR, major storms, inbreeding depression). Each 13th ts is plotted, representing means for December.

Inbreeding depression

Inbreeding is the factor that has the largest impact on population persistence in the model. Without inbreeding depression (Figure 19, purple line), populations have a 60% risk of extinction in 80 years, as opposed to 100% for all scenarios that include inbreeding depression. Although simulations without inbreeding depression have a greater chance of persisting, the population still declines quickly (Figure 20). Gene diversity is still lost and inbreeding is still accumulating, but these small inbred populations are not pushed as rapidly to extinction without the negative consequences of inbreeding depression.

The precise sensitivity of the SDP subpopulation to inbreeding depression (i.e., lethal equivalents) is uncertain. The SQ model input value of 2.36 LE (with no lethal alleles) represents a realistic and potentially conservative estimate of inbreeding sensitivity for SDP. Exploration of higher (4.72 LE) and lower (1.18 LE) values confirms that the degree of inbreeding depression has corresponding impacts on population size and extinction risk over time (Figures 21&22). While actual LE is not known, it is probable that inbreeding depression contributes negatively to long-term viability of the small, isolated SDP subpopulation but is **not the sole factor responsible for decline and high extinction risk**.

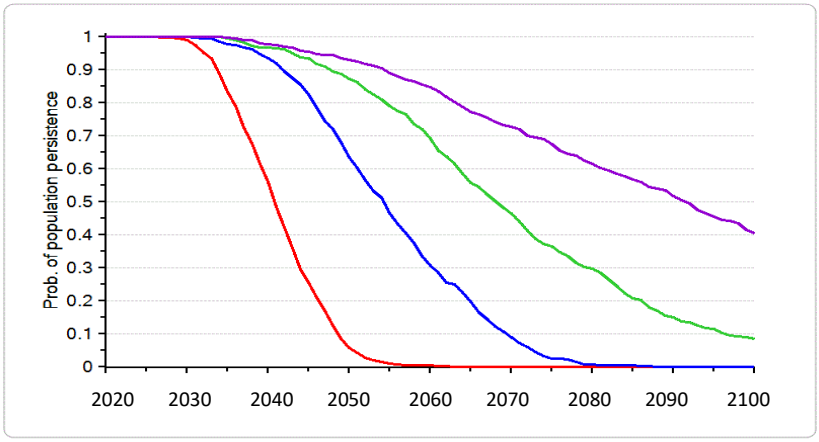


Fig. 21. Probability of persistence for the SDP subpopulation over 80 years with various inbreeding impacts (LE=4.72, 2.36, 1.18 and 0).

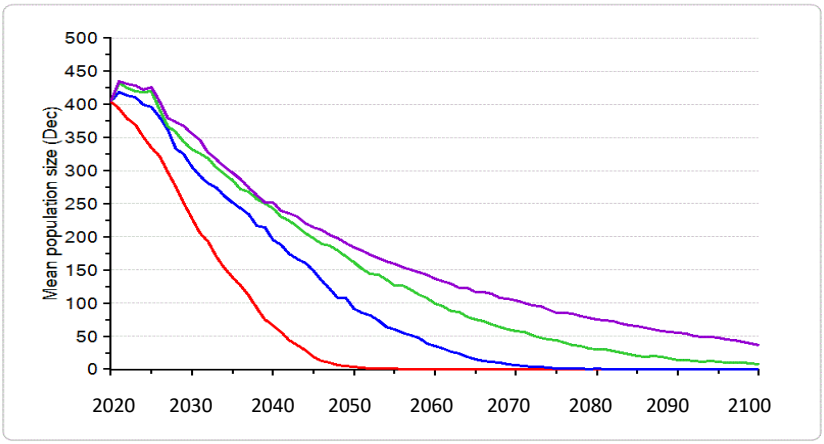


Fig. 22. Mean population size in December for the SDP subpopulation over 80 years with various inbreeding impacts (LE=4.72, 2.36, 1.18 and 0). Each 13th ts is plotted, representing means for December.

Impact of major storms

The other significant factor affecting long-term SDP subpopulation viability in the SQ model is the impacts of major storms. The population declines more slowly in the absence of storms (Figure 20, green line), although extinction risk increases after 40 years (Figure 19), likely due to inbreeding depression. Two types of storms are included in the SQ model – hurricanes in summer/fall, and Nor'Easters in spring. Each storm type was given the same annual risk of occurrence and same impacts. Impacts include demographic effects (direct mortality and temporary cessation of reproduction) and habitat effects (loss of K, with partial recovery). These impacts were examined in exploratory scenarios to assess the effect of demographic vs habitat impacts, and the effect of season of storm occurrence.

Removing the demographic impacts of storms (i.e., direct mortality and two-month cessation of reproduction) has essentially no impact on probability of persistence (Figure 23) or mean population size (Figure 24) in the SQ model (green line). On the other hand, removing the impacts on habitat (K) does affect viability (red line). Hurricanes and Nor'Easters have similar impacts, suggesting that seasonality of storm effects is not a primary factor. Beach mice are adapted to this dynamic environment with a high intrinsic biological growth rate that allows the species to grow quickly when resources are available.

The primary aspect of major storms influencing viability in the SQ model is the loss of habitat. Model input values produce a 40% reduction in K when a major storm occurs, and only allows the SEBM habitat to recover to 70%. On average, the frequency of storms combined with these reductions in K mean that habitat loss (i.e., reduction in K for SEBM) is driven by storms, not by SLR and HTF, in the SQ model (Figure 25).

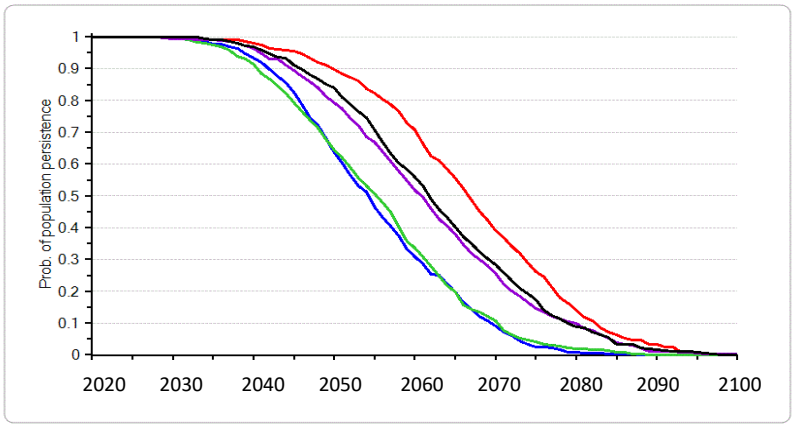


Fig. 23. Probability of persistence for the SDP subpopulation over 80 years for five scenarios that include or exclude different aspects of major storms.

All storm impacts (SQ)
K impacts removed
Dem impacts removed
NE storms removed
Hurricanes removed

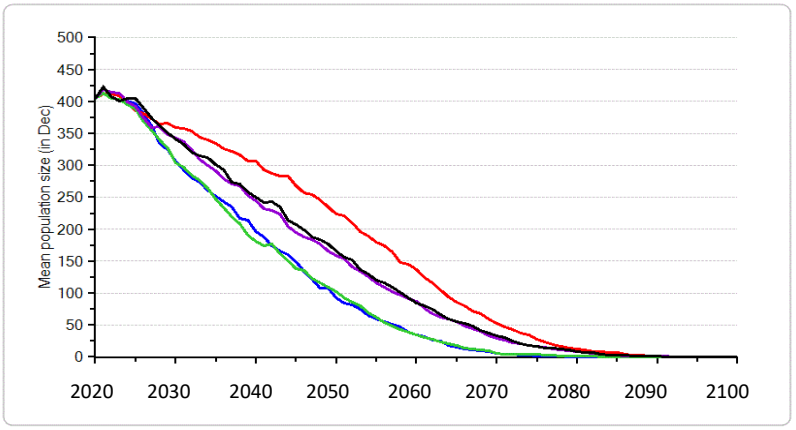


Fig. 24. Mean population size in December for the SDP subpopulation over 80 years for five scenarios that include or exclude different aspects of major storms. Each 13th ts is plotted, representing means for December.

All storm impacts (SQ)
K impacts removed
Dem impacts removed
NE storms removed
Hurricanes removed

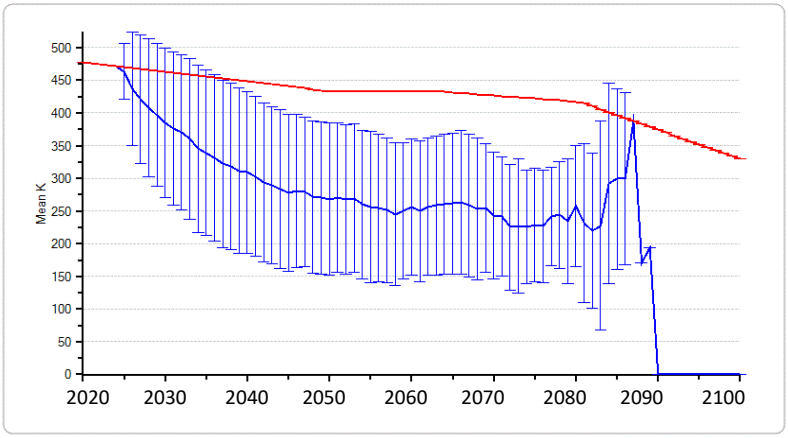


Fig. 25. Mean carrying capacity (K) for the SDP subpopulation over 80 years as imposed by SLR/HTF (red line) and due to storms (bars represent ± 1 SD). Storm events were stochastic in the model while SLR/HTF was modeled as a deterministic impact.

Implications: uncertainty, knowledge gaps, and management options

From the above exploratory analysis, we can infer the following. First, we need not be too concerned that either environmental variation of demographic rates, or the impact of storms on these rates, are overestimated in the model (although they may be underestimated). These parameters are not high priority data gaps in terms of assessing SEBM population viability, nor is management to mitigate these effects a priority. Other areas are more important to focus on.

Inbreeding depression, as modeled, is a driving factor for the long-term viability of the SEBM SDP subpopulation. Modeling projects that mean inbreeding will rise to the level of full-sibling mating after about 20 years and to about F=0.5 (representing self-fertilization) in about 50 years. Better understanding of inbreeding sensitivity and impacts in SEBM would be useful in developing the most effective and efficient management strategy. Even without additional information, however, we know that inbreeding levels will increase in this isolated population, with a corresponding reduction in genetic variation that fuels adaptive potential (an effect that is not included in this model). The SQ model incorporates estimated inbreeding impacts that are guided by genetic data and expert opinion and are conservative with respect to most wild vertebrate populations. Thus, while there is some uncertainty around the precise impacts of inbreeding depression in timing and strength, it is likely that **genetic augmentation may be an important strategy** to reduce inbreeding and its impacts while also offsetting genetic drift.

Loss of habitat (and the resulting reduction in K for beach mice) is an important concern in the SQ model. The current parameterization of the loss of habitat due to storms overrides the projections for SLR and HTF given the model assumptions, although either threat drives the subpopulation smaller and increases its vulnerability to inbreeding depression and other stochastic impacts. The model inputs for storm impacts were less data driven than SLR-HTF projections and may be an important source of uncertainty in SQ model projections. Better understanding of the projected impacts of storms on SEBM habitat in SDP would improve viability projections for this subpopulation.

Management options to promote habitat restoration following storms may be more feasible, and possibly more effective, than options to mitigate sea level rise. Any management efforts to restore, maintain or improve habitat quality for SEBM will promote the viability of this small, isolated subpopulation by allowing the population to remain as large as possible, which in turn will slow genetic drift and the accumulation of inbreeding.

Alternative Management Scenarios

Genetic Augmentation Scenarios

Several alternative management scenarios were developed to explore the potential impacts of genetic augmentation on the SEBM SDP subpopulation. All scenarios included the periodic release of subadult (4 week old) mice, of equal sex ratio, with releases occurring in October, per PVA team discussions. Translocated mice were modeled as unrelated to the SDP subpopulation and are assumed to be obtained from the large Beach 2 (Cape beach area) wild subpopulation. Releases of 30-50 mice were suggested as numbers that are both feasible and potentially effective.

Little information exists on the survival and reproduction of beach mice that are translocated between wild populations. To address this uncertainty, the following strategy was followed. No translocation mortality is included in the model, and the model assumes that translocated mice have the same mortality and reproductive rates as resident mice. Essentially this means that the number of releases in the model reflect the number of *successful* releases. The number of actual mice needed to be released to achieve this level would need to be calculated based on the success (survival and reproduction) observed for actual translocations.

No outbreeding effects were included in the translocation scenarios. While this is another area of uncertainty, it is hypothesized that outbreeding is not likely to be a main concern given the historical connectivity of these populations in the past. There are some laboratory data that indicate that crosses between *P.p.* subspecies may even show strong hybrid vigor instead of outbreeding depression (Lacy, pers. comm.).

Scenarios with translocations only in the initial years or decades (years 2025 to 2045) show some genetic benefit during the translocation period (with a corresponding decrease in extinction risk); however, these benefits quickly disappear once translocations cease. Populations still decline, mean inbreeding still rises to F=0.5, and extinction risk remains 100% within 80 years, with little improvement on long-term viability.

Two scenarios were developed to explore periodic, continual translocation. Each modeled an initial release of 50 mice (in 2025), followed by releases of 30 mice every 5 or every 10 years. Periodic augmentation maintains higher gene diversity (Figure 26) and controls the accumulation of inbreeding (Figure 27). However, the population still declines, and the risk of extinction before the next scheduled translocation is still relatively high (42-67% in the last decade modeled) (Figure 28). Poor long-term viability is still driven by the loss of habitat and maximum carrying capacity for beach mice even with genetic augmentation. This matches the earlier model scenarios that suggest that **removing inbreeding depression is insufficient to ensure long-term viability in SDP**.

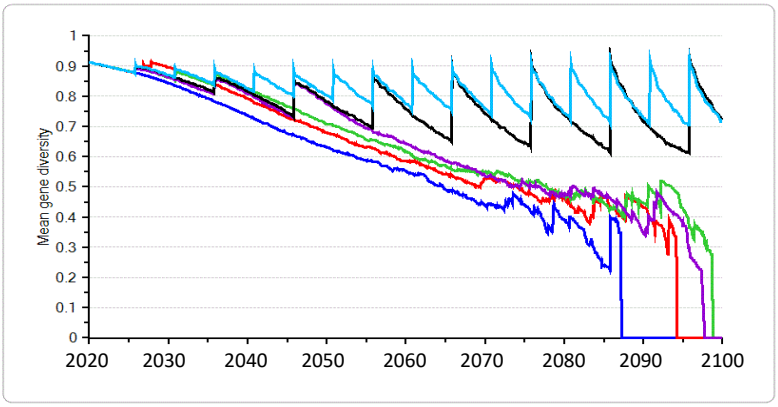


Fig. 26. Mean gene diversity for the SDP subpopulation over 80 years under five translocation scenarios. Tr#=# mice in initial release, followed by years in which releases occur. All initial releases occur in 2025 (Y6), followed by 30 mice in each subsequent release.

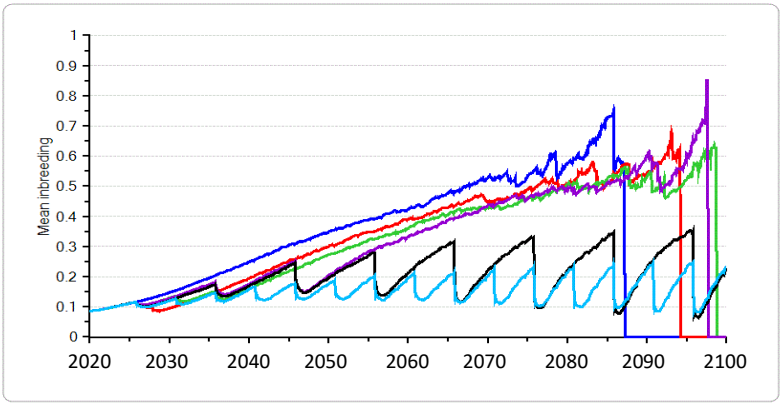


Fig. 27. Mean inbreeding coefficient for the SDP subpopulation over 80 years under five translocation scenarios. Tr#=# mice in initial release, followed by years in which releases occur. All initial releases occur in 2025 (Y6), followed by 30 mice in each subsequent release.

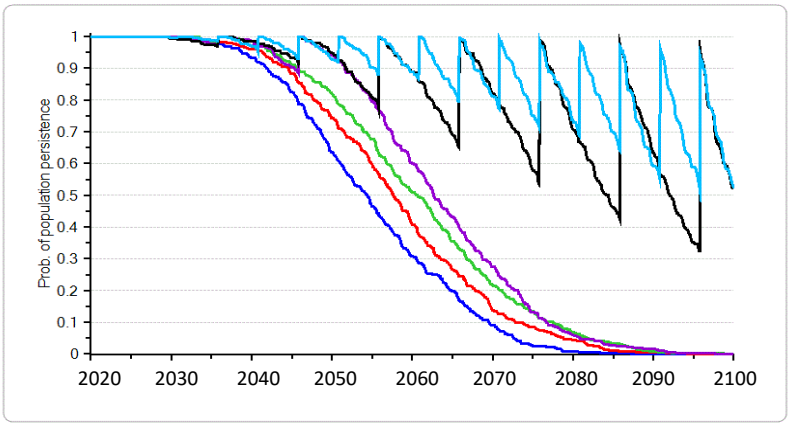


Fig. 28. Probability of population persistence for the SDP subpopulation over 80 years under five translocation scenarios. Tr#=# mice in initial release, followed by years in which releases occur. All initial releases occur in 2025 (Y6), followed by 30 mice in each subsequent release.

Habitat Recovery Scenarios

Reduction in K due to habitat loss caused by storms is an important driver of viability in the SQ model. It may be possible to improve the recovery of habitat quality and quantity for beach mice through habitat restoration. Three hypothetical habitat restoration scenarios were tested: 1) habitat recovery to 100% of pre-storm value in 10 years, which represents the same rate of recovery as the SQ model but continues until all habitat is recovered; 2) habitat recovery to 100% of pre-storm value in 5 years, representing the same recovery period as the SQ model but at twice the recovery rate; and 3) habitat recovery to 100% of pre=storm value in 3 years, simulating intensive nourishment to quickly increase the habitat's carrying capacity for SEBM. Habitat recovery never exceeded the available habitat after SLR and HTF.

These hypothetical post-storm habitat restoration models lead, on average, to higher SEBM carrying capacity but still resulted in loss of habitat significantly greater than that due only to SLR and HTF (Figure 29). Restoring SEBM habitat (K) completely in 3-5 years has a small benefit in mean population size and probability of persistence (median time to extinction is 7-9 years longer than for SQ scenario), while taking 10 years for full recovery results in no increase in viability (population size or persistence) (Figure 30). **Post-storm habitat restoration alone is insufficient for long-term viability in the absence of genetic augmentation**, given the inputs and assumptions in the SQ model.

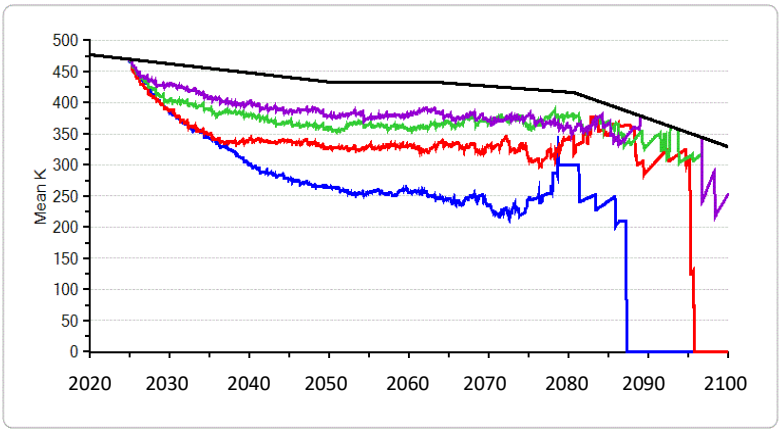


Fig.29. Mean carrying capacity (K) for the SDP subpopulation over 80 years with different post-storm habitat recovery scenarios. Black line represents K based only on SLR/HTF.

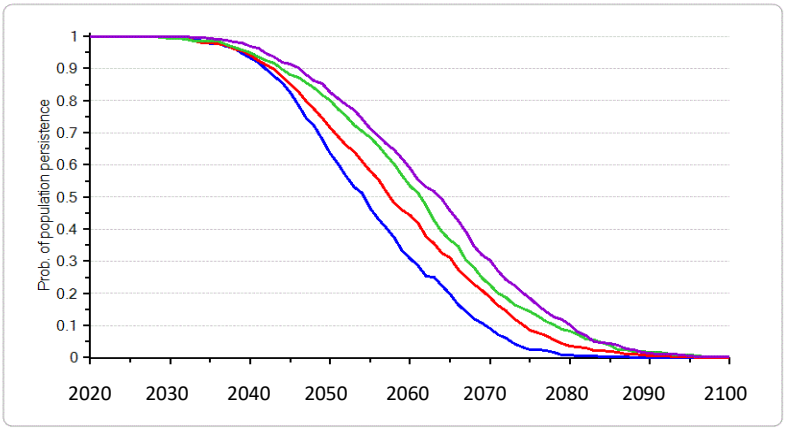


Fig.30. Probability of population persistence for the SDP subpopulation over 80 years with different post-storm habitat recovery scenarios. Black line represents K based only on SLR/HTF.

Mixed-Strategy Scenarios

A series of scenarios were developed that combined translocation and post-storm habitat restoration to simultaneously address both key drivers of population viability in the SQ model. All three habitat restoration schemes were tested (100% post-storm recovery of K in either 3, 5 or 10 years). Two translocation schemes were included: an initial release of either 50 or 90 mice (in 2025) followed by the release of 30 mice every 5 years (i.e., 2035, 2045, etc.). The six possible combinations of these schemes were modeled and compared to the SQ model projections.

Probability of persistence is much improved with all six mixed-strategy scenarios (Figure 31), with slower decline and larger mean population size over time (Figure 32). Faster habitat restoration (within 3-5 years) leads to better viability, possibly a result of the larger mean K over time (Figure 33). The patterns of persistence probability suggest that slower habitat restoration (10 years) increases the chance that SEBM may go extinct following a storm and later be re-established with the next translocation. A larger initial release (90 vs 50 mice) has no discernible benefits on population size or persistence. All four schemes maintain greater gene diversity (76-90%) (Figure 34) and controlled inbreeding at or below 20% (Figure 35).

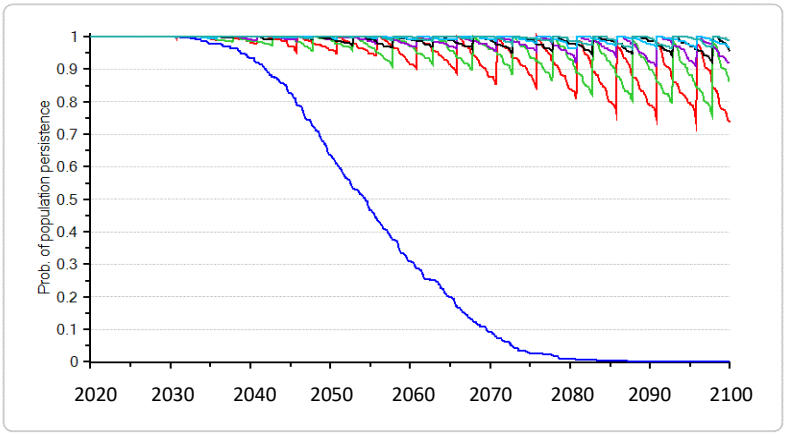


Fig. 31. Probability of population persistence for the SDP subpopulation over 80 years for six mixed-strategy management scenarios. Blue line represents SQ scenario with no additional management.

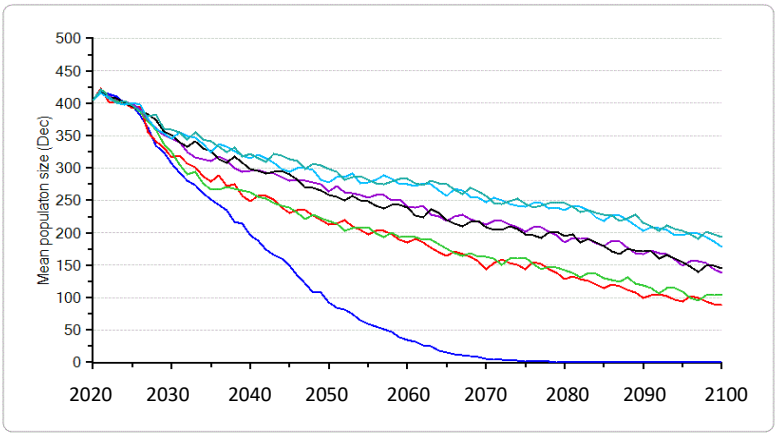


Fig. 32. Mean population size in December for the SDP subpopulation over 80 years for six mixed-strategy management scenarios. Blue line represents SQ scenario with no additional management.

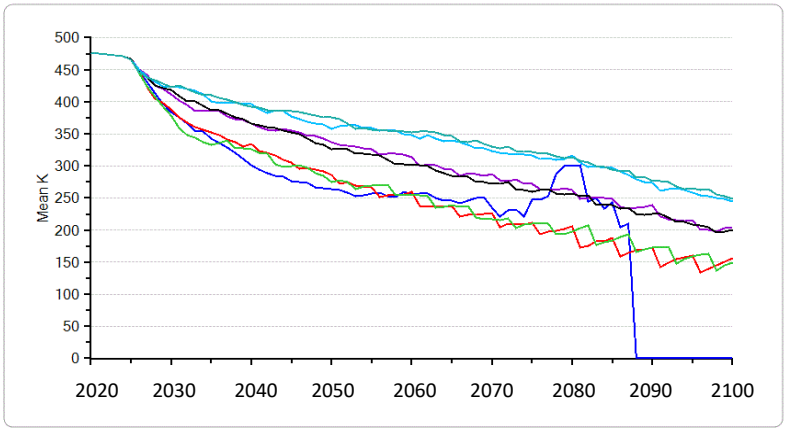


Fig. 33. Mean carrying capacity (K) for the SDP subpopulation over 80 years for six mixed-strategy management scenarios. Blue line represents SQ scenario with no additional management.

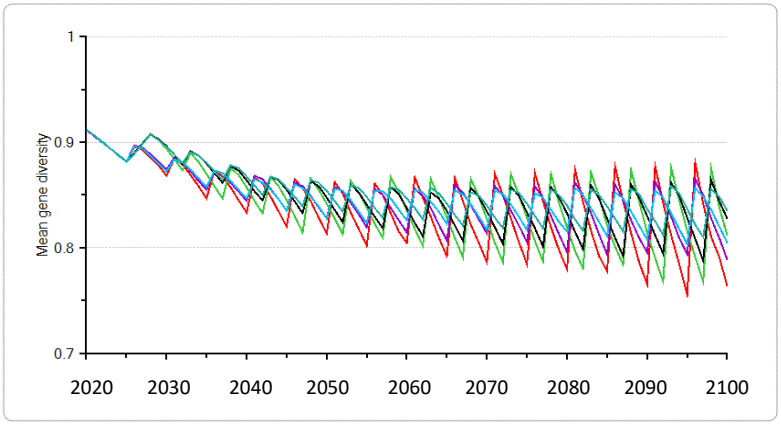


Fig. 34. Mean gene diversity for the SDP subpopulation over 80 years for six mixed-strategy management scenarios. SQ scenario not shown.

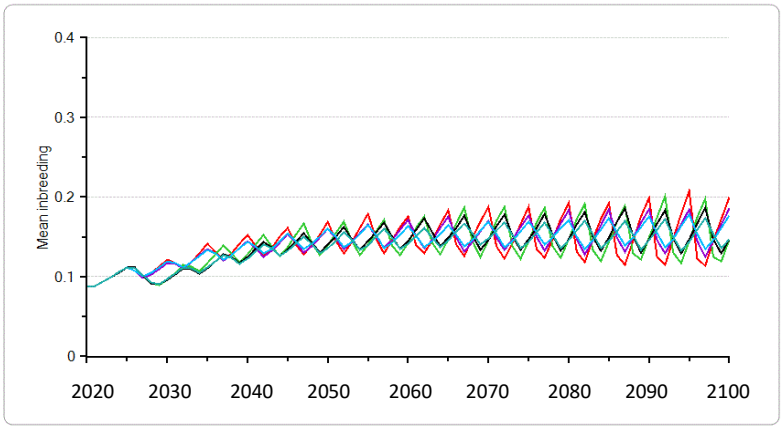


Fig. 35. Mean inbreeding coefficient for the SDP subpopulation over 80 years for six mixed-strategy management scenarios. SQ scenario not shown.

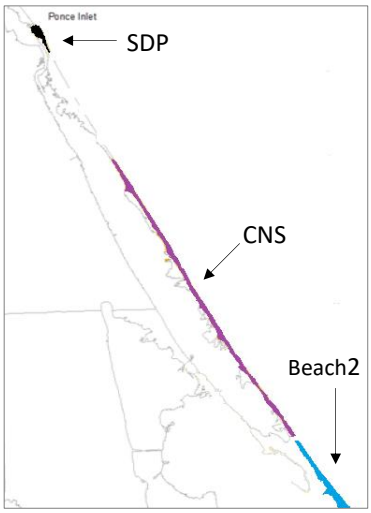
Summary

A Status Quo model for the SEBM SDP subpopulation was developed based on the best available data and expert opinion. Given model inputs and assumptions, the projected long-term viability of SEBM in SDP is poor, with population decline and eventual extinction. The key driving factors affecting this projection are inbreeding depression (due to small population size and genetic isolation) and habitat loss (especially due to storms that lower the carrying capacity for mice beyond the habitat loss caused by sea level rise and high tide flooding). The greatest uncertainty surrounding these projections is around the impact of storms on SEBM habitat and the recovery of that habitat, followed by uncertainty in the sensitivity of SEBM to inbreeding depression. Modeling of alternative management scenarios suggest that effective management may need to be two-fold: prolonged periodic genetic augmentation, in concert with effects to restore, maintain and/or increase SEBM habitat and its carrying capacity to support beach mice. While other habitat management actions were not explicitly modeled, any efforts to increase K for beach mice are likely to be beneficial, including post-storm habitat restoration as well as other potential habitat management strategies. Exploratory translocation efforts may help to identify the most effective release strategies and fill in data gaps on post-release survival and reproduction. While uncertainty limits the precision of these modeling results, they provide insight into future SEBM SDP trajectories under current and alternative future management strategies. Given the vulnerability of this potentially genetically differentiated isolated subpopulation, the PVA team also recognized that careful and strategic translocation of SEBM from SDP to other vulnerable dune habitats in need of augmentation may be an option to reduce the risk of loss of genetic diversity represented by the SDP subpopulation and promote Redundancy for the subspecies. Evaluation of the potential benefits and risks of such a strategy could be addressed through future additional population modeling.

CANAVERAL NATIONAL SEASHORE SOUTHEASTERN BEACH MOUSE SUBPOPULATION

The Southeastern beach mouse (SEBM) inhabits the narrow strip of coastal dunes along the Canaveral National Seashore (CNS), which represents the northern subpopulation of the Cape Canaveral Complex (CC) metapopulation (Figure 36, adapted from 2020 SSA). Once contiguous with the Smyrna Dunes region decades ago, the CNS subpopulation is now separated from the SDP subpopulation to the north by heavy development. Connectivity to the south is limited to a narrow corridor near the boundary between CNS and Merritt Island National Wildlife Refuge (MINWR), which provides some connectivity with the larger SEBM subpopulation along the Cape Canaveral coast (Beach2). This narrow CNS dune habitat, behind which are wetlands unsuitable for beach mice, is vulnerable to impacts such as beach erosion, storms and sea level rise. Habitat for the CNS beach mouse subpopulation consists of 238 acres (96.4 ha) of coastal dune habitat, as reported in the 2020 SSA (USFWS 2020). Habitat quality for SEBM varies across CNS and may be lower in the north.

Fig. 36. General location of CNS subpopulation.



A Status Quo (SQ) *VORTEX* model was developed for the CNS SEBM subpopulation to represent the best available estimates of current and future conditions, assuming there are no changes in current management. The SQ model enabled projections of future viability under these conditions, sensitivity testing regarding the relative contribution of various threats to future viability, and exploration of alternative management scenarios and their impact on viability. All model scenarios began in January 2020 and were run for 80 years (i.e., to 2100). Each scenario was run for 500 iterations.

Model Inputs

The SEBM demographic base model was used, which includes seasonal fluctuations in sex- and age-specific reproduction and survival (see earlier descriptions of this model). The following information summarizes the subpopulation-specific inputs and revisions (see the Appendix B for summary table of model inputs and details on the derivation of habitat- and catastrophe-related inputs).

Initial carrying capacity (K)

Discussions by the PVA team recognized some uncertainty regarding the average density of SEBM across the CNS habitat. The team recommended testing two densities, 5 mice/ha (D5) and 7 mice/ha (D7), which result in an estimated initial K of 482 and 675, respectively based on 96.4 ha in 2020.

Initial population size

The SQ model begins in early January, when the mouse population is projected to average about 85% of carrying capacity. This is based on stochastic runs of the base model with seasonal demographic rates, resulting in a mean N/K of 85%. The resulting initial population size of 410 and 574, respectively was distributed across age and sex classes based on a stable distribution using the base model winter demographic rates. Full juvenile mortality rates (including mortality due to death of dam) were applied to generate this distribution across age classes starting at A=1 (4 weeks old).

Initial genetic status

Recent analyses by Austin (2021; unpublished data) found no significant genetic differentiation in the SEBM population across the CC. Since the base demographic rates were taken from this slightly inbred CC subpopulation, these rates account for any historical inbreeding impacts. Thus, initial kinships and initial inbreeding (F) were set to 0 in the CNS population (i.e., the current genetic state of the population is set as the baseline for modeling future declines in genetic diversity and impacts of any additional inbreeding). Additional future inbreeding in the model will be subject to inbreeding impacts, as described below.

Inbreeding impacts

Inbreeding effects are applied in *VORTEX* as lower juvenile survival, parameterized as lethal equivalents (LE). Brewer *et al*. (1990) calculated 5.52 LE in SEBM under laboratory conditions (using lab stock derived from founders captures in 1984 from an area in the middle of the CNS subpopulation. This estimate was reanalyzed by Lacy *et al*. (1996) to account for probability of breeding, resulting in a new estimate of 4.72 LE. Laboratory conditions are relatively benign compared to challenges faced by species in their natural environment, resulting in greater inbreeding effects in the wild (Frankham *et al*. 2010). O'Grady *et al*. (2006) estimate a combined mean effect of inbreeding on fecundity and first year survival for wild vertebrate populations to be 6.29 LE (default value in *VORTEX*). This value was used in sensitivity testing of the base model and was found to have an impact on all population sizes tested (K = 50 to 1000), with smaller populations impacted more quickly and leading to population decline and higher extinction risk.

The PVA team chose to use the more conservative laboratory estimate of 4.72 LE for modeling the wild SEBM subpopulations. While wild populations are generally thought to show greater inbreeding effects than those of the same species under laboratory conditions, the stochastic nature of beach mouse population dynamics may have purged some of the effects in the wild and so the laboratory value was not increased. These impacts were distributed across two mechanisms in the model using the default setting of 50% due to recessive lethal alleles and 50% due to lower survival of inbred juveniles. This is believed to be a conservative estimate of anticipated inbreeding effects. No additional inbreeding impacts were added to survival of other age classes or to reproductive rates.

Connectivity

Subpopulations designated as CNS and Beach2 in this PVA are connected by a narrow corridor near the boundary between CNS and MINWR. Dispersal across this boundary was calculated using the same methodology developed for the 2005 Alabama beach mouse (ABM) PVA (Traylor-Holzer *et al.* 2005). Mice are assumed to disperse from their natal area randomly in all directions, which means that some of the mice born within the average dispersal range of this boundary will cross into the adjacent subpopulation. Acreage within the average dispersal range, on each side of the boundary, was calculated based on GIS analysis, and divided by the total acreage for each subpopulation to determine the percent of each subpopulation within its respective dispersal zone. These percentages were applied to juvenile (4 week old) mice to calculate trans-boundary dispersal rates in each direction.

The large size of the Beach2 subpopulation prohibits running a CNS-Beach2 metapopulation model that includes kinship-based inbreeding depression. Therefore, CNS scenarios were run with CNS as a stand-alone population. The dispersal of juvenile mice across the boundary to Beach2 was modeled using

probabilistic harvest of a proportion of juveniles in each timestep (ts) (i.e., 0.736% of juveniles dispersing to Beach2/ts). To account for new SEBM immigrating from MINWR, results from the Beach2 status quo model (using 0.5-mile impact zones at 50% density) were used to calculate the average number of mice expected to disperse into CNS. This number varies seasonally (summer vs other timesteps) and over time as K declines, from a high of 1.6 mice/ts (Spring 2020) to a low of 0.65 mice/ts (Summer 2099). New mice were added to CNS using the Supplementation option, which assumes that new mice are unrelated to the CNS subpopulation. This is a reasonable assumption given the large Beach2 subpopulation.

Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in CNS was estimated using a combination of projections for sea level rise (SLR) and high tide flooding (HTF) from NOAA in 2017 (as reported in the 2020 SSA) and in 2022 (provided by PVA Team), GIS analysis, and extrapolation. Remaining SEBM habitat in CNS was modeled based on the High SLR and Moderate HTF scenarios for Trident Pier for the years 2050, 2080 and 2110. Loss of SEBM habitat in years (and time steps) between these point estimates was extrapolated, assuming that the loss will be linear between point estimates (see Appendix for calculation details and assumptions). K was calculated based on the density of 5 (or 7) mice/ha x remaining ha of habitat (Figure 37). Note that the *VORTEX* model is not spatially explicit. The subpopulation is assumed to be panmictic and not linear or fragmented. **By 2100, about 37% of SEBM habitat is estimated to be lost, reducing K to 303 and 424 mice, respectively**. Figure 37 illustrates how this projection of habitat loss is driven by HTF (estimated at 2.76ft, leaving ~87ha of SEBM habitat, with K~=435 and 609) until about 2063, when SLR overtakes HTF estimates and drives further habitat loss.

The 2020 SSA reports that the corridor connecting the CNS and Beach2 subpopulations will narrow by 2050 and will be broken by 2070 when the 2017 SLR projection is 4.46ft. The 2022 projections achieve this same level in 2080. Thus, the **SQ CNS model assumes loss of connectivity with the Beach2 subpopulation in 2080** (i.e., no emigration to Beach2 ("Harvest") and no immigration from Beach2 ("Supplementation")).

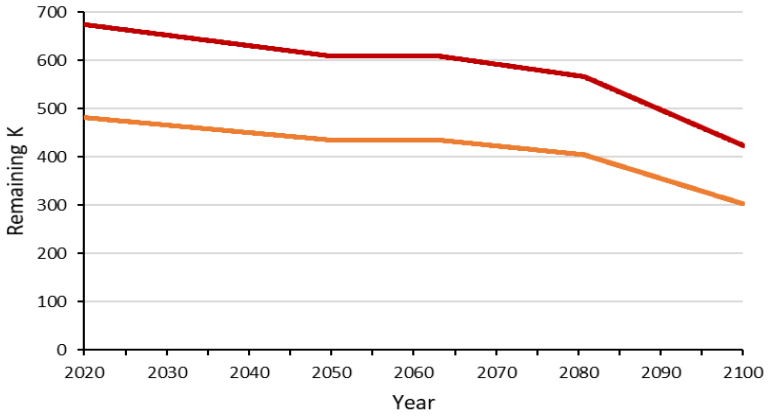


Fig. 37. Estimated K for SEBM CNS subpopulation over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110.

Impacts of storms

CNS is estimated to have the same risk of experiencing a hurricane (annual risk of 4%, occurring in summer and fall) or Nor'Easter storm (annual risk of 4%, occurring in March-April) as estimated for other SEBM dune habitat. The PVA team estimated the following impacts if a hurricane or Nor'Easter occurs: 1) 30% mortality of mice during the initial timestep; 2) no successful reproduction (i.e., births) during the

initial or next timestep; 3) loss of habitat (K) for SEBM to 40% of pre-storm conditions; and 4) recovery of the habitat (K) to 50% of pre-storm conditions in 5 years, with no further recovery. The SQ model does not allow the habitat (K) to recover above that available due to SLR and HTF.

In addition, the SQ model includes a 50% chance of permanent fragmentation with hurricanes. The SQ model assumes fragmentation into two equal-sized habitats (i.e., each with ½ K at the time of fragmentation). The southern fragment remains connected to Beach2 at the same rate as before the event, while the northern fragment becomes isolated. Unlike the continual loss of K due to SLR and HTF included in the model (Figure 37), loss of K due to storms and fragmentation are stochastic in the model.

Impacts of recent storms

The 2022 tropical storm season had a large and deleterious effect on the shoreline at Canaveral National Seashore due to Tropical Storm Ian after crossing the Florida Peninsula (September 29), and Hurricane Nicole making landfall south of the park at Vero Beach (November 10). The average net shoreline movement (NSM) between April and November 2022 was 15m (SD ±5.7m) along the 38.6km shoreline, according to the National Park Service (NPS) Vital Signs Monitoring Program (CNS 2023). This represents 57.9ha, which is 60% of the CNS SEBM habitat. A similar event occurred on October 10, 2024 as Hurricane Milton passed over the Florida Peninsula, per PVA team.

Since the *VORTEX* SQ models begin in January 2020, it is important to include these events in future projections. The model incorporates a **loss of 60% of the initial K in Timestep 38 (i.e., November 2022) and again in Timestep 63 (October 2024)**. After each event, habitat (K) recovers at the same rate as that designated for hurricanes and continues until K matches imposed by SLR or subsequent storms. No other storm events were allowed to occur in the model from January 2020 to December 2024 (time of modeling), as they were known not to occur. Annual risk of hurricanes (4%) and Nor'Easters (4%) was implemented starting in January 2025.

Beach erosion

Canaveral National Seashore is a narrow land strip subject to erosion. Rough calculations by members of the PVA team found beach erosion in CNS shows a three-fold increase from the historical period of 1943-2021 (0.6ha/year) to more recent period of 2012-2021 (1.8ha/year), just prior to the 2022 storm events above. Erosion was not considered to be additive with other beach habitat loss factors (i.e., storms, SLR, HTF) as these factors contributed to erosion. The estimated habitat loss from the 2022 and 2024 storm events and projected SLR and HTF result in a greater rate of habitat loss than that estimated for beach erosion. Thus, no additional loss of K for SEBM was added to the SQ CNS model.

Other threats/management

Invasive species impacts and habitat management (e.g., FEMA berm in public areas) are assumed to remain the same as in 2020 at the onset of the projection.

Status Quo Model Results

Like the base demographic model, the Status Quo model for the CNS subpopulation displays seasonal fluctuations in population size. In addition, demographic variation, environmental variation and other stochastic processes contribute to fluctuations in demographic rates and population size that are evident in this small population. Figure 38 illustrates the seasonal dynamics of mean population size (density = 5 mice/ha), as well as the standard deviation around that mean, for the first five years of the model (January 2020-December 2024). The red line represents the population if recent storm events are excluded. This seasonal pattern is a reasonable representation of beach mouse population biology. The blue line in Figure 36 depicts the same subpopulation when the 2022 and 2024 fall storm events are included in the model. The lack of births for two timesteps, followed by the three timesteps for mice to reach maturity, means that the new cohorts of breeders begin breeding in spring and summer when reproductive rates are lower. This contributes to the significant time lag before the population begins to grow in the subsequent early fall. The results for higher density (D7) show similar patterns.

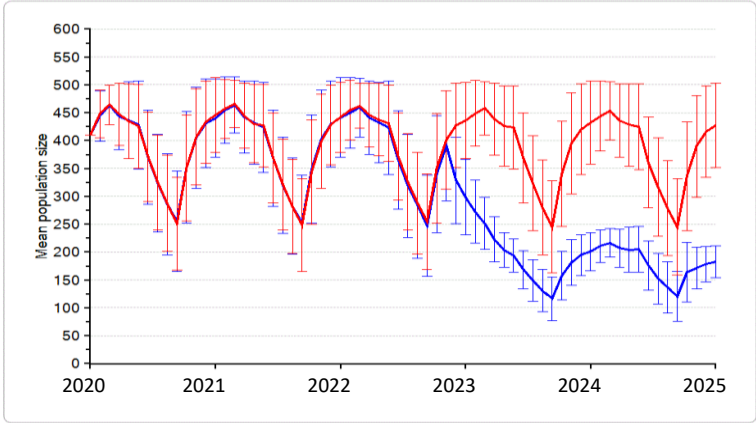


Fig. 38. Mean population size for the first five years of the Status Quo (D5) model, with (blue line) and without (red line) recent storm events (2022-2024). Bars represent ± 1 SD.

The Status Quo model combines the impacts of demographic and genetic stochastic processes on the small CNS subpopulation with loss of habitat (carrying capacity) and connectivity due to progressive sea level rise, recent erosion events, and major storms. These factors have been parameterized as accurately as possible based on best available data and expert opinion; however, many of these parameters are not well understood. The model results are influenced by the model inputs and assumptions. **Thus, rather than focusing on precise estimates of future population projections, this model should be viewed as a tool to investigate the primary driving factors influencing viability. This can help us to understand the degree of uncertainty in model projections, identify the most important data gaps (and potential research needs), and help to prioritize management actions that are most likely to be effective.**

Given the model assumptions and inputs, the resulting projection over 80 years for the CNS SEBM subpopulation is overall decline and eventual extinction. Figure 39 shows the probability of population persistence over time. There is a small risk of extinction (~2%) by 2040, after which extinction risk increases. Median time to extinction is 45-52 years (594-673 timesteps), and extinction risk increases dramatically after connectivity is lost with Beach2 subpopulation in 2080. Probability of extinction (PE) for the D5 model is 99.2% by 2100. Projected viability is slightly higher (PE=97.4%) if we assume the higher density of 7 mice/ha, but both densities tested follow a similar pattern, with high extinction risk.

After an initial recovery period following recent storms, mean population size decreases over time. Figure 40 demonstrates seasonal variation in mean population size, while Figure 41 show the mean population size for December (each 13th timestep) for all iterations, and the standard deviation around these means. One yearly census is plotted for illustrative purposes. A similar trend (decline and variations around the mean) is seen for other timesteps, with lower mean population size in summer. Both seasonal variation and stochastic variation among iterations contribute to the significant variation in population size at any one timestep or calendar year, but the long-term trend is overall decline. For the few iterations that did not go extinct (0.8%for D5; 2.6% for D7), the mean population size in 2100 was 58 and 115, respectively.

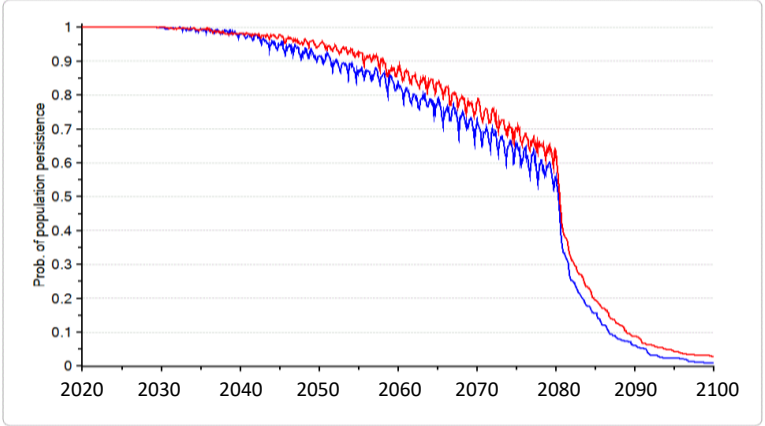


Fig. 39. Probability of persistence (i.e., at least one mouse of each sex) for the CNS subpopulation over 80 years. Blue=D5; red=D7.

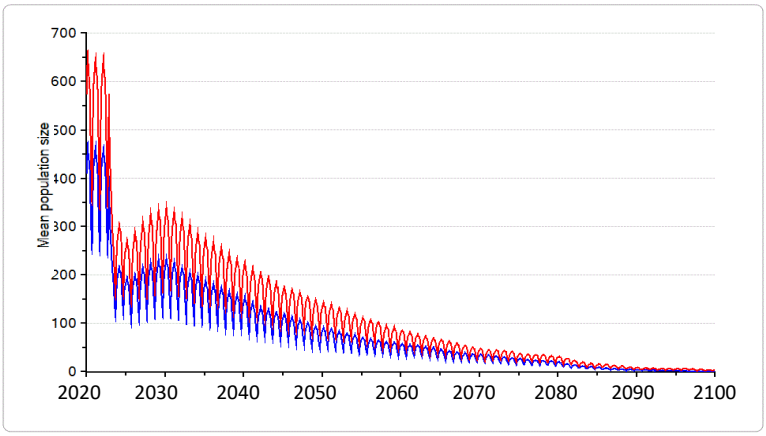


Fig. 40. Mean population size for the CNS subpopulation over 80 years. Each timestep (ts) is plotted, demonstrating seasonal variation in mean population size. Blue=D5; red=D7.

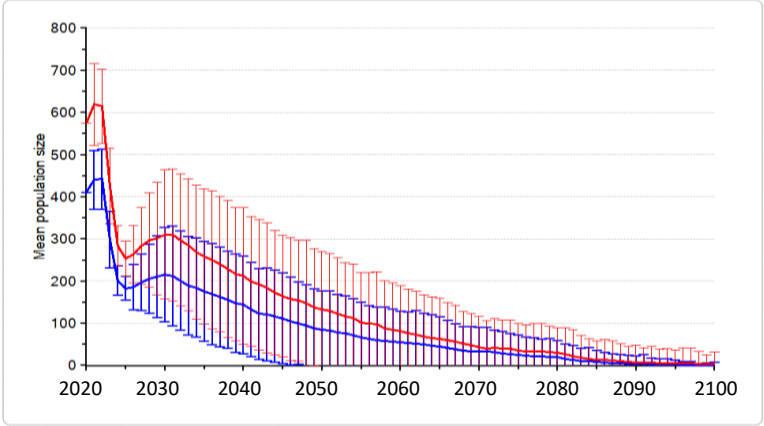


Fig. 41. Mean population size in December for the CNS subpopulation over 80 years. Each 13th ts is plotted, demonstrating variation in mean population size in December. Bars represent ± 1 SD. Blue=D5; red=D7.

The significant early decline in population size is attributable to the assumptions regarding the impact of 2022 and 2024 storm events on SEBM carrying capacity. While mean population size is initially greater using the higher density of 7mice/ha (red lines), this difference lessens as K and N become smaller. Note that while mean population size is small (<80) by 2060, probability of persistence is still 55-63% for the next 20 years (>30 generations) until connectivity with Beach2 is lost (in the model) in 2080 (Figure 39), indicating that this connectivity is supporting the persistence of a small SEBM population in CNS. Unlike the isolated SDP subpopulation, gene diversity in the CNS subpopulation remains high (>87%) while connected to the Cape, but immediately drops once connectivity is lost in 2080 (Figure 42). Similarly, inbreeding remains relatively low (<4%), despite generations of small population size, until connectivity is lost in 2080 (Figure 43). (Note that the odd patterns for these means in later years are artifacts of only a few persisting simulations, containing only a small number of mice.) In order to improve visual clarity, graphs plot every 13th timestep (i.e., December means).

These results indicate that the CNS subpopulation is being genetically rescued by mice continually crossing the park boundary from the Cape. The oscillating pattern in population persistence (Figure 39) indicates seasonal demographic rescue as well, with extinction risk being highest in summer when the population is smaller. On average, the CNS subpopulation is recolonized after extirpation 7 times (at 5 mice/ha) from Beach2 between 2025 and 2080 in the model; recolonization rate is slightly less (5 times between 2025-2080) at the higher density of 7 mice/ha.

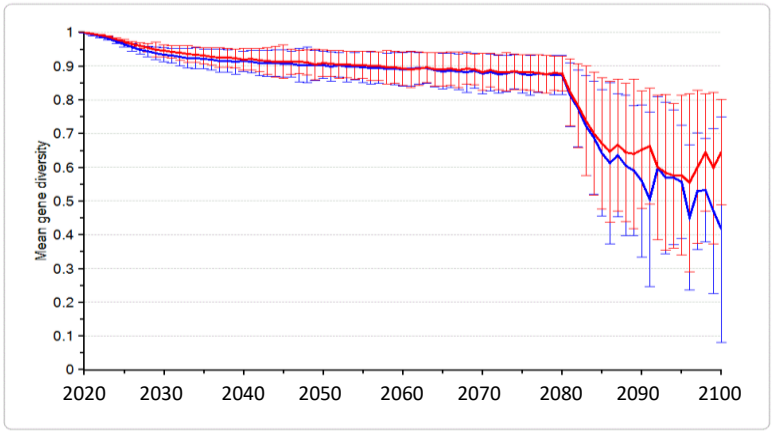


Fig. 42. Mean gene diversity for the CNS subpopulation over 80 years (each 13th ts is plotted, representing means for December). Bars represent ± 1 SD. Blue=D5; red=D7.

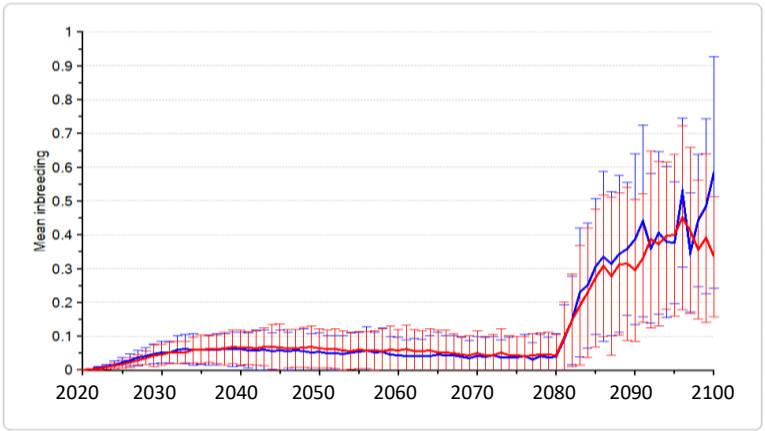


Fig. 43. Mean inbreeding coefficient for the CNS subpopulation over 80 years (each 13th ts is plotted, representing means for December). Bars represent ± 1 SD. Blue=D5; red=D7. F=0.25 is equivalent to full-sibling mating, while F=0.5 is equivalent to self-fertilization.

Taken together, the **Status Quo results demonstrate a declining population, with a persistent small SEBM population that is dependent upon connectivity with other SEBM subpopulations for demographic and genetic rescue**. Loss of this connectivity in the future may jeopardize the persistence of SEBM in Canaveral National Seashore in the absence of population restoration actions.

Driving Factors in SQ Projections

Several concurrent factors impact the CNS SEBM population, both in the model and in the wild. Exploratory analyses of factors impacting the slightly smaller, isolated SDP subpopulation to the north found that the loss of K due to major storms, combined with inbreeding depression, were the driving factors for population viability. The situation is more complex for the CNS SEBM subpopulation due to its connectivity with the large Beach2 subpopulation; as long as connectivity is maintained, the influx of immigrants keeps inbreeding relatively low and often recolonizes extirpated subpopulations in the model. Several exploratory scenarios were developed to isolate the effects of some of these factors in order to better understand how they are impacting the SQ projections for the CNS subpopulation.

Impact of recent storms

While significant beach erosion events are known to have occurred in 2022 and 2024 in CNS, the actual impact on SEBM and on SEBM habitat is uncertain. The SQ model incorporates 60% loss of K in both 2022 and 2024, followed by slow recovery, with no additional demographic impacts on beach mice. Scenarios without the 2022 and 2024 events were compared with the SQ scenarios. These recent events have no appreciable impact on long-term persistence of mice in CNS (Figure 44), likely due to

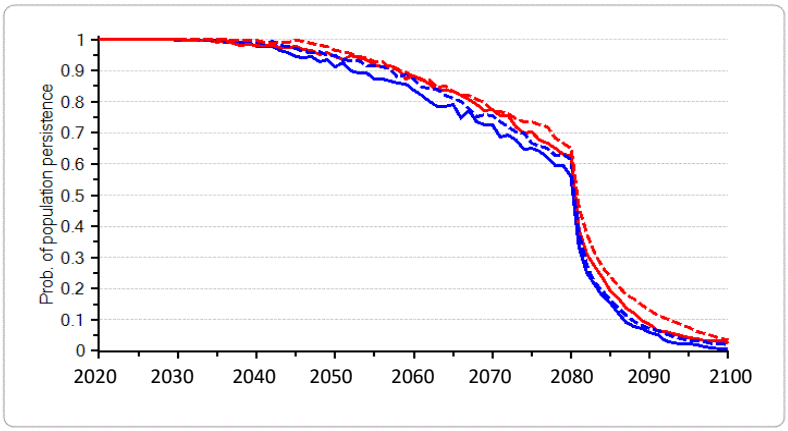


Fig. 44. Probability of persistence (in December) for the CNS subpopulation over 80 years. Solid lines=SQ scenarios; dashed=with 2022&2024 events removed. Blue=D5; red=D7.

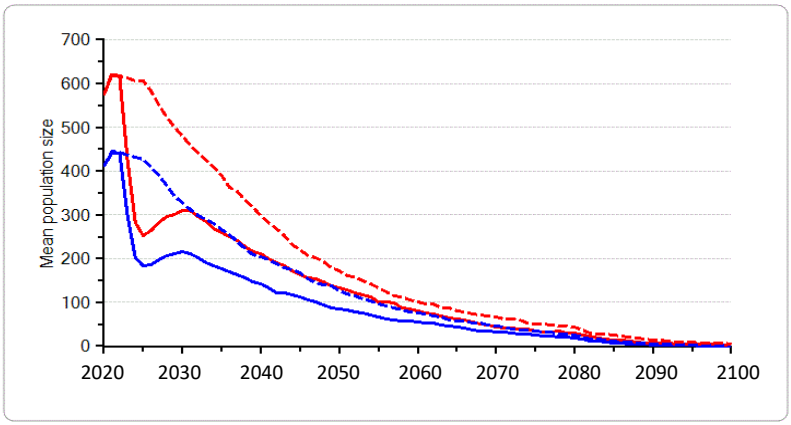


Fig. 45. Mean population size in December for the CNS subpopulation over 80 years. Solid lines=SQ scenarios; dashed=with 2022&2024 events removed. Blue=D5; red=D7.

connectivity to and continual recolonization from mice on the Cape. Mean population size is substantially larger in the short term without these loss of K events, especially if we assume a higher mouse density (Figure 45). However, all scenarios indicate significant decline (>50%) by 2040, and mean population size is small (~<100) for all scenarios by 2060. Thus, assumptions regarding the impacts of recent erosion events are not a significant driver of long-term viability projections in the SQ model.

Impact of future storms

Future storms have multiple impacts in the CNS SQ model. As with the SDP SQ model, exploratory scenarios found that the demographic impacts of reduced survival and reproduction have almost no effect on long-term viability as modeled, nor does the seasonality of storms. Storms have a significant impact on K, with reduction to 40% of pre-storm values and recovery only to 50% of pre-storm values (in contrast to 70% recovery for the SDP SQ model). This means that successive storms increasingly lower future K in the model especially for CNS. Additionally, there is a 50% risk of fragmentation when a major storm occurs, simulating the central region of CNS washing through and permanently separating the subpopulation into two fragments. The likelihood of fragmentation increases over time as more storms occur. Together, the impacts of loss of K and fragmentation significantly reduce mean population size (Figure 46), and increase extinction risk by 2100 from 5% (7 mice/ha) and 13% (5 mice/ha) to 97-99%. Exploratory scenarios were developed to isolate the relative impacts of loss of K vs fragmentation.

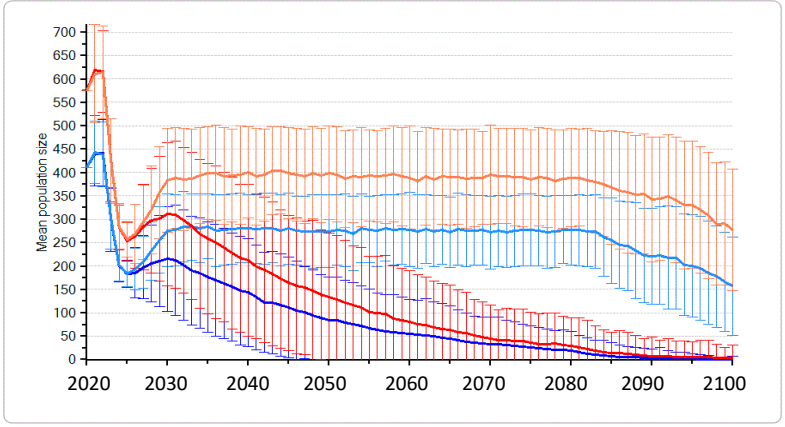


Fig. 46. Mean population size in December for the CNS subpopulation over 80 years for status quo and with storms removed. Bars represent ± 1 SD. Blue/light blue=D5; red/orange=D7.

Loss of K

Loss of K due to storms was found be a major driver of long-term viability for the SDP subpopulation, and a similar trend can be observed in the CNS model results. When the loss of K due to storms is removed (but risk of fragmentation remains), extinction risk to 2080 drops to zero (Figure 47; dotted lines), and mean population is significantly higher (Figure 48). As seen in the SQ model, population size declines and extinction risk increases once connectivity to the Cape is lost in 2080 and the CNS subpopulation is no longer demographically and genetically augmented by Beach2 immigrants.

Fragmentation

The probability of population persistence to 2080 is much higher (85-90%) when there is no fragmentation (but still loss of K), but shows the similar sharp decline as observed in the SQ model when the CNS subpopulation becomes isolated (Figure 47; dashed lines). Mean population size is similar with

and without fragmentation (Figure 48), and is driven by the loss of K over time by both storm types. Median time to extinction increases by 9 years (D7) to 14 years (D5) with no fragmentation risk.

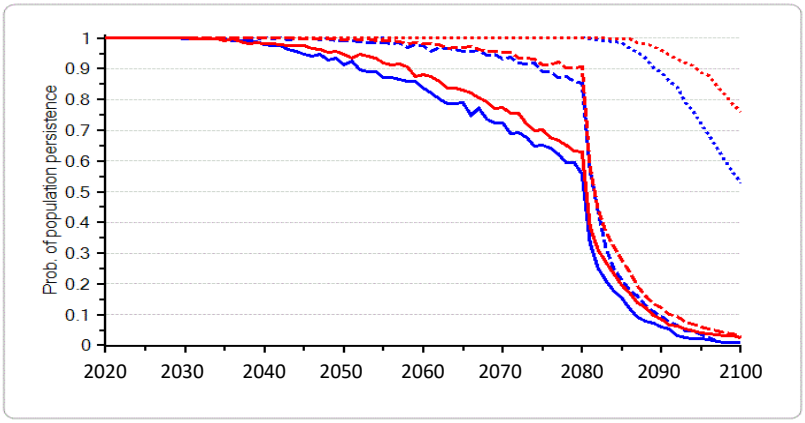


Fig. 47. Probability of persistence (in December) for the CNS subpopulation over 80 years for status quo (solid), storm K loss removed (dashed), and fragmentation removed (dotted). Blue=D5; red=D7.

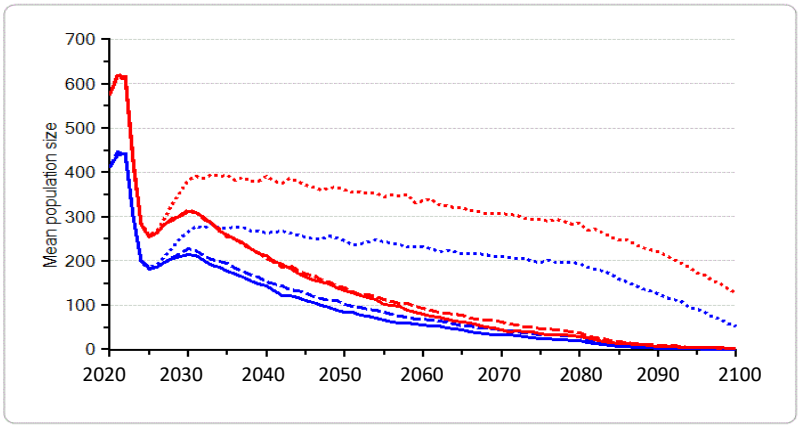


Fig. 48. Mean population size in December for the CNS subpopulation over 80 years for status quo (solid), storm K loss removed (dashed), and fragmentation removed (dotted). Blue=D5; red=D7.

To better understand the impact of fragmentation, it is important to realize that Figure 48 represents 500 iterations, with only a small risk of fragmentation in each hurricane season. Fragmentation will occur, on average, in 1-3 of the 500 iterations (i.e., <1%) for each timestep during hurricane season, evening out the impact across the 80-year projection. Figure 49 shows results if fragmentation is forced to occur in 2051 (approximate median time to fragmentation). The isolated northern fragment (red line) declines and goes extinct, while the southern fragment near the Cape (blue) is maintained through

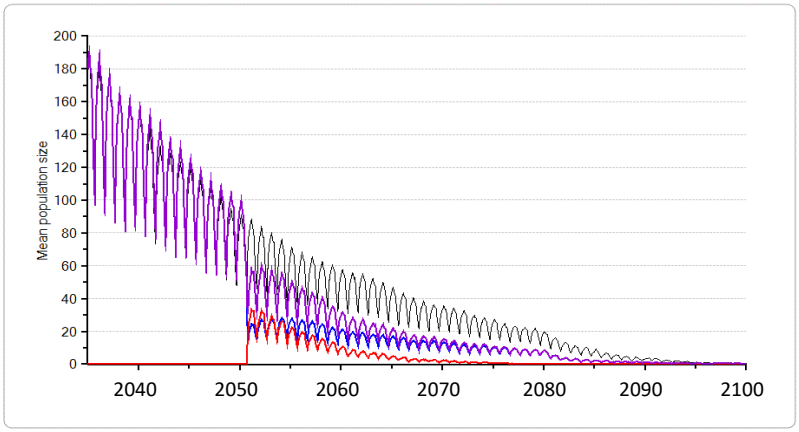


Fig. 49. Mean population size from 2030 to 2080, with forced fragmentation in 2051 (ts 400), along with the SQ5 projection (gray line). Assumes 5 mice/ha (D5).

connectivity to Beach2. The purple line represents the total population on CNS (both fragments). Thus, fragmentation affects SEBM distribution, as the isolated northern fragment declines toward extinction while a small southern subpopulation usually persists as long as it remains connected to Beach2.

Summary

Taken together, the impacts of major storms on SEBM habitat are projected to significantly affect CNS SEBM subpopulations. **The loss of K due to storms impacts population size and therefore extinction risk, while fragmentation affects SEBM distribution within CNS and shortens median time to extinction**.

Impact of sea level rise

The rise in sea level, including high tide flooding, is projected to reduce SEBM habitat in CNS, accelerating after 2080. Similar to the results for the SDP subpopulation, removing the loss of habitat due to SLR and HTF has very little effect on the model results on either the probability of persistence or mean population size. This is not surprising, as other factors have a greater impact on K. Figure 50 illustrates the projected loss of K due to SLR & HTF (blue line), recent 2022 and 2024 erosion events (green line), and future storms (red line, mean ±SD). Across the 80-year model timeline, recent and future storms drive loss of K in the model, although in reality all of these factors may be inter-related.

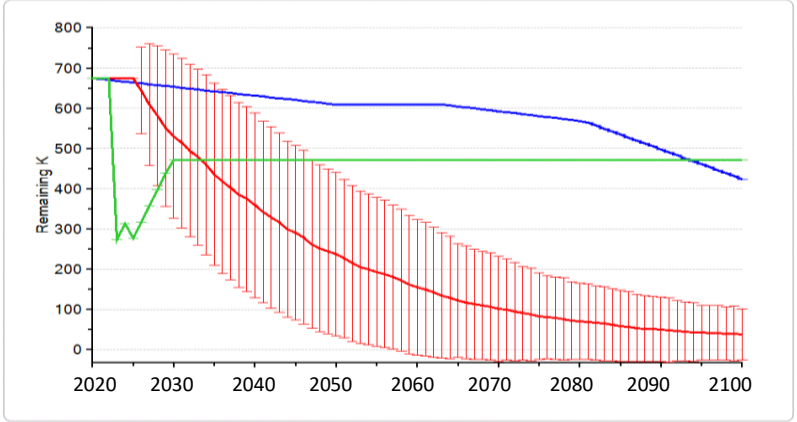


Fig. 50. Mean carrying capacity (K) for CNS SEBM subpopulation over 80 years as imposed by SLR&HTF (blue), recent erosion events (green), and storms (red; bars represent ± 1 SD). Future storm events are stochastic in the model while SL&HTF was modeled as a deterministic impact.

Loss of connectivity

In the model, SLR also eventually leads to the loss of connectivity between the CNS and Beach2 SEBM subpopulation. This is modeled as occurring in 2080, approximately when SLR is projected to accelerate, but could occur, or not, at a different time. Here we use the SQ and exploratory scenarios to further investigate the importance of connectivity.

Figure 51 compares the probability of persistence for the CNS subpopulation if connectivity is maintained (Connected; green line), lost in 2080 (status quo; blue), or lost at the simulation onset (Isolated; red). Results are shown for D5, but D7 scenario results follow the same pattern. **Extinction risk increases rapidly when the subpopulation is isolated and cannot be demographically or genetically rescued**. The earlier isolation occurs, the more rapidly population size declines and extinction risk accelerates.

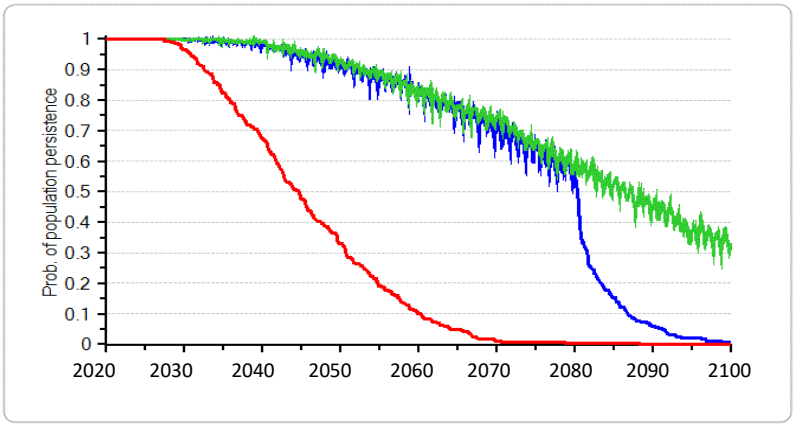


Fig. 51. Probability of persistence for the CNS subpopulation over 80 years for status quo (blue), with constant connectivity to B2 (green), and if completely isolated from B2 (red). Assumes 5 mice/ha (D5).

Genetic rescue contributes to CNS subpopulation persistence. Gene diversity remains high (>83% in 2100) if connectivity is maintained, but drops quickly when the subpopulation is isolated (Figure 52). Inbreeding follows the reverse pattern, remaining low (<6%) with habitat connectivity but rising quickly with isolation (Figure 53). The positive genetic impacts of connectivity may be slightly overestimated, as the model assumes that all new migrants from Beach2 are unrelated, but the general trend should be the same.

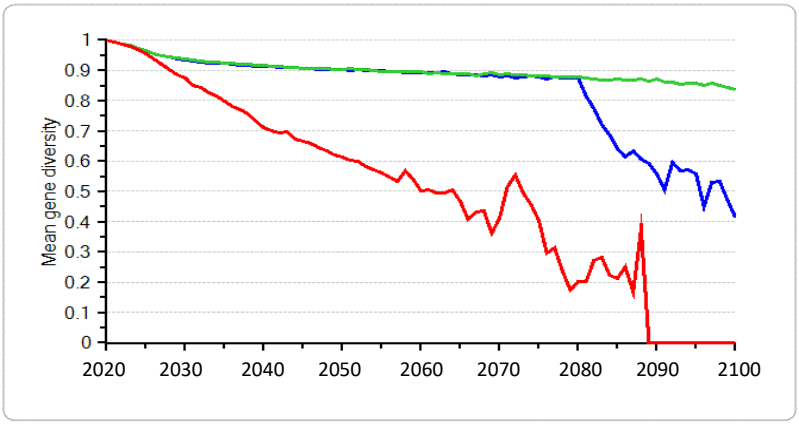


Fig. 52. Mean gene diversity for the CNS subpopulation over 80 years for status quo (blue), with constant connectivity to B2 (green), and if completely isolated from B2 (red). Assumes 5 mice/ha.

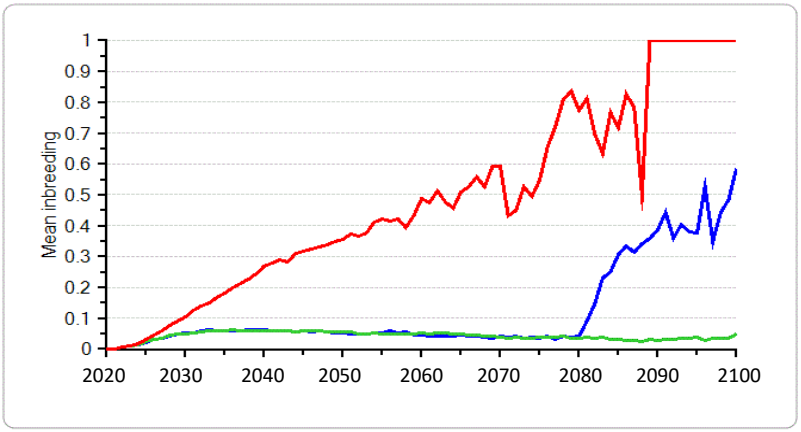


Fig. 53. Mean inbreeding for the CNS subpopulation over 80 years for status quo (blue), with constant connectivity to B2 (green), and if completely isolated from B2 (red). Assumes 5 mice/ha.

Inbreeding depression, however, is not the sole factor responsible for the higher extinction risk observed with habitat isolation. Figure 54 depicts population persistence projections from years 2070 to 2100 and demonstrates that if the effects of inbreeding depression are removed (red line) from the status quo model, extinction risk is lower after isolation but is still high (92% by 2100). On the other hand, if connectivity is maintained (green), extinction risk by 2100 is 68%, suggesting that demographic augmentation is also a primary factor. The cyclic nature of the projections indicates seasonality in extinction risk and demographic rescue, with extinction more likely to occur in late summer when SEBM numbers are low and there is risk of hurricanes and associated effects. Mean population size is small by 2070, due to habitat loss, fragmentation and storms, such that **demographic augmentation and recolonization are important mechanisms for SEBM to persist on CNS**.

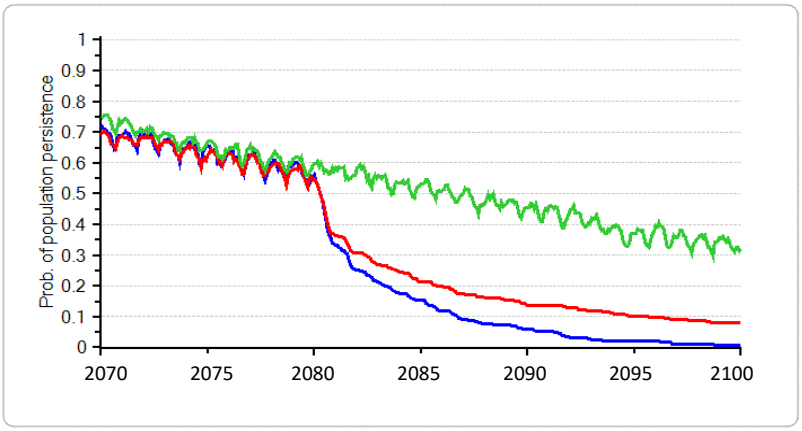


Fig. 54. Probability of persistence for the CNS subpopulation from 2070 to 2100 for status quo (blue), SQ with no inbreeding depression (red), and with constant connectivity to B2 (green). Assumes 5 mice/ha.

Implications: uncertainty, knowledge gaps, and management options

From the above exploratory analysis, we can infer several conclusions. As seen with the SDP model, we need not be too concerned that either environmental variation of demographic rates, or the impact of storms on these rates, are overestimated in the model (although they may be underestimated). These parameters are not high priority data gaps in terms of assessing SEBM population viability, nor is management to mitigate these effects a priority. Higher mouse density (i.e., higher K and initial N) results in slightly higher mean population size and probability of persistence, but the current uncertainty regarding density also is not a primary area of concern affecting long-term viability projections.

Loss of habitat (and the resulting reduction in K for beach mice) is a primary concern, as is habitat fragmentation. Recent storm events may have significantly affected SEBM habitat on CNS.
The current parameterization of habitat loss due to storms overrides the projections for SLR and HTF given the model assumptions, although either threat drives the subpopulation smaller and increases its vulnerability to inbreeding depression and other stochastic impacts. Habitat fragmentation can isolate segments of the SEBM subpopulation on CNS, and, if permanent, is likely to lead to SEBM extirpation in these areas. The model inputs for storm impacts were less data driven than for SLR-HTF projections and may be an important source of uncertainty in SQ model projections. Better understanding of recent and projected impacts of storms on SEBM habitat in CNS would improve viability projections for this subpopulation.

Connectivity to the Beach2 subpopulation is critical to the persistence of SEBM in CNS. This is increasingly vital as SEBM habitat on CNS is reduced and the mouse population declines. The movement of SEBM from Beach2 onto CNS provides both demographic and genetic rescue of small or extirpated populations. This suggests that the translocation of beach mice to CNS may be a potential strategy to maintain SEBM on CNS, especially if any areas of CNS become fragmented or isolated from the Cape.

Any management efforts to restore, maintain or improve habitat quality and connectivity for SEBM will promote the viability of this small subpopulation by allowing the population to remain as large as possible, which in turn will reduce slow genetic drift and the accumulation of inbreeding and reduce risk of extinction.

Alternative Management Scenarios

Genetic Augmentation Scenarios

Several alternative management scenarios were developed to explore the potential impacts of demographic and genetic augmentation on the SEBM CNS subpopulation. All scenarios included the periodic release of subadult (4 week old) mice, of approximately equal sex ratio, with releases occurring in October, per PVA team discussions. Translocated mice were modeled as unrelated to the CNS subpopulation and are assumed to be obtained from the large Beach2 wild subpopulation. Releases of 30-50 mice were suggested as numbers that are both feasible and potentially effective.

Little information exists on the survival and reproduction of beach mice that are translocated between wild populations. To address this uncertainty, the following strategy was followed (same as for the SDP model). No translocation mortality is included in the model, and the model assumes that translocated mice have the same mortality and reproductive rates as resident mice. Essentially this means that the number of releases in the model reflect the number of *successful* releases. The number of actual mice needed to be released to achieve this level would need to be calculated based on the success (survival and reproduction) observed for actual translocations.

Translocation model scenarios for the SDP SEBM subpopulation found that short-term releases are not effective to improve long-term viability, and that any augmentation strategy should consist of periodic, continual releases. This is further supported by model results that show rapid decline in viability when a part of the CNS subpopulation is isolated, whether via fragmentation or loss of connectivity to the Cape.

Scenarios were developed to explore periodic, continual translocation for CNS, with the release of 30 mice every 5 or every 10 years. Results indicate that periodic translocations on this scale have little to no impact on the probability of persistence of mice on CNS, nor on mean population size, gene diversity or inbreeding, as long as there is connectivity with the Cape. This is not surprising, as the model assumes constant immigration from Beach2 and at a rate higher than of translocation efforts, as modeled. Immigration rates are seasonal and decline over time as the boundary narrows, but range from a high of ~21 mice/year at the onset of the model to ~8.5/year by 2099. These results suggest that the immigration rates used in the model are sufficient and that translocations are not needed into habitat that is connected to Beach2, provided transboundary movement is not overestimated in the model. That said, the model assumes a panmictic breeding subpopulation, while in reality the genetics from Beach 2 migrants are less likely to reach the northern areas of CNS.

Periodic translocations may provide some benefit to isolated population fragments. Scenarios were developed in which the subpopulation was forced to fragment in 2051 (rather than spreading this impact across the model timeline), and each resulting fragment was supplemented via translocation of 50 mice initially and 30 mice every 5 or 10 years thereafter. Under these model conditions, mice are more likely to persist in the northern isolated fragment but as a small population (typically fewer than 100 mice), with a high likelihood of extirpation between translocations (Figure 55). Almost all iterations (98%) went extinct at least once, and were re-established via translocation 3-6 times between 2051 and 2099. The southern fragment shows little benefit from translocation while connected to the Beach2 subpopulation, but follows a similar pattern to the northern fragment translocation results once isolated in 2080, i.e., a small population at risk of extirpation between translocation events (Figure 56).

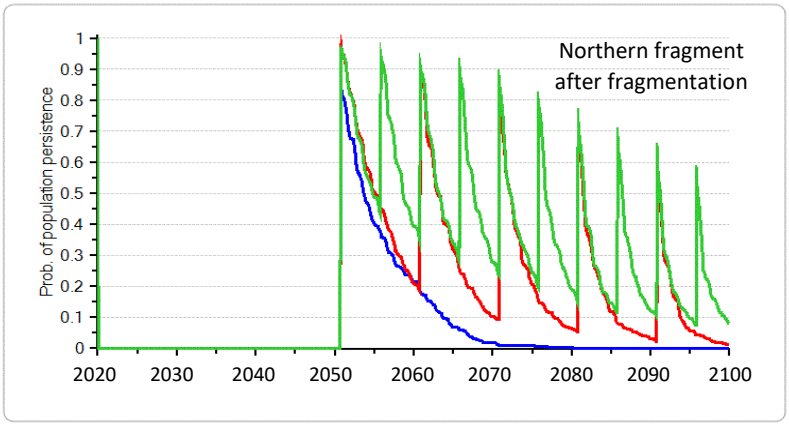


Fig. 55. Probability of persistence for the isolated northern population fragment (after fragmentation in 2051), with no translocation (blue), and with translocations every 10 years (red) and every 5 years (green). Assumes 5 mice/ha (D5).

5_No Trans
5_Tr_10Y
5_Tr_5Y

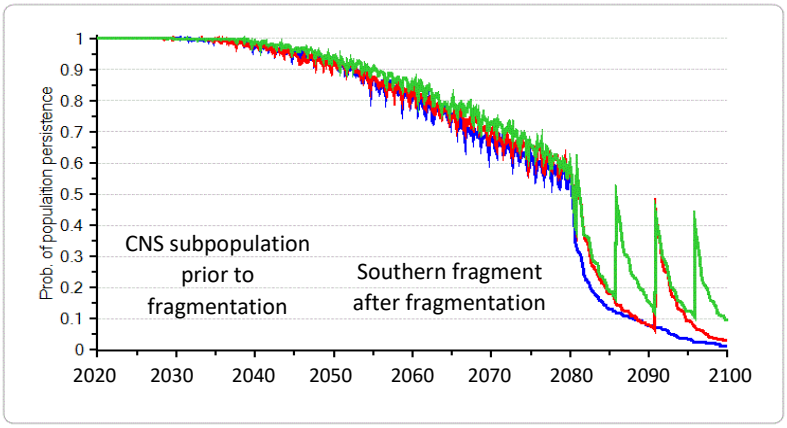


Fig. 56. Probability of persistence for the entire CNS subpopulation (prior to fragmentation) and the southern population fragment (after 2050), with no translocation (blue), and with translocations every 10 years (red) and every 5 years (green). Dashed line=time of fragmentation. Assumes 5 mice/ha (D5).

5_No Trans
5_Tr_10Y
5_Tr_5Y

While translocation can be used to re-establish extirpated isolated beach mice populations, or augment small existing population fragments through demographic and genetic augmentation, poor long-term viability on CNS is still driven by the loss of habitat and carrying capacity for beach mice especially due to some permanent habitat loss during each major storm event. **Thus, translocations alone are unlikely to be sufficient for long-term viability on CNS without habitat restoration efforts**.

Habitat Recovery Scenarios

Reduction in K due to habitat loss caused by storms is an important driver of viability in the SQ model. It may be possible to improve the recovery of habitat quality and quantity for beach mice through habitat restoration. Three hypothetical habitat restoration scenarios were tested: 1) HR1: habitat recovery to

80% of pre-storm value in 10 years, which represents twice the recovery rate as the SQ model and continues until a much larger portion of SEBM habitat is recovered; 2) HR2: habitat recovery as described in #1, as well as prevention of habitat fragmentation; and 3) HR3: habitat recovery and restoration as in #2, as well as maintenance of connectivity with Beach2. In these scenarios, habitat recovery is not allowed to exceed the available habitat based on SLR & HTF projections. Revised recovery rates were applied to future storms (2025 or later). These hypothetical scenarios can serve as benchmarks for management effort vs benefit for SEBM.

These habitat restoration models lead, on average, to higher SEBM carrying capacity but still result in loss of habitat significantly greater than that due only to SLR and HTF. All three habitat recovery strategies lead to lower extinction risk (Figure 57), larger population size (Figure 58), high gene diversity (88-90%), and low inbreeding (Figure 59) as long as connectivity to the Cape is maintained. Inbreeding and extinction risk rise rapidly once connectivity is lost (i.e., 2080 for the SQ, HR1 and HR2 scenarios). In contrast, population persistence is high (i.e., low risk of extinction) when connectivity is maintained and mice are able to move between these subpopulations, and is greatly improved compared to the scenario that maintains connectivity without additional habitat recovery efforts (see Figure 51).

Summary

A Status Quo model for the SEBM CNS subpopulation was developed based on the best available data and expert opinion. Given model inputs and assumptions, the projected long-term viability of SEBM in CNS is poor, with population decline and eventual extinction. The key driving factors affecting this projection are habitat loss (especially due to storms that lower the carrying capacity for mice beyond the habitat loss caused by sea level rise and high tide flooding) and loss of connectivity to the SEBM Beach2 subpopulation, which provides both demographic and genetic augmentation of this small subpopulation. The greatest uncertainty surrounding these projections is around the impact of storms on SEBM habitat and the recovery of that habitat, including higher fragmentation than modeled here, followed by uncertainty in the rate of movement of mice between these adjacent subpopulations.

Modeling of alternative management scenarios suggest that effective management may need to address two issues: carrying capacity and population augmentation. Foremost, it is important to restore, maintain and/or increase SEBM habitat and its carrying capacity to support beach mice. While all potential habitat management actions were not explicitly modeled, any efforts to increase K for beach mice are likely to be beneficial, including post-storm habitat restoration as well as other potential habitat management strategies, including efforts to address recent habitat loss. Restoration should include efforts to prevent fragmentation along the CNS, with special attention to prevent the loss of connectivity with the Cape.

Demographic and genetic augmentation is essential to the long-term persistence of SEBM in CNS. Under the model assumptions, sufficient augmentation is provided through current connectivity with the Cape. The model assumes, however, that the CNS subpopulation is panmictic, whereas in reality this is a thin, linear habitat with connectivity only along a small southern border. If natural immigration is insufficient (especially in the northern area of CNS), or if fragmentation or isolation occurs, then translocation may be a valuable tool to boost the SEBM subpopulation. Exploratory translocation efforts may help to

identify the most effective release strategies and fill in data gaps on post-release survival and reproduction before the need develops for this management tool.

While uncertainty limits the precision of these modeling results, they provide insight into future SEBM CNS trajectories under current and alternative future management strategies. This SEBM subpopulation would benefit from monitoring to help identify the need for translocation and to better understand how beach mice are impacted by the stochastic forces impacting this dynamic habitat.

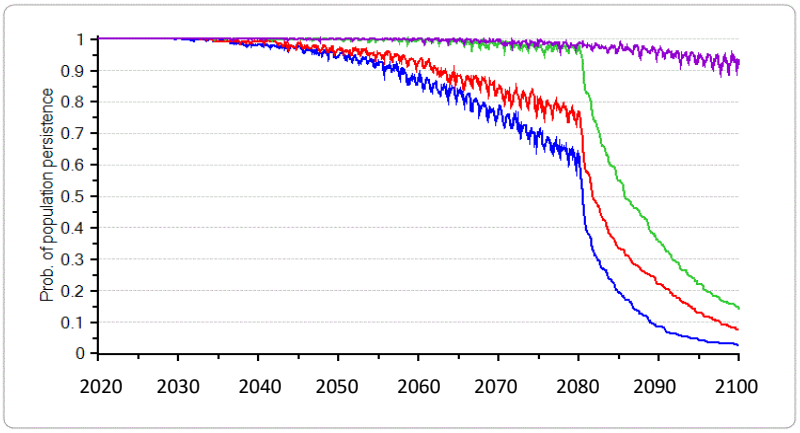


Fig.57. Probability of population persistence for the CNS subpopulation over 80 years with different habitat recovery scenarios. Assumes 7 mice/ha (D7).

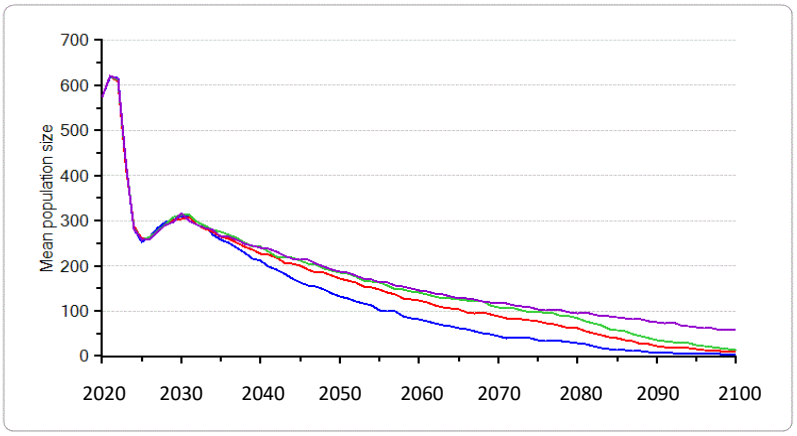


Fig. 58. Mean population size in December for the CNS subpopulation over 80 years for status quo (blue), and three alternative habitat recovery scenarios. Assumes 7 mice/ha (D7).

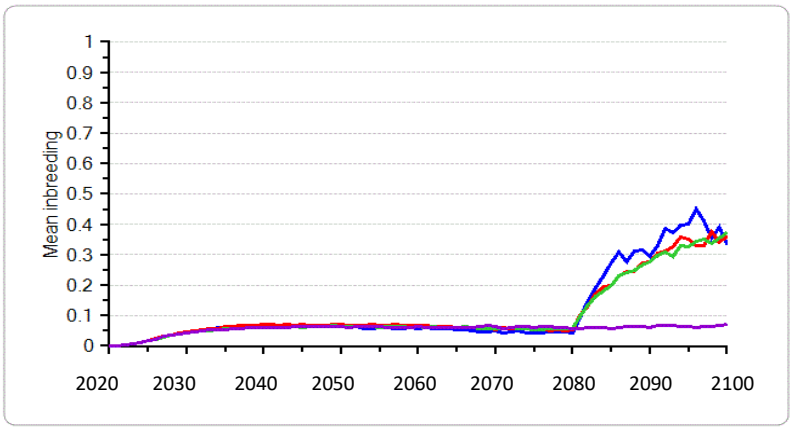
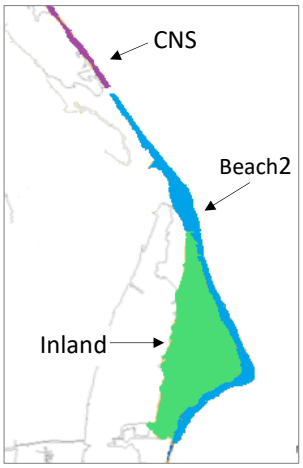


Fig. 59. Mean inbreeding for the CNS subpopulation over 80 years for status quo (blue), and three alternative habitat recovery scenarios. Assumes 7 mice/ha (D7).

CAPE CANAVERAL SOUTHEASTERN BEACH MOUSE SUBPOPULATION

Fig. 60. General map of Beach2 and Inland subpopulations.



The Southeastern beach mouse (SEBM) ranges across Cape Canaveral, both in coastal dune beach areas and inland habitat. Coastal habitat on the northern section of the Cape at Merritt Island National Wildlife Refuge (MINWR) is contiguous with Canaveral National Seashore (CNS), providing a narrow corridor that connects the Cape and CNS SEBM subpopulations. To the south, SEBM subpopulations extend to Point Canaveral Entrance Channel. For the purposes of this PVA, the SEBM population on the Cape is separated into two subpopulations: mice living in the coast dune and strand habitat (Beach2), and mice living in the inland ruderal and oak scrub habitat (Inland). Figure 60 is a general illustration of these subpopulations but is not exact in terms of width and area. Beach2 represents 1238.3 ha of dune habitat (USFWS 2020) that wraps around the Cape and includes MINWR and the Kennedy Space Center (KSC). This habitat is not entirely homogenous and is eroding in the north but accretes in the south. The large area (3786.4ha) of inland scrub habitat on the Cape Canaveral Space Force Station (CCSFS) also supports beach mice and is contiguous with the Beach2 habitat along a long habitat transition zone border. Cape Canaveral is larger and less vulnerable to habitat loss due to storms than SEBM habitat in Smyrna Dunes Park and Canaveral National Seashore but is subject to other habitat-related issues such as development and management.

A Status Quo (SQ) *VORTEX* metapopulation model was developed for the two Cape SEBM subpopulations to represent the best available estimates of current and future conditions, assuming there are no changes in current management. The SQ model enabled projections of future viability under these conditions, sensitivity testing regarding the relative contribution of various threats and data uncertainty to future viability, and exploration of alternative management scenarios and their impact on viability. All model scenarios began in January 2020 and were run for 80 years (i.e., to 2100). Due to the large population size on the Cape, which requires long computing times, each scenario was run for 250 iterations. Simulations for this large metapopulation were less stochastic than for other, smaller SEBM subpopulations, allowing good precision with fewer simulations.

Model Inputs

The SEBM demographic base model was used, which includes seasonal fluctuations in sex- and age-specific reproduction and survival (see earlier descriptions of this model). The PVA team recognized that these rates and seasonality are based on data for SEBM living in dune habitat. Comparable data for SEBM in inland habitat were not available at the time of this PVA; therefore, the model assumes that the Inland subpopulation has the same seasonal reproductive and survival rates as the other (beach) SEBM subpopulations.

The following information summarizes the subpopulation-specific inputs and revisions for the two Cape SEBM subpopulations (see Appendix B for summary table of model inputs and details on the derivation of habitat- and catastrophe-related inputs).

Initial carrying capacity (K)

Discussions by the PVA Team recognized some uncertainty regarding SEBM density across the Cape, especially for inland habitat. Dune habitat on the Cape is estimated to be of good quality and to support the highest densities of SEBM. Conversely, the inland habitat is variable and estimated to support the lowest SEBM densities. The team estimated an average of 9 mice/ha for Beach2 and 4 mice/ha in Inland, which results in an estimated initial K of 11145 for Beach2 based on 1238.3ha, and an initial K of 15146 for Inland based on 3786.4ha.

The PVA team recognized that the presence of numerous launch sites on the Cape is likely to affect the density of SEBM in the surrounding area due to lighting, noise and other disturbance due to human activities. GIS analysis determined the area represented by a surrounding impact zone of either 1/4-mile (14z, with 'z' designating the width of the impact zone in a scenario) or 1/2-mile (12z) around these sites. The resulting impact zones were then estimated to have either 50% of normal SEBM density (50m) or to have no mice (0m). This resulted in five scenarios:

1) No impact zones (i.e., no effect on SEBM)
2) 1/4-mile impact zones with 50% SEBM density (14z50m)
3) 1/4-mile impact zones with no SEBM (14z0m)
4) 1/2-mile impact zones with 50% SEBM density (12z50m)
5) 1/2-mile impact zones with no SEBM (12z0m)

Impact zones affect both Beach2 and Inland habitat. Revised carrying capacities were calculated for each subpopulation for each of these five scenarios (Table 4). All five scenarios were tested initially to determine the sensitivity of model results to impact zone size and mouse density within this zone.

Table 4. Area of two types of impact zones and resulting carrying capacity (K) and initial population size (N).

	Beach2			Inland		
Scenario	**Zone (ha)**	**Initial K**	**Initial N**	**Area (ha)**	**Initial K**	**Initial N**
No impact zones	0	11145	9473	0	15146	12874
1/4 mi @50% density	18.1	11063	9404	181.2	14783	12566
1/4 mi with no mice	18.1	10982	9334	181.2	14421	12258
1/2 mi @50% density	113.6	10634	9039	487.7	14170	12045
1/2 mi with no mice	113.6	10123	8604	487.7	13195	11216

Initial population size

The SQ model begins in early January, when the mouse population is projected to average about 85% of carrying capacity. This is based on stochastic runs of the base model with seasonal demographic rates, resulting in a mean N/K of 85%. The resulting initial population size of 9473 (Beach2) and 12874 (Inland) before impact zones are considered. Table 4 gives the reduced initial Ns for each impact zone scenarios. Individuals in the initial population were distributed across age and sex classes based on a stable distribution using the base model winter demographic rates. Full juvenile mortality rates (including mortality due to death of dam) were applied to generate this distribution across age classes starting at A=1 (4 weeks old).

Initial genetic status

Recent analyses by Austin (2021; unpublished data) found no significant genetic differentiation in the SEBM population across the Cape Canaveral Complex (CC), which includes the CNS, Beach2 and Inland subpopulations. Since the base demographic rates were taken from this slightly inbred subpopulation, these rates account for any historical inbreeding impacts. Thus, initial kinships and initial inbreeding (F) were set to 0 in all three of these subpopulations (i.e., current genetic state of each population is set as the baseline for modeling future declines in genetic diversity and impacts of any additional inbreeding). Additional future inbreeding in the model will be subject to inbreeding impacts, as described below.

Inbreeding impacts

Inbreeding effects are applied in *VORTEX* as lower juvenile survival, parameterized as lethal equivalents (LE). Brewer *et al*. (1990) calculated 5.52 LE in SEBM under laboratory conditions (derived from founders captured in 1984 from CNS). This estimate was reanalyzed by Lacy *et al*. (1996) to account for probability of breeding, resulting in a new estimate of 4.72 LE. Laboratory conditions are relatively benign compared to challenges faced by species in their natural environment, resulting in greater inbreeding effects in the wild (Frankham *et al*. 2010). O'Grady *et al*. (2006) estimate a combined mean effect of inbreeding on fecundity and first year survival for wild vertebrate populations to be 6.29 LE (default value in *VORTEX*). This value was used in sensitivity testing of the base model and was found to have an impact on all population sizes tested (K = 50 to 1000), with smaller populations impacted more quickly and leading to population decline and higher extinction risk.

The PVA team chose to use the more conservative laboratory estimate of 4.72 LE for modeling the wild SEBM subpopulations. While wild populations are generally thought to show greater inbreeding effects than those of the same species under laboratory conditions, the stochastic nature of beach mouse population dynamics may have purged some of the effects in the wild. While *VORTEX* allows LEs to be distributed across two mechanisms in the model (lower survival in inbred juveniles vs lethal alleles), it is not computationally practical to apply lower survival to inbred juveniles to this large population. Thus, inbreeding depression was modeled exclusively using recessive lethal alleles, which means that some impacts are subjected to purging over time. This is believed to be a conservative estimate of anticipated inbreeding effects. No additional inbreeding impacts were added to survival of other age classes or to reproductive rates.

Connectivity

Subpopulations designated as CNS and Beach2 in this PVA are connected by a narrow corridor near the boundary between CNS and MINWR. In addition, the two Cape subpopulations (Beach2 and Inland) share a long boundary along the habitat transition area on the Cape. Dispersal rates across both boundaries were calculated using the same methodology developed for the 2005 Alabama beach mouse (ABM) PVA (Traylor-Holzer *et al.* 2005). Mice are assumed to be evenly distributed within each subpopulation habitat and to disperse from their natal area randomly in all directions, which means that some of the mice born within the average dispersal range of a boundary will cross into the adjacent subpopulation. Acreage within the average dispersal range, on each side of the boundary, was calculated based on GIS analysis, and divided by the total acreage for each subpopulation to determine the percent of each subpopulation within its respective dispersal zone. These percentages were applied to juvenile (4 week old) mice to calculate trans-boundary dispersal rates in each direction.

The two Cape subpopulations were combined into a single metapopulation model for each scenario. The Dispersal functionality in *VORTEX* was used to model transboundary movement. Adjustments were made to both Beach2 and Inland to incorporate different SEBM density in the launch impact zones, which had some overlap with the transboundary dispersal zones. These calculations led to 9.1-10.4% of 4-week old mice in Beach2 moving to the Inland subpopulation (depending on the impact zone scenario), and 1.9-2.4% moving from Inland to Beach2.

Including the CNS subpopulation in this metapopulation model was impractical, as calculation time and power needed for simulating kinship-based inbreeding depression (important for CNS scenarios) was prohibitive with a model metapopulation of this size. Instead, CNS scenarios were run as a stand-alone subpopulation. The dispersal of juvenile mice across the boundary between CNS and Beach2 was modeled using the Harvest and Supplementation options in *VORTEX*. Movement of mice out of Beach2 to CNS was modeled by harvesting 1.74 (x $N/K_{initial}$) juveniles per ts. Results from the CNS status quo model were used to estimate the average number of mice (0.35 juvenile mice per ts) expected to disperse from CNS to Beach2, ending in 2060, as there is a high risk of extinction of the CNS subpopulation by then. The Supplementation option assumes that new mice (immigrating from CNS) are unrelated to the Beach2 subpopulation (and to each other). While this method underestimates inbreeding and overestimates gene diversity due to CNS-Beach2 dispersal, this small level of migration is genetically inconsequential to the large and genetically diverse Beach2 subpopulation.

Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in Beach2 and Inland was estimated using a combination of projections for sea level rise (SLR) and high tide flooding (HTF) from NOAA in 2017 (as reported in the 2020 SSA) and in 2022 (provided by PVA Team), GIS analysis, and extrapolation. Remaining SEBM habitat was modeled based on the High SLR and Moderate HTF scenarios for Trident Pier for the years 2050, 2080 and 2110. Loss of SEBM habitat in years (and time steps) between these point estimates was extrapolated, assuming that the loss will be linear between point estimates (see Appendix for calculation details and assumptions). K was calculated based on the density of 9 mice/ha (Beach2) or 4 mice/ha (Inland) x remaining ha of habitat (Figure 61). Note that the *VORTEX* model is not spatially explicit, and each subpopulation is assumed to be panmictic. **By 2100, about 55% of SEBM Beach2 habitat is estimated to be lost**. Figure 61 illustrates how this projection of habitat loss is driven by HTF (estimated at 2.76ft) until about 2063, when SLR overtakes HTF estimates and drives further habitat

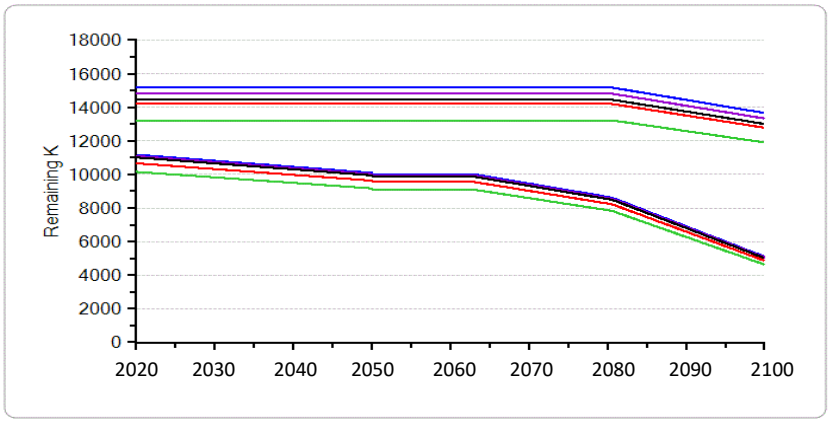


Fig. 61. Estimated K for SEBM Beach2 and Inland subpopulations over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110. Lines represent different impact zone scenarios.

loss. **Inland SEBM habitat is not expected to be affected by SLR and HTF until 2080, but is estimated to lose about 10% of SEBM carrying capacity in the subsequent 20 years**.

The 2020 SSA reports that the corridor connecting the CNS and Beach2 subpopulations will narrow by 2050 and will be broken by 2070 when the 2017 SLR projection is 4.46ft. The 2022 projections achieve this same level in 2080; thus, the **SQ models assume loss of connectivity in 2080**.

Impacts of storms

The Cape metapopulation is estimated to have the same risk of experiencing a hurricane (annual risk of 4%, occurring in summer and fall) as other SEBM dune habitat. The PVA team estimated the following impacts if a hurricane occurs: 1) 10% mortality of mice during the initial timestep; and 2) no successful reproduction (Beach2) or 50% normal reproduction (Inland) during the initial timestep. Habitat carrying capacity is not affected; while Beach2 is estimated to have some loss (erosion), this is estimated to be balanced by gain (accretion) in other areas, with no net change in K. Direct mortality is estimated to be lower on the Cape, as mice have more options to move to less affected areas. To simulate this, the dispersal rate between Beach2 and Inland is doubled for two timesteps (initial and next ts) in the model. The same annual risk and impacts were applied to Beach2 for Nor'Easters (occurring in March-April), but no Nor'Easter impacts were applied to Inland except for increased dispersal with Beach2.

Beach erosion

Erosion is occurring in the northern areas of Beach2, while beach is accreting in the south around the Cape. As with storm impacts, the model assumes no net loss of SEBM habitat due to erosion that is not already covered by SLR and HTF projections.

Other threats/management

Invasive species impacts and habitat management assumed to remain the same as in 2020 at the onset of the projection.

Status Quo Model Results

Like the base demographic model, the Status Quo model for the Beach2 and Inland subpopulations displays seasonal fluctuations in population size. Figure 62 illustrates the seasonal dynamics of mean population size for each subpopulation and for the metapopulation (i.e., Cape=Beach2+Inland), as well as the standard deviation around that mean, for the first five years of the model (January 2020-

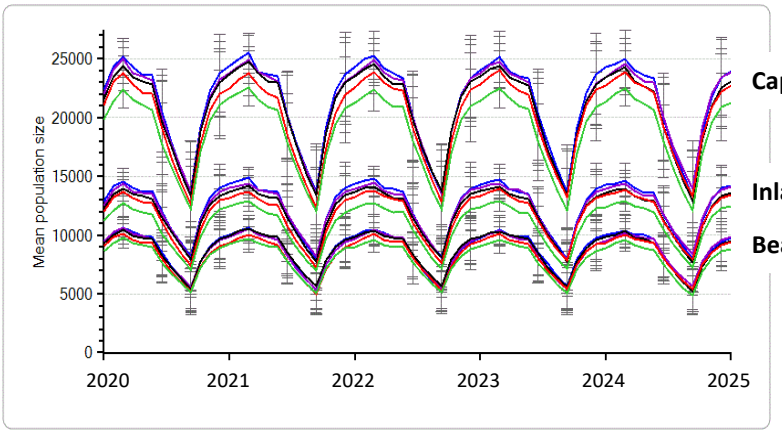


Fig. 62. Mean population size for the first five years of the Status Quo model, with different impact zone assumptions. Bars represent ± 1 SD.

December 2024) and across the five different impact zone scenarios. This seasonal pattern is a reasonable representation of beach mouse population biology.

The Status Quo model combines the impacts of demographic and genetic stochastic processes with loss of habitat (carrying capacity) and connectivity due to progressive sea level rise and development (launch sites). These factors have been parameterized as accurately as possible based on best available data and expert opinion; however, many of these parameters are not well understood. The model results are influenced by the model inputs and assumptions. **This model should be viewed as a tool to investigate the primary driving factors influencing viability.** This can help us to understand the degree of uncertainty in model projections, identify the most important data gaps (and potential research needs), and help to prioritize management actions that are most likely to be effective.

Given the model assumptions and inputs, the resulting projection over 80 years for the Cape SEBM subpopulations and metapopulation is a genetically diverse and viable population, vulnerable to future decline as sea level rises. Both subpopulations show no extinction risk (i.e., 100% persistence) in all impact zone scenarios, and maintain >98% gene diversity with little inbreeding (≤1.5%). Figures 63 and 64 show the mean size of each subpopulation for the two impact zone scenario extremes (no launch zone impacts vs 1/2-mile zone with no mice). Mean population size remains high for the Inland subpopulation until 2080 and then begins a slight decline. The Beach2 subpopulation shows a gradual decline that accelerates starting in 2080. Impact zone assumptions have a greater effect on the mean size of the Inland subpopulation than on the Beach2 subpopulation but do not affect long-term viability

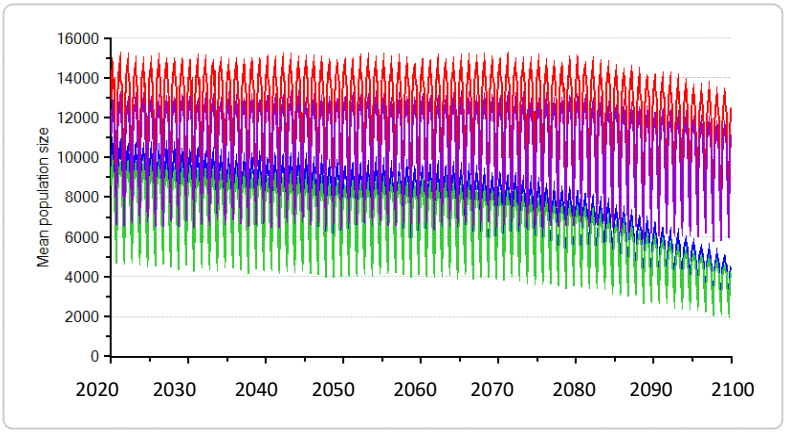


Fig. 63. Mean population size for the Inland and Beach2 subpopulations over 80 years, under two impact zone assumptions. Each timestep (ts) is plotted, demonstrating seasonal variation in mean population size.

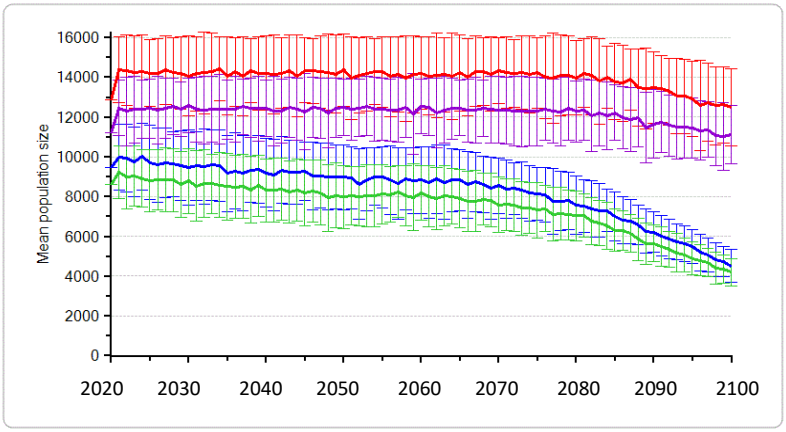


Fig. 64. Mean population size in December for the Inland and Beach2 subpopulations over 80 years, under two impact zone assumptions. Each 13th ts is plotted, demonstrating variation in mean population size in December. Bars represent ± 1 SD.

given the model assumptions. Sea level rise (and HTF) have a greater impact on the Beach 2 sub-population and result in substantial decline (>50%) in SEBM numbers in the dune habitat over 80 years.

Uncertainty in SQ Projections

There are several areas of uncertainty in model inputs for Status Quo projections for the SEBM subpopulations on the Cape. These include data gaps related directly to beach mice as well as to SEBM habitat and threats. Exploratory scenarios presented below used the 1/2-mile with no mice impact zone scenario as a base, as it is the most conservative estimate of subpopulation size.

Mouse density

One area of uncertainty is SEBM density in the two different habitat types. Exploratory scenarios were run to test other densities. The SQ model uses 9 mice/ha as the density for Beach2, the highest SEBM density estimate in this PVA. An alternative scenario based on 7.5 mice/ha for Beach2 was also tested (i.e., density estimate for good habitat in SDP). While mean subpopulation is slightly smaller with this lower density due to smaller K (Figure 65, red line), Beach2 retains high gene diversity (>98%) with no risk of extinction at this density. Although the two subpopulations are connected, with more mice on average going from Beach2 to Inland, lower density in Beach2 does not affect the Inland subpopulation, and the combined metapopulation is slightly smaller.

The SQ model uses a density of 4 mice/ha averaged across the heterogeneous habitat of the Inland subpopulation. This is an estimate, as there are few data for SEBM density in scrub habitat. A higher density of 5 mice/ha was also tested, which could represent an adjustment to a possible underestimate of current mouse density, or alternatively could represent habitat management that increases the carrying capacity of inland habitat for SEBM. As shown in Figure 65 (green line), higher density and K results in larger mean population size (with high gene diversity and no extinction risk). There is no discernable impact on the Beach2 subpopulation, and the combined metapopulation is slightly larger.

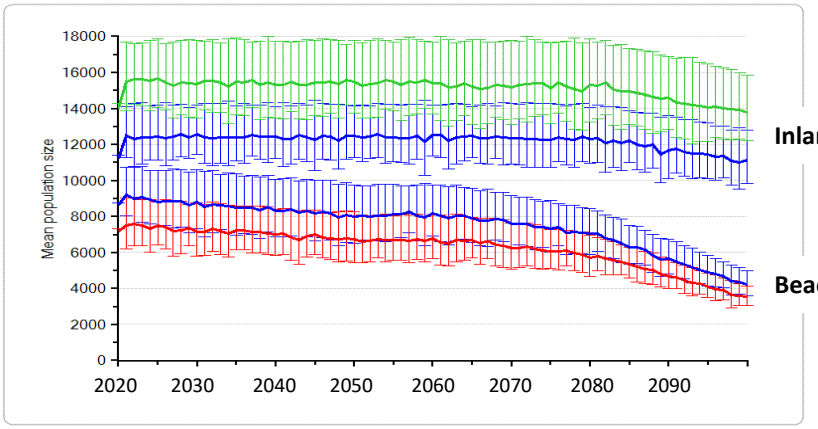


Fig. 65. Mean population size in December over 80 years for Inland and Beach2 (12z0m), with different densities. Each 13th ts is plotted. Blue=SQ; red=Beach2 7.5m/ha; green = Inland 5m/ha.

Demographic rates

In the absence of available demographic data for SEBM in scrub habitat, the model assumes that demographic rates and seasonality are the same for the Inland subpopulation as for Beach2. Alternative scenarios were run that reduced reproduction by 25% or 50% in the Inland subpopulation during the three timesteps in the Fall to serve as a proxy for lower lambda in scrub habitat. Fall is the season of

highest population growth in the model; thus, reducing reproduction in the Fall dampens the seasonality of population growth in Inland habitat. This reduction in reproduction results in a decrease in stochastic growth r from 0.0993 (SQ) to 0.0790 (25% reduction in Fall reproduction) and 0.0549 (50% reduction in Fall reproduction). The model result is an approximately 7% and 18% reduction, respectively, in mean population size for the Inland subpopulation in Winter (Figure 66) when population size is high, although there is little to no impact on mean population size in Spring and Summer. These reduced demographic growth rates have no impact on Beach2 and does not change the overall viability projections in terms of population trend, gene diversity or extinction risk. This suggests that, while better data for demographic rates of SEBM in inland habitat would be valuable to improve population projections for beach mice living in scrub or ruderal habitat, it is not essential to Inland subpopulation viability projections.

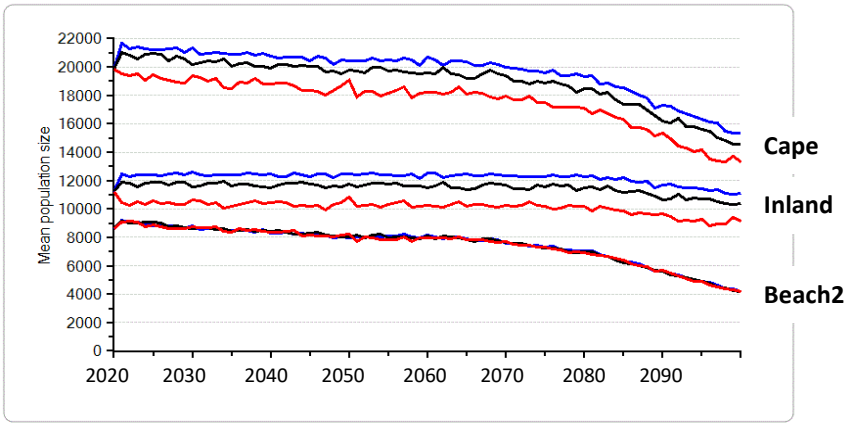


Fig. 66. Mean population size in December over 80 years for Beach2, Inland and the metapopulation (12z0m), with different Fall reproduction in Inland. Each 13th ts is plotted. Blue=SQ; black=75% fall reproduction in Inland; red=50% fall reproduction in Inland.

SQ Inland
Rep75 Inland
Rep50 Inland

Dispersal rates

The model also assumes that juveniles randomly disperse between the two habitat types (with increased movement during major storm events). Given the higher density of mice in Beach2 in narrow habitat, this means that a greater proportion of SEBM in Beach2 live within the transboundary dispersal area than in Inland. In the model, this results on average in more mice moving from Beach2 to Inland than in the other direction, assuming that both populations are near carrying capacity. If this dynamic is accurate, then the Beach2 subpopulation potentially could be acting as a source for both CNS and Inland. Alternative scenarios were run (using the 1/2-mile with no mice impact zone scenario as a base) with 50% reduction in Beach2-Inland dispersal rates (both directions) and with no movement of mice between Beach2 and Inland to better assess the impact of transpopulation movement. Both subpopulations are projected to have good demographic and genetic long-term viability if transpopulation dispersal reduced or no exchange. Mean population size remains the same for both subpopulations and with no extinction risk. Gene diversity is slightly reduced if there is no dispersal between these two subpopulations but still remains high (>97% for Beach2, >98% for Inland). Therefore, the uncertainty regarding the movement rate of SEBM between coastal and inland areas is not an issue of concern for viability projections given other model assumptions.

Impacts on carrying capacity

The impact of disturbance around launch sites on beach mice is not known, including potential effects on density, reproduction and survival. Under the model assumptions (i.e., reduced density but no demographic effects), results show that exclusion of beach mice up to 1/2 mile surrounding these sites

leads to lower K (and thereby lower N) (Figures 63 & 64) but does not significantly alter long-term population viability.

Sea level rise projections result in projected SEBM population decline, especially in dune habitat (Beach2) and especially after 2080, again via a reduction in K over time. While mouse numbers are projected to decline, the results show no extinction risk over 80 years, given model assumptions. Given the rate of sea level rise in the last 20 years of the model (and projected to continue at least to 2110), SLR could drive continued decline of this subpopulation beyond the projected timeline.

The reduction in habitat (K) due to major storms was found to be a significant driver of SEBM population viability for SEBM living in narrow beach habitat of Smyrna Dunes Park and Canaveral National Seashore. The Cape metapopulation SQ model does not include any impacts of storms on SEBM habitat, as the model assumes that there is no net loss of K (i.e., accretion ~$\geq$ erosion). *VORTEX* is not spatially explicit, and treats the subpopulation as panmictic. While there may be no net loss of K, in reality the number of mice may decline along the northern coast and increase along the southern beaches as habitat shifts.

Finally, if any of these habitat impacts (launch sites, SLR, storms) have different or greater impacts than modeled here, this could impact long-term viability especially for the smaller Beach2 subpopulation.

Summary

PVA models project SEBM subpopulations on the Cape (both dune and inland habitat) to persist and remain genetically diverse to 2100, given model assumptions. **Both launch sites and sea level rise may result in smaller SEBM numbers over time, especially in dune habitat**, but are not projected to threaten SEBM with extinction on the Cape in the next 80 years. Improved SLR projections and better understanding of potential storm impacts may strengthen these projections. Data gaps regarding SEBM distribution, density and demographic rates in inland habitat are not likely to affect overall population viability projections, but would strengthen the precision of such projections.

Source Population Scenarios

The Beach2 subpopulation has been proposed as a potential source population for SEBM for any translocation efforts, including reinforcement of existing populations such as SDP or reintroduction efforts to reestablish SEBM in unoccupied habitat. Several alternative management scenarios were developed to explore the ability for the Beach2 subpopulation to provide juvenile mice for release without negatively impacting Beach2 viability. These scenarios assume that the Beach2 subpopulation continues to exchange mice with both CNS and Inland (SQ model), recognizing that more juvenile mice are estimated to move out of Beach2 along these boundaries than into Beach2 and so transboundary migrations could affect these results.

The PVA team concurred that translocations are most likely to occur in October. Model results show that removing 200 juvenile mice (~equal sex ratio) *every October* has no impact on the Beach2 subpopulation viability, with <1% of attempted harvests not able to be completed due to insufficient number of juveniles. These results apply for SEBM densities in Beach2 of either 9 mice/ha (SQ) or 7.5 mice/ha. No conditions were set in the model to alter harvest if a recent hurricane occurred. This number of harvests could support one large or several smaller translocation events.

It is possible that some mice may be needed or desired for translocation during other times of the year. Results from additional scenarios show that 50 juvenile mice can be removed *every timestep* from Beach2 with no impact on mean population size or gene diversity and with no risk of extinction, regardless of which density is applied. Removing 100 mice *every timestep* results in a smaller mean population size (~16-25% reduction by 2100, depending upon density) but does not reduce gene diversity or incur risk of extinction. These results suggest that the subpopulation should be able to sustain occasional harvests in months outside of October. However, continual year-around trapping has a higher risk of being incomplete (i.e., insufficient juveniles to achieve the full harvest number), with 6.5-8% of timesteps resulting in incomplete harvest. While harvest in the model has access to all juveniles in the subpopulation, actual trapping success restricted to particular locations may be substantially less.

Like population size, the number of juveniles in the subpopulation is expected to vary seasonally, which is a likely contributor to incomplete harvests in the *VORTEX* model. On average, fewer juveniles are available in summer months when reproduction is lower and first-month mortality is high. Figure 67 shows the mean (and SD) number of 4-week-old mice in the Beach2 subpopulation for each timestep (except ts7, which is similar to ts6), assuming the conservative estimates of lower density of 7.5 mice/ha and 1/2-mile impact zones with no mice (12z0m). Juveniles were counted after normal mortality was applied but prior to truncation to K. Graphs in the left column represent winter and spring, while those in the right column represent summer and fall. Months indicated should be considered as approximate. While juvenile numbers are high during much of the year, there is significant variation in numbers during much of the year and especially in spring, sometimes bringing juvenile numbers close to zero in "bad" years. In contrast, the number of juveniles in the fall remains high and less variable, with 95% of the distribution of model results above 500 juveniles. These model results suggest that fall and early winter trapping is mostly likely to result in sufficient juveniles trapped to meet translocation efforts.

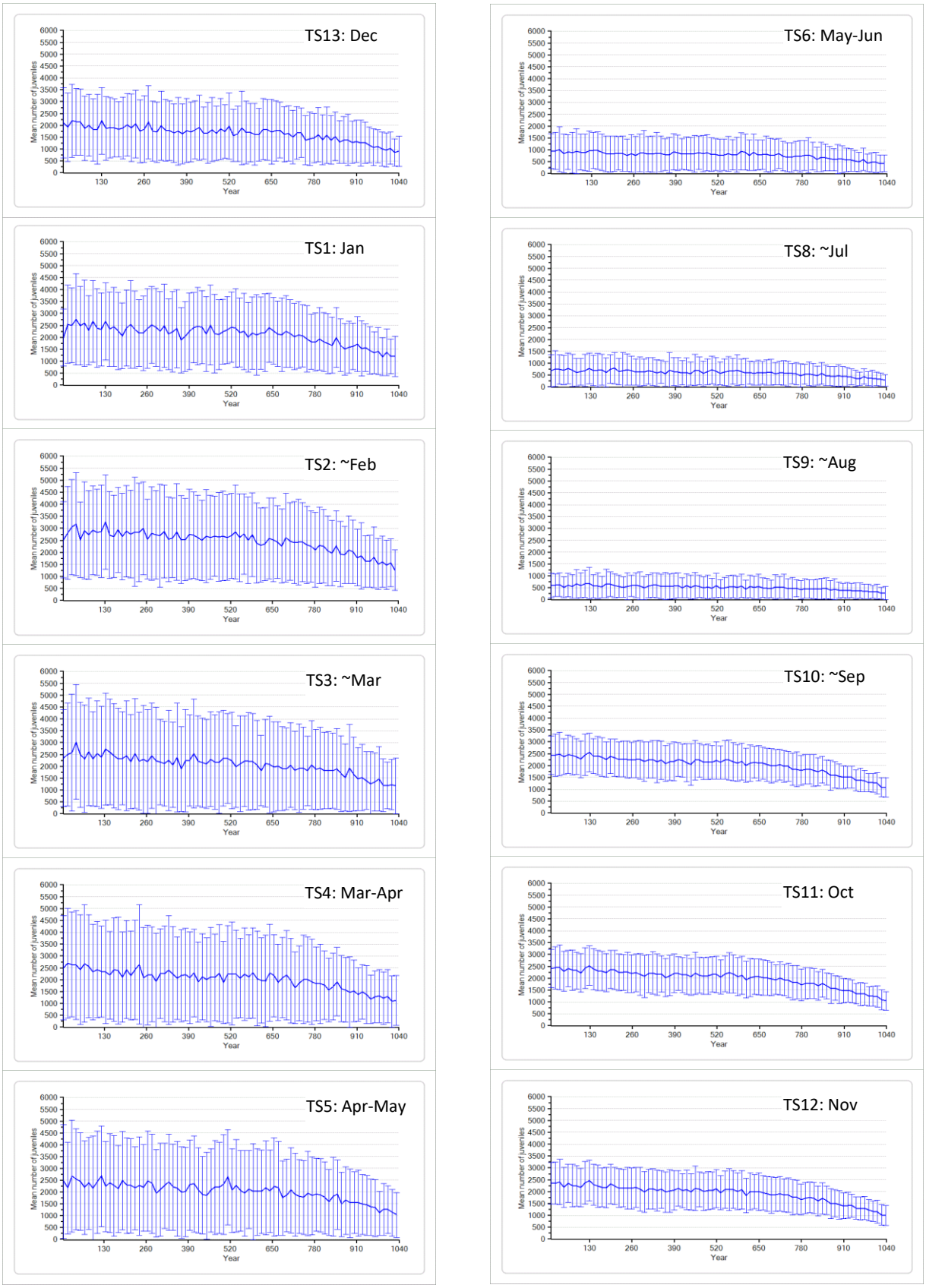
Together, model results project that the Beach2 subpopulation can be considered as a viable source for mice for conservation translocation efforts with no loss of long-term viability. Trapping success rates and population monitoring may be useful tools in assessing the real impact if translocation efforts include numerous large and/or prolonged harvests.

Summary

A Status Quo metapopulation model for the SEBM Beach2 and Inland subpopulations was developed based on the best available data and expert opinion. Given model inputs and assumptions, the projected long-term viability of SEBM on Cape Canaveral is good, with high gene diversity and no risk of extinction over 80 years, but with projected decline in numbers as habitat is lost due to projected SLR. The key driving factors affecting this projection are sea level rise and high tide flooding for the Beach2 subpopulation. The greatest uncertainty surrounding these projections is how or if demographic rates differ between habitat types and if there are any impacts of storms, especially on habitat, that are not included in these projections.

No specific habitat management scenarios were modeled, but scenarios that involved different K (i.e., different density, different launch impact zones) can serve as a surrogate for increased or decreased K. Alternative scenarios that explored the removal of juvenile mice indicate that the Beach2 subpopulation is a viable option as a source for SEBM translocation efforts.

Fig. 67. Mean number of juvenile mice over time by timestep (12z0m; 7.5m/ha); every 13th ts plotted. Bars=SD.



STATUS QUO VIABILITY PROJECTIONS: SUMMARY

The Status Quo model scenario results represent population viability projections for the Southeastern beach mouse based on the best available information and expert knowledge. These projections incorporate demographic and genetic stochastic processes as well as projected threats to SEBM and SEBM habitat and assume all other threats and management not explicitly modeled remain constant.

While there is no singular definition of population viability, common indicators include population persistence, demographic growth rate or sustainability, population size, and high genetic diversity. For this SEBM PVA, the PVA team identified extinction risk, population size, and gene diversity as measures to be calculated at years 2050, 2070 and 2100 to match time benchmarks in the 2020 SSA. Table 5 details these measures for each of the four existing subpopulations as well as the entire combined metapopulation. Results are given for the end of the second timestep of the year (late February); this generally represents the time of peak population size in the model. Probability of extinction (PE) is calculated as the proportion of iterations that do not contain at least one individual of each sex at that timestep. N_{all} represents the mean population size calculated across all iterations (including N=0 for extinct iterations), while N_{ext} represents the mean population size calculated across only those iterations that are not extinct. GD_{ext} represents the mean gene diversity (relative to starting level in 2020) calculated across only those iterations that are not extinct.

The projected viability of SEBM varies across the taxon's range. Given model inputs and assumptions, the projected long-term viability of SEBM on Cape Canaveral (Beach2 and Inland subpopulations) is good with respect to genetics and persistence, with SEBM typically numbering in the 1000s, retaining high gene diversity ($\geq$ 98.5%), and showing no risk of extinction over 80 years. Short-term population trend is good, but future population numbers are projected to decline as habitat is lost due to projected SLR. This is especially of concern in dune habitat, although connectivity spreads the effects across the Cape.

Table 5. Population viability indicators for Status Quo Scenarios at years 2050, 2070 and 2100, by subpopulation. PE=probability of extinction; N_{all}=mean N for all iterations; N_{ext}=mean N for only iterations that are not extinct; GD_{ext}=mean GD for iterations that are not extinct. Ns given for late February (average peak population census).

Subpopulation	Indicator	2050	2070	2100
Smyrna Dunes	PE	0.366	0.91	1
	N_{all}	105	7	0
	N_{ext}	165	82	0
	GD_{ext}	0.629	0.434	--
Beach1 (CNS)	PE	0.048 / 0.088	0.212 / 0.290	0.968 / 0.992
7 mice per ha / 5 mice per ha	N_{all}	143 / 91	48 / 36	4 / 1
	N_{ext}	150 / 100	61 / 50	110 / 72
	GD_{ext}	0.910 / 0.905	0.883 / 0.882	0.599 / 0.460
Beach2 (Cape)	PE	0 / 0	0 / 0	0 / 0
No zones / ½ mile with 0 mice	N	9398 / 8468	9032 / 8101	5026 / 4547
	GD	0.995 / 0.994	0.992 / 0.991	0.987 / 0.985
Inland	PE	0 / 0	0 / 0	0 / 0
No zones / ½ mile with 0 mice	N	14763 / 12880	14770 / 12846	13219 / 11564
	GD	0.996 / 0.995	0.993 / 0.992	0.989 / 0.987
Metapopulation	PE	0 / 0	0 / 0	0 / 0
Best case / worst case	N	24476 / 21348	23945 / 20947	18355 / 16111
	GD	~0.996 / ~0.995	~0.993 / ~0.992	~0.989 / ~0.987

In contrast, projected long-term viability is poor for the smaller SDP and CNS subpopulations inhabiting habitat more vulnerable to sea level rise and erosion due to major storms (hurricanes and Nor'Easters). These smaller subpopulations are projected to decline and eventually go extinct without management intervention. The key driving factors affecting these projections are habitat loss (especially loss due to storms that can lower the carrying capacity for mice beyond sea level rise and high tide flooding impacts) and inbreeding depression. Inbreeding threatens SDP due to its small size and genetic isolation, while CNS is threatened both demographically and genetically by the risk of fragmentation and projected loss of connectivity with the Cape in the future due to sea level rise.

The greatest uncertainty surrounding these PVA projections is around these two key drivers. Most of the storm-related effects on habitat and K are estimated in the model, yet they significantly affect projected viability. This important source of uncertainty has two implications:

1) **Research and monitoring to better understand the impact of storms on SEBM habitat, and the post-storm recovery of that habitat** with respect to its ability to support SEBM, would be valuable to address these important data gaps; and

2) **Management actions that** reduce storm damage to SEBM habitat, facilitate more rapid and/or more complete recovery, or otherwise **improve the carrying capacity of the habitat for SEBM especially following major storm events** will promote SEBM numbers, persistence and viability.

Similarly, genetic processes affect demography and therefore viability projections in the model, yet are estimated. Implications from this uncertainty include the following:

1) Continued monitoring in the field, as well as possible *ex situ* studies, may lead to a **better understanding of the sensitivity of SEBM to inbreeding (and outbreeding) depression**. Such information would inform the need for and estimated benefits of genetic augmentation, and help promote success and reduce risks in genetic augmentation efforts.

2) **Management actions that help to maintain or restore connectivity between SEBM subpopulations or prevent fragmentation** are likely to improve viability and reduce the need for genetic augmentation. This is especially true for narrow dune habitat on barrier islands, such as the Canaveral National Seashore.

Modeling of alternative management scenarios for SDP and CNS suggest that effective management may need to be two-fold: prolonged periodic genetic augmentation, in concert with efforts to restore, maintain and/or increase SEBM habitat and its carrying capacity to support beach mice. While all potential habitat management actions were not explicitly modeled in this PVA, any efforts to increase K for beach mice are likely to be beneficial, including post-storm habitat restoration as well as other potential habitat management strategies. Smaller subpopulations are more likely to require periodic genetic augmentation, especially if isolated. In general, the larger and more stable the habitat for a SEBM subpopulation, and the better the connectivity with other SEBM subpopulations, the less management effort will be needed to promote population viability.

While PVA results suggest good future viability for SEBM on Cape Canaveral, this alone does not meet the PVA team's vision for the taxon as "to have a viable metapopulation across the taxon's former range comparable to that at the time of Federal listing in 1989". The potential need for, and benefits of, population reinforcement of other existing SEBM subpopulations (i.e., SDP, CNS) was evaluated through

genetic augmentation scenarios in the preceding report sections for existing subpopulations. However, re-establishment of SEBM populations in some areas of former occupancy also would be needed to achieve this vision. The following report section explores potential reintroduction of SEBM to each of three former sites in the taxon's range where SEBM are believed to be extirpated. Model results indicate that the SEBM subpopulation in dune habitat on the Cape is likely of sufficient size, demographic health, and genetic diversity to serve as a source population for periodic translocations to reinforce existing SEBM subpopulations and/or in reintroduction efforts to re-establish extirpated SEBM populations in former habitat.

A final note: The PVA model results presented in this report are influenced by the model inputs and assumptions surrounding them. As described above and in each subpopulation PVA section, there are numerous areas of uncertainty, some of which may be inconsequential while others may significantly affect future viability projections. Therefore, it is suggested that readers and managers do not focus on precise estimates of future population projections found in this report, but rather view this model and scenario results as a tool to better understand the primary driving factors influencing SEBM viability and to identify the most vulnerable subpopulations and situations. As outlined above, this can help to guide species and habitat managers in addressing important data gaps through relevant research and monitoring, and assist efforts to prioritize management actions that are most likely to be effective to meet program goals across the taxon's distribution and former range.

SOUTHEASTERN BEACH MOUSE REINTRODUCTION SCENARIOS

The Southeastern beach mouse (SEBM) has been extirpated from parts of its historical range, including likely some inland areas on Cape Canaveral as well as areas south of Point Canaveral Entrance Channel. In order to achieve the PVA team's vision to secure a viable metapopulation of SEBM across the historical range at the time of listing (1989), future reintroductions likely will be needed to re-establish extirpated SEBM subpopulations. Successful reintroductions will help to increase the Redundancy, Resiliency and Representation across the subspecies' range, especially when considering the potential loss of habitat to sea level rise (USFWS 2020). Three reintroduction areas were explored in this PVA:

1) Happy Creek (HC): inland habitat on Cape Canaveral, north of the Inland subpopulation;
2) Sebastian Inlet to Pelican Island (SIPI): narrow length of dune habitat ending with wider inland habitat, on Orchid island south of Sebastian Inlet; and
3) Avalon State Park to Fort Pierce Inlet State Park (AFP): narrow dune habitat on North Hutchinson Island, south of Orchid Island.

Each of these potential reintroductions was modelled through a series of scenarios to explore potential translocation schedules and parameter uncertainty (500 iterations per scenario). All translocations are assumed to be juvenile (4 week old) mice of approximately equal sex ratio, trapped from the Beach2 subpopulation. The *VORTEX* model assumes these mice to be unrelated to each other (or, more accurately, equally slightly related at the same background level as the entire Beach2 subpopulation). Therefore, this assumes that translocated mice are trapped from different locations within Beach2 SEBM habitat. The same level of inbreeding depression was applied to translocated mice as were used in the Beach2 and Inland subpopulation models (4.72 LE), applied via either 100% (HC) or 50% (SIPI & AFP) lethal alleles. Unless otherwise indicated, translocations are modeled to occur in October.

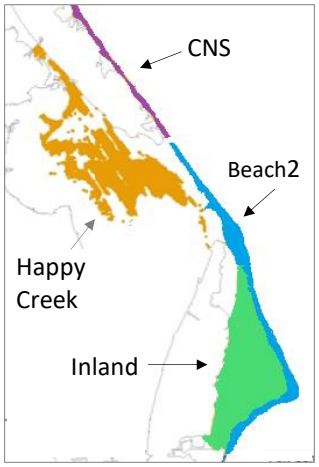
Little information exists on the survival and reproduction of beach mice that are translocated between wild populations, either for population reinforcement or reintroduction. To address this uncertainty, the following strategy was followed and is the same as that used in the SDP and CNS models. No trans-location mortality is included in the model, and the model assumes that translocated mice have the same mortality and reproductive rates as resident mice. Essentially this means that the number of releases in the model reflect the number of *successful* releases. The number of actual mice needed to be released to achieve this level would need to be calculated based on the success (survival and reproduction) observed for actual translocations. Since mice are modeled as translocated just as they reach 4 weeks of age (one timestep), they cannot produce offspring for the first two timesteps after release.

These reintroduction scenarios have been parameterized as accurately as possible based on best available data and expert opinion; however, many of the parameters for SEBM translocation and reintroduction, as well as habitat-related inputs for these potential reintroduction sites, are not well understood. **These reintroduction models should be viewed as a tool to investigate the primary driving factors influencing viability and successful reintroduction. Exploratory scenarios have been included to help us to understand the degree of uncertainty in model projections, identify the most important data gaps (and potential research needs), and help to prioritize or adjust management actions that are most likely to be effective in re-establishing extirpated SEBM subpopulations.**

Happy Creek (HC)

This area consists of 1401 ha of oak scrub (inland) of former SEBM habitat, which is north of the Inland subpopulation and west of the coast (Figure 68, adapted from the 2020 SSA, depicts the general location). Few, if any, SEBM are believed to inhabit this area, which has no current connectivity with other SEBM subpopulations. There is no estimated impact from current launch sites for this area. The PVA team estimated the average carrying capacity of this habitat to be similar to that for the Inland sub-population (i.e., 4 mice/ha), with the potential for higher density (5 mice/ha) with some habitat improvement. Projected density results in an estimated initial K of 5604 mice based on 1401 ha. These scenarios assume that there are no SEBM in this area prior to translocation. Due to the larger potential population size, inbreeding depression was applied via lethal alleles only to avoid very slow computer run times required by other genetic mechanisms.

Fig. 68. General map of existing and potential sites on the Cape.



Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in Happy Creek due to sea level rise and high tide flooding was estimated using the same methodology and projections as for Beach2, assuming equal elevation and impacts on the ocean (Beach2) and bay (Happy Creek) sides of the peninsula. **By 2100, about 51% of SEBM Happy Creek habitat is estimated to be lost** (Figure 69). In relation to this projected decline in K, the first translocations are modeled as occurring in October 2025; however, there is little projected change in K between 2025 (K=~5500) and 2063 (K=~5000) in this area and no other sources of habitat loss, so these general results should be applicable to translocation projects that begin later than 2025.

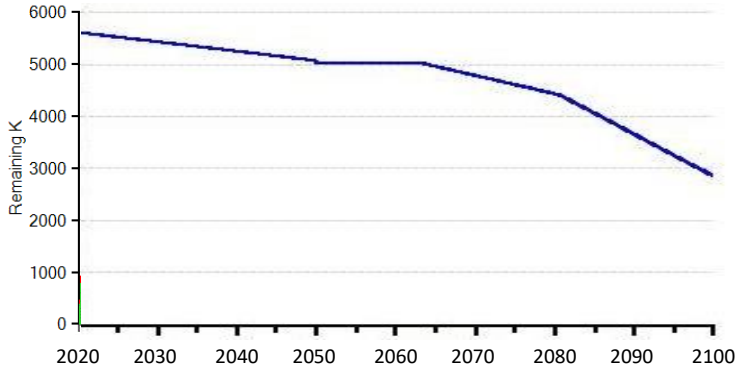


Fig. 69. Estimated K for SEBM habitat in Happy Creek over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110.

Impacts of storms

Happy Creek is estimated to have the same risk of experiencing major storms, and storm impacts, as the Beach2 subpopulation. There is a 4% annual risk of hurricanes (occurring in summer and fall) and 4% risk of Nor'Easters (occurring in March-April), with the following demographic impacts if a storm occurs: 1) 10% mortality of mice during the initial timestep; and 2) no successful reproduction during the initial timestep. No impacts of storms on SEBM habitat were included in these reintroduction scenarios. PVA modeling for the SDP and CNS SEBM subpopulations found that loss of K due to major storms, and incomplete recovery to pre-storm conditions, can have significant impacts on SEBM population viability. While inland oak scrub habitat is not vulnerable to the same beach erosion experienced by dune habitat

along the coast, any significant impacts of storms in this low-lying area could impact the success of reintroduction in this area.

Model Results

Reintroduction scenarios were developed that involved multiple releases of 50 juvenile mice. This release number was found to be effective in modeling of the SDP subpopulation and was also suggested by the PVA team as logistically feasible. The first set of scenarios involve **releases of 50 mice every October for five consecutive years** (total of 250 mice released). Figure 70 illustrates mean population size (and SD) for the five years of releases and subsequent 5 years with no additional releases (January 2025-December 2034). Projected mouse numbers are relatively low for the first ~3-4 years after the first release but then grow and become more seasonally dynamic, reaching ~90% of K, on average, in ~8 years from initiation of the translocation efforts. **Long-term projections indicate good viability, with mean population size tracking K imposed by SLR, and high gene diversity (92.6%) in 2100, with no probability of extinction**. The following analyses explore various areas of uncertainty regarding this projection.

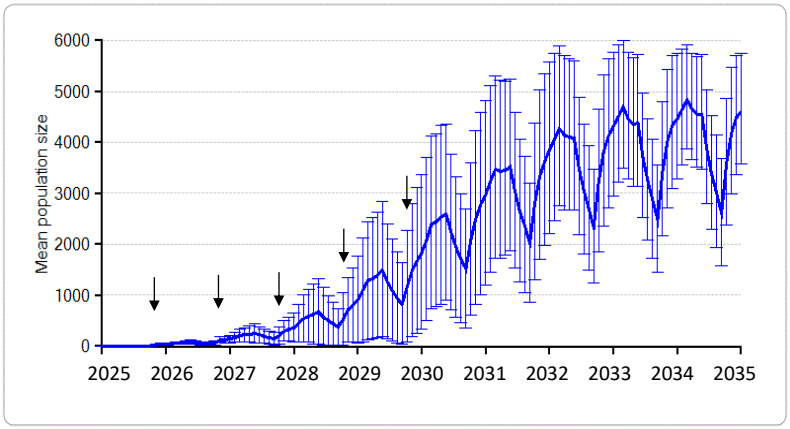


Fig. 70. Mean population size for the five release years and subsequent five years. Bars represent ± 1 SD. Arrows indicate timesteps for releases.

Post-release survival

These results assume the release of 50 juvenile mice that show similar survival and reproductive rates as non-translocated mice. Figure 71 provides the mean number of translocated mice still alive in the model over the same 10-year period. About 5 translocated mice (10% of releases) are still alive one year after release. If post-release survival is significantly lower, additional releases may be needed to match these viability projections.

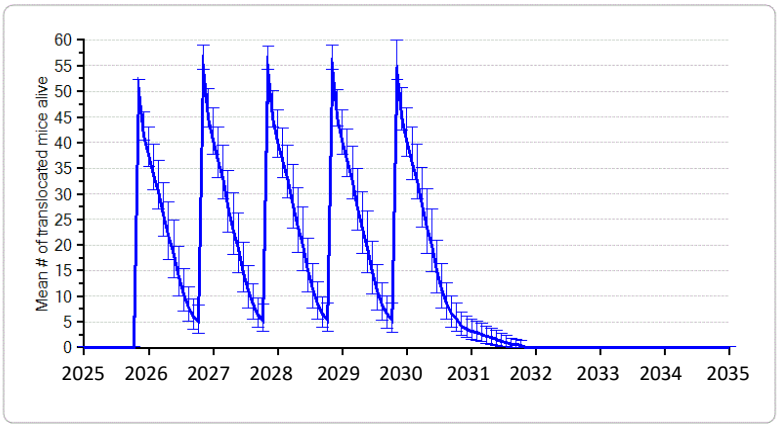


Fig. 71. Mean number of released mice alive over time. Bars represent ± 1 SD.

Mouse density

One area of uncertainty is habitat quality and resulting expected SEBM density. The PVA team estimated an overall density of 4 mice/ha, noting that this would require some initial habitat restoration. Habitat management for other species, such as the Florida scrub-jay (*Aphelocoma coerulescens*), has the potential to benefit or otherwise impact SEBM carrying capacity. Densities of 3 mice/ha and 5 mice/ha were also explored to encompass uncertainty in habitat quality and future habitat management. Lower density means lower SEBM carrying capacity.

Figure 72 provides projections of mean population size for Happy Creek using the same reintroduction scheme modeled above but using different mouse densities. Population size is lower with lower density and lower K, although there is no risk of extinction by 2100 for any of these densities. Gene diversity is lower as density declines from 5 to 4 to 3 mice/ha (93.6%, 92.6% and 91.2% in 2100, respectively) but remains above 90%, while inbreeding remains below 9%. Improved habitat quality (i.e., higher mouse density) improves long-term viabilty, especially as habitat is lost to SLR over time; however, **the reintroduced HC subpopulation shows good projected viability across this range of densities**.

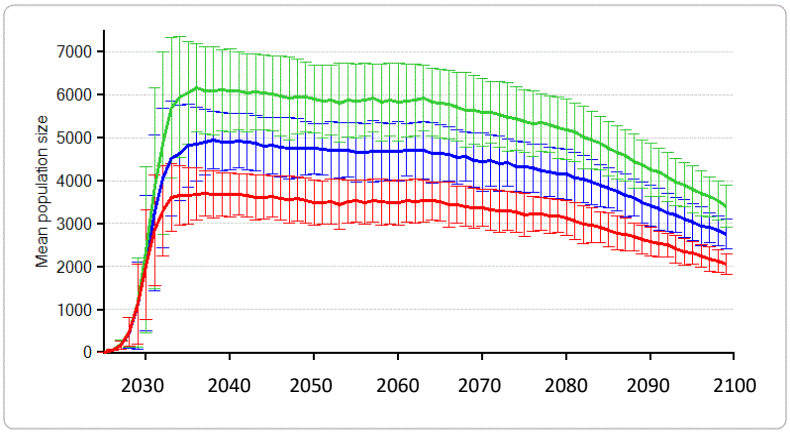


Fig. 72. Mean population size in December over 75 years (2025-2100) for HC reintroduced population, across three mouse densities. Every 13th ts plotted. Bars represent ± 1 SD.

Reproductive rates

As indicated in the Inland subpopulation PVA, there is little available information on the seasonality of reproductive rates in scrub habitat. The base SEBM demographic model is based on SEBM dune data and shows strong seasonality. As was done for the Inland PVA, a HC scenario was tested with 50% reduction in reproduction in the fall (the season of peak reproduction in the base model). This reduction does not impact released mice, as these mice are not old enough to breed (in the model) until winter, and only ~10% of releases survive to the following fall. Rather, reduced reproduction in this scenario primarily impacts HC-born mice. Lower reproduction impacts the ability of the subpopulation to grow following reintroduction, with mean population size ~50% of that based on base model fall reproduction after ~3 years after the first release, despite the same K and same number of mice released (Figure 73). Lower reproduction in fall also reduces the subpopulation's ability to recover from 'bad years' due to environmental variation and major storms. This results in lower mean population size over time, making it less likely that the subpopulation reaches K (Figure 74) and imparting a small extinction risk (2.2%) by 2100. Population size is most affected in the first decades, with the impact of lower fall reproduction lessening over time as K declines with increasing SLR. Smaller initial population size results in more rapid and variable loss of gene diversity due to genetic drift, on average falling below 90% by 2100 (Figure 75).

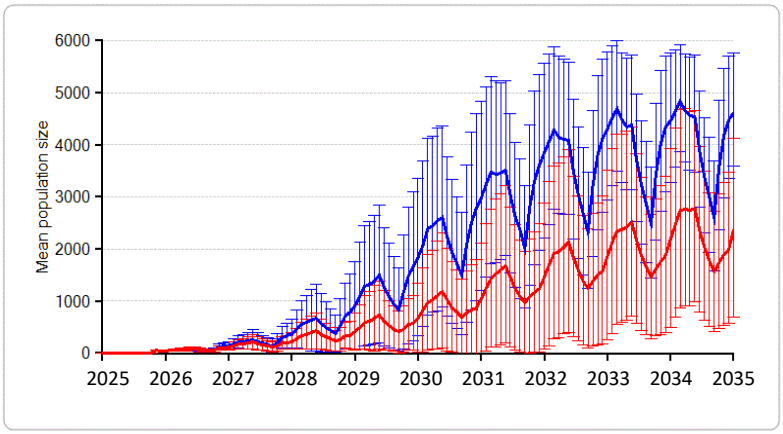


Fig. 73. Mean population size for the five release years and subsequent five years, with SQ and 50% fall reproduction. Bars represent ± 1 SD.

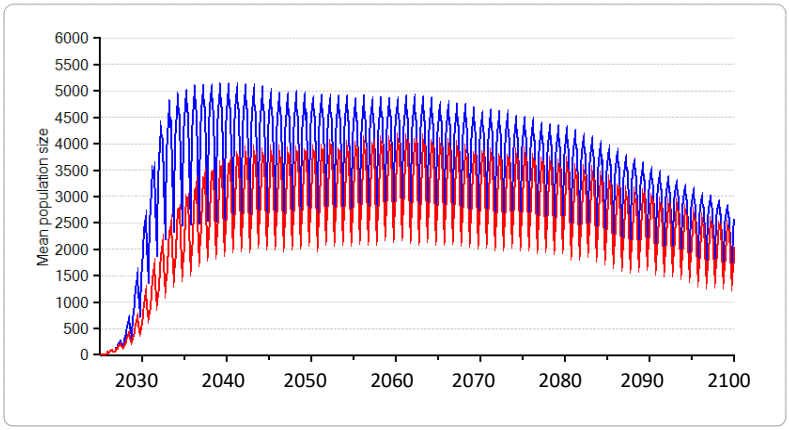


Fig. 74. Mean population size over 75 years (2025-2100) for HC reintroduced population, with SQ and 50% fall reproduction.

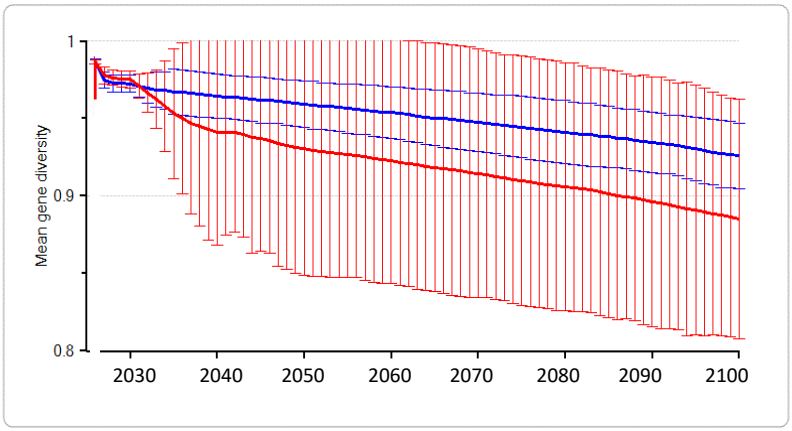


Fig. 75. Mean gene diversity over 75 years (2025-2100) for HC reintroduced population, with SQ and 50% fall reproduction. Bars represent ± 1 SD.

Genetic augmentation

Happy Creek is estimated to have potential carrying capacity for several thousand SEBM. If reintroduced mouse populations are able to grow quickly into the thousands, this favors capturing most of the genetic diversity represented in the founders (released mice) and will minimize genetic drift and accumulation of inbreeding. If mouse populations grow slowly (i.e., lower fall reproduction), the reintroduced population may benefit from periodic genetic augmentation. Periodic releases of 50 mice every 10 years following the initial 5-year reintroduction effort is projected to maintain >90% gene diversity by 2100 (Figure 76). Releases every 5 years results in similar GD in year 2100 as projected with base model fall reproduction. Either scheme of periodic augmentation brings extinction risk by 2100 to zero. This

suggests that periodic releases may be beneficial to the viability of the reintroduction effort depending upon conditions and the response of the mouse population to saturate the available habitat.

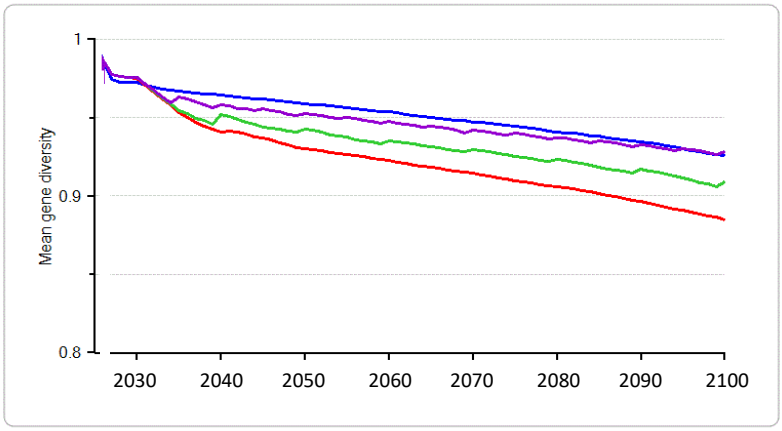


Fig. 76. Mean gene diversity over 75 years (2025-2100) for HC reintroduced population, with SQ and 50% fall reproduction under different augmentation schemes.

Implications for reintroduction in Happy Creek

Happy Creek represents a relatively large area that has the potential to support a viable SEBM subpopulation. Modeling suggests that the degree of translocation effort required to establish and maintain a viable subpopulation is feasible. The size, number and schedule of translocations is impacted by uncertainty both related to SEBM and to the habitat.

Any factors that reduce the carrying capacity of the area for SEBM will reduce population size, including SLR or reductions in habitat management. Other factors that were not included in the Happy Creek model but have the potential to reduce K and mouse numbers are loss of habitat due to storms or to development such as launch sites with impact zones that extend into Happy Creek. Conversely, habitat management efforts to improve or restore habitat quality for SEBM will promote larger SEBM numbers and minimize the need for additional augmentation.

There is some uncertainty regarding SEBM survival and reproduction in the Happy Creek area. Poor post-release survival or reproduction may require larger or more frequent releases to establish the subpopulation. Population growth in the initial years of reintroduction are important in capturing gene diversity and reducing inbreeding, suggesting value in short-term management efforts to support mice and habitat quality. Monitoring of beach mice to understand population status and support effective responsive management will be a valuable and recommended component of any reintroduction efforts.

A final, and perhaps most important, area of uncertainty is why there is currently no SEBM population in this area. Happy Creek is believed to have little, if any, connectivity with existing SEBM subpopulations. SEBM inhabited this large area in the past but now HC is estimated to have few if any SEBM. **It is important to ascertain if there is some threat or factor present that is not accounted for in this PVA**, such as disease, inadequate food or nesting resources, competing species, natural or invasive predators, or human disturbance. Any reintroduction program should follow the IUCN SSC *Guidelines for Reintroductions and Other Conservation Translocations* (IUCN/SSC, 2013), which recommends efforts to confirm that threat(s) that caused extirpation have been identified and removed or sufficiently reduced.

Sebastian Inlet to Pelican Island (SIPI)

Two potential reintroduction sites south of the Cape were included in this PVA, and lie on Orchid and North Hutchinson Islands indicated in green in Figure 77 (from the 2020 SSA). The potential reintroduction area on Orchid Island consists of habitat approximately from Sebastian Inlet through Sebastian Inlet State Park south to and including Pelican Island National Wildlife Refuge. SEBM habitat includes 193ac (78ha) of dune habitat along ~7 miles of coastline, as well as 205ac (83ha) of inland habitat in Pelican Island NWR that is separated from the dune habitat by State Road A1A. Figure 78, adapted from the 2020 SSA, illustrates this approximate area that consists primarily of public lands (yellow) separated by some private lands (blue).

Fig. 77. General map of existing and potential SEBM sites.

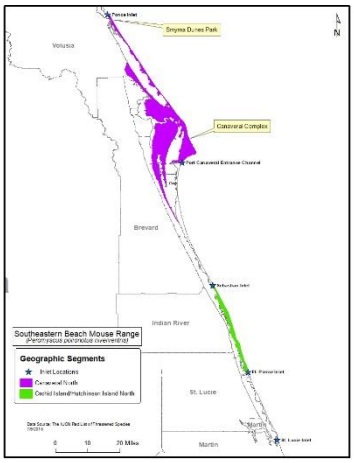


Fig. 78. Orchid Island habitat.

The PVA for SIPI reintroduction area was developed as a meta-population with two subpopulations: dune habitat (labeled in the PVA as 'Sebastian' but includes dune habitat along the length of this area), and inland habitat (labeled here as 'Pelican'). The PVA team estimated the average carrying capacity of the dune habitat as 5 mice/ha, while K for the inland habitat to be 4 mice/ha. Projected density results in an estimated initial K of 390 mice (Sebastian) and 332 mice (Pelican). These scenarios assume that there are no SEBM in this area prior to the first translocation.

Dispersal between habitats

The general reintroduction strategy modeled here is the **simultaneous release of mice in two locations** – Sebastian Inlet SP and Pelican Island NWR – with the expectation that populations would expand along the coast from the north and across A1A in the south. This model approximates the exchange of mice between these two habitats/ subpopulations through inter-population dispersal using the same approach as for other SEBM subpopulations (e.g., CNS-Beach2-Inland). The inter-population boundary was estimated to be 1.23 miles of State Road A1A separating PINWR from the southern dunes. Because the habitats do not abut directly but are separated by a road, dispersal across the road is expected to be less than random dispersal from natal burrows in any direction. Expected dispersal across the road was reduced by 50% of that expected with random movement (i.e., from ~3% to 1.5% of juveniles of each subpopulation), with 5% mortality applied to road-crossing individuals to account for some road-related mortality). Dispersal was allowed in the model once the initial five years of releases were completed, at which time both subpopulations would be expected, on average, to have expanded to near K.

Inbreeding impacts

Inbreeding depression was set at the same level as the Beach2 source subpopulation (4.72 LE). Computational constraints necessitated applying inbreeding depression in relatively large SEBM subpopulations (Beach2, Inland, HC) only via recessive lethal alleles (100%). Since the maximum population size (K) is relatively small for SIPI, inbreeding impacts can be applied using 50% due to recessive lethal alleles and 50% due to reduced survival of inbred juveniles (as for CNS and AFP).

Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in SIPI due to sea level rise and high tide flooding was estimated based on 2022 projections using High SLR and Moderate HTF scenarios for Trident Pier and applying the same methodology as was used for the other SEBM subpopulations. **By 2100, about 52% of SEBM dune habitat (Sebastian) and 99% of inland habitat (Pelican) is estimated to be lost, for a total reduction of ~74%** (Figure 79). There is little projected loss of dune habitat from 2020 to 2080 but then a rapid decline due to SLR (orange line), while SLR impacts begin earlier for inland habitat (blue line). This projection does not include habitat loss due to major storms.

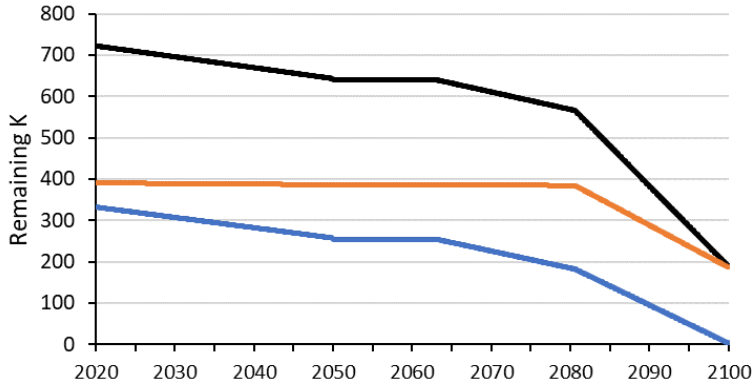


Fig. 79. Estimated K for SEBM habitat in SIPI over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110.

Impacts of storms

SIPI is estimated to have the same risk of experiencing major storms as other SEBM subpopulations. There is a 4% annual risk of hurricanes (occurring in summer and fall) and 4% risk of Nor'Easters (occurring in March-April). Estimated storm impacts differ between the two areas. Sebastian dune habitat is estimated to have similar impacts as CNS, with 30% mortality and no reproduction for two timesteps, and with habitat loss down to 40% of pre-storm conditions. Habitat recovery is estimated to be similar to that in SDP, with recovery to 70% of pre-storm conditions in 5 years (constrained by any SLR loss). Inland habitat (Pelican) is estimated to be similar to the Inland subpopulation, with 10% mortality (both storm types) and 50% reduction in reproduction for one timestep (hurricanes only), with no habitat loss. The model does not allow the habitat to recover above that available due to SLR & HTF.

Model Results

Reintroduction scenarios were developed that involved multiple releases of 50 juvenile mice. This release number was found to be effective in modeling of the SDP subpopulation and was also suggested by the PVA team as logistically feasible. The first set of scenarios involve releases of 50 mice every October for five consecutive years in each of the two areas (total of 500 mice released over 5 years). The first translocations are modeled as occurring in October 2025, but these general results should be applicable to translocation projects that begin later than 2025, with one caveat. There is relatively little change in K due to SLR projected between 2020 and 2065; however, if any major storms occur before reintroduction begins, the associated loss of habitat (K) could limit population growth and reduce the retention of founder gene diversity in the subpopulation.

Figure 80 illustrates mean population size (and SD) for the five years of releases and subsequent 5 years with no additional releases (January 2025-December 2034). Projected mean population size peaks about 4 years after releases begin at ~84% K (determined by SLR) and then starts to slowly decline (blue line). If storms are removed from the model (red line), mean population size reaches ~93% of K. The base projection with storms is lower due to some iterations that experienced major storms, which reduce K by 60% in the model. This pattern is driven by the impact of storms on dune habitat (Sebastian), as storms do not impact K for the inland habit (Pelican) in the model. The actual realized population growth after reintroduction may be determined, in part, by whether a major storm occurs in the first few years of the reintroduction effort and its impact on SEBM carrying capacity in dune habitat.

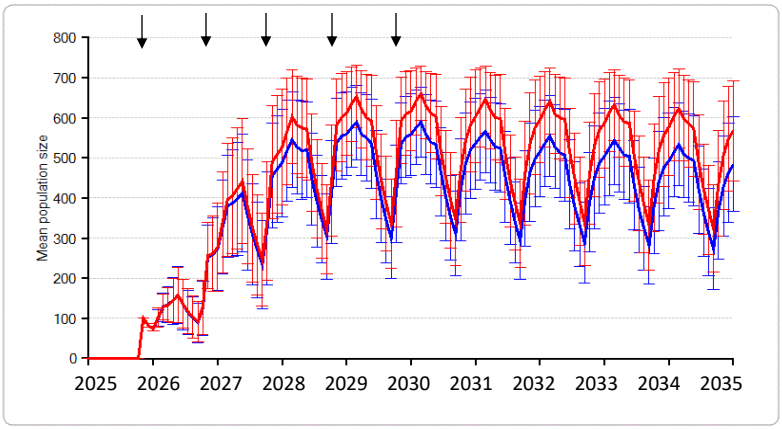


Fig. 80. Mean population size for the five release years and subsequent five years. Bars represent ± 1 SD. Arrows indicate timesteps for releases.

Long-term viability of this scenario (i.e., release of 50 mice for 5 consecutive years, with no further releases) to establish the SIPI reintroduced subpopulation is poor, with declining mean population size and gene diversity, and 95% extinction risk by 2100. Driving factors for viability in the PVA model are similar to the isolated, relatively small SDP and AFP subpopulations, with both loss of K and inbreeding depression as the primary contributors to poor long-term viability (Figures 81 and 82).

In the first few decades after reintroduction, reduction of K through habitat loss is the larger factor in population decline. Both sources of habitat loss contribute to metapopulation decline, with storms impacting dune habitat and SLR impacting the inland area of Pelican Island. If both sources of habitat loss are removed (i.e., K remains constant; orange line), the SIPI metapopulation still declines on average to ~26% of K by 2100 due to inbreeding depression. Mean inbreeding accumulates to ~25% (equivalent to full sibling relationship) after ~30 years of isolation following the last release and begins to contribute more heavily to population decline. SLR accelerates starting in 2080 and is projected to eliminate the inland Pelican subpopulation by 2100, leaving the small, isolated Sebastian dune subpopulation vulnerable to inbreeding depression as mean inbreeding exceeds 40%. Since the *VORTEX* model is not spatially explicit but assumes that each subpopulation is panmictic, this projection may be optimistic. It is possible that inbreeding depression will exert an impact sooner, especially in the Sebastian Inlet area distant from immigrants from Pelican island. In addition, potential fragmentation of dune habitat was not included in the model but also could lower viability of SIPI metapopulation. Given these results, both periodic genetic supplementation and dune habitat restoration are likely be valuable and potentially necessary strategies for maintaining a viable SEBM subpopulation in SIPI. This is not unexpected, given that SEBM have disappeared from these areas in the absence of such efforts.

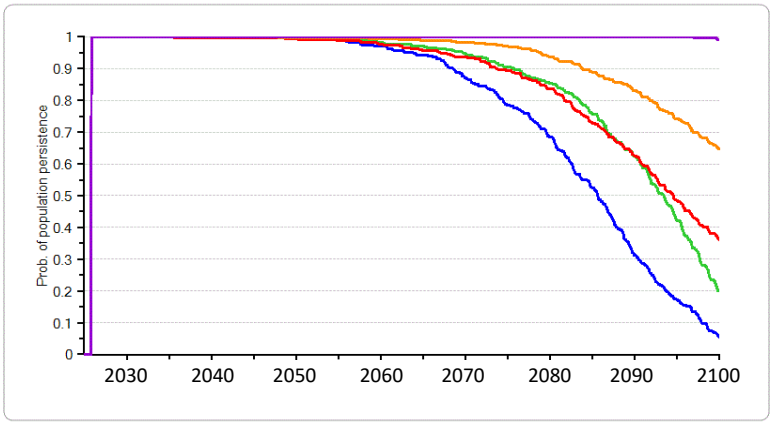


Fig. 81. Probability of population persistence over 75 years for SIPI reintroduced population, with 5 annual releases and no subsequent augmentation, with exclusion of different threats (habitat loss (HL), SLR, storm K loss, inbreeding impacts).

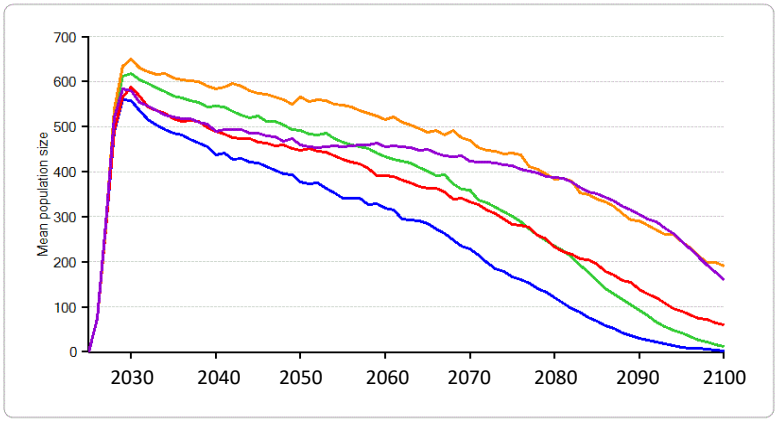


Fig. 82. Mean population size in December over 75 years for SIPI reintroduced population, with 5 annual releases and no subsequent augmentation, with exclusion of different threats (habitat loss (HL), SLR, storm K loss, inbreeding impacts).

Connectivity

The viability of a reintroduced SEBM metapopulation in SIPI is impacted by the movement of mice across the available habitat, including between dune and inland habitat. Connectivity is especially important for the Sebastian subpopulation due to the loss of K due to storms. A model scenario was tested with no connectivity between the two subpopulations and found that the median time to extinction after the last release in 2029 decreases from 55 years to 40 years. Loss of gene diversity through genetic drift is faster if the two subpopulations are isolated, and inbreeding accumulation (and therefore inbreeding depression) accelerates. Figure 83 shows the projected mean population size for the SIPI metapopulation with (blue line) and without (red line) connectivity. More rapid population

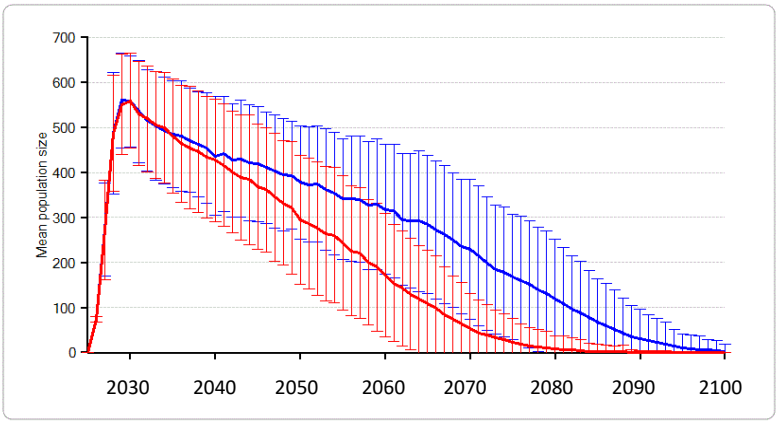


Fig. 83. Mean population size over 75 years for SIPI reintroduced population, with 5 annual releases and no further augmentation, with and without connectivity. Bars represent ± 1 SD.

decline in the absence of connectivity is driven by inbreeding rather than by loss of K. Thus, genetic isolation is projected to be an important factor in SIPI metapopulation viability. **Management that promotes connectivity** across the habitat (along the coast as well as between dune and inland habitat on Pelican island) may be **important for viability**, whether it is achieved through natural movement, facilitated corridors, or translocation.

Genetic augmentation

Exploratory scenarios were tested with additional releases of 50 mice every 5 or 10 years, either in one or both of the two subpopulations. As projected for several other smaller SEBM subpopulations, **viability is improved with periodic genetic augmentation**, leading to significantly higher gene diversity, lower inbreeding, and <1% risk of extinction before 2096, with slow metapopulation decline until 2080, after which SLR accelerates and both subpopulations decline. Releases every 5 years vs 10 years leads to better viability projections, as observed in the SDP PVA. Long-term genetic augmentation is valuable especially for the Sebastian dune subpopulation, as the entire metapopulation becomes more dependent upon the dune subpopulation as SLR floods the inland habitat. Releasing 50 mice every 5 years in Sebastian (purple line) is almost as beneficial as releasing 50 mice in both subpopulations (100 total; green line) every 5 years in maintaining gene diversity (Figure 84) and limiting mean inbreeding (Figure 85). While these model projections are only guidelines, they suggest that both frequency and location of releases can be important for the long-term viability of a reintroduced SIPI meta-population. Frequency and location may become increasingly important over time as habitats (and K) change.

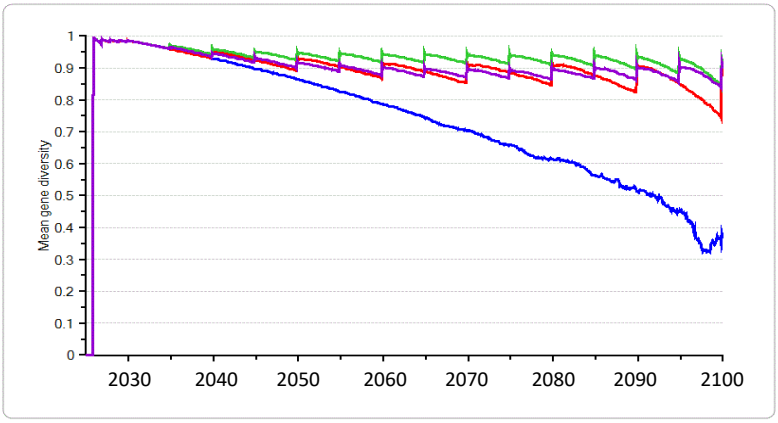


Fig. 84. Mean gene diversity over 75 years for SIPI reintroduced population, with 5 annual releases, with and without augmentation. S50=release of 50 mice; Y=years; both=releases in both subpopulations; Seb=releases in Sebastian only.

No augmentation
S50 every 10Y (both)
S50 every 5Y (both)
S50 every 5Y (Seb only)

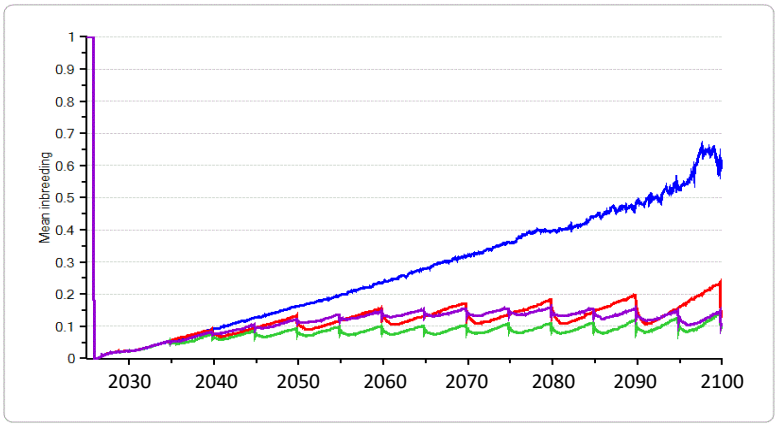


Fig. 85. Mean inbreeding over 75 years for SIPI reintroduced population, with 5 annual releases, with and without augmentation. S50=release of 50 mice; Y=years; both=releases in both subpopulations; Seb=releases in Sebastian only.

No augmentation
S50 every 10Y (both)
S50 every 5Y (both)
S50 every 5Y (Seb only)

Augmentation with habitat restoration

In the model, this potential reintroduction area consists of two areas, each assuming to be able to support relatively equal numbers of mice and having similar demographic rates. Model assumptions are that dune habitat (Sebastian) is vulnerable to loss of K due to storms but experiences little SLR impact until 2080, while inland habitat (Pelican) deteriorates due to SLR but suffers no storm impacts on K. Both areas decrease after 2080, when SLR accelerates and floods inland habitat. Exploratory scenarios were tested in which habitat restoration efforts following storms returned K to pre-storm values in 5 years in dune habitat. Figure 86 shows how post-storm habitat restoration efforts (in combination with periodic releases every 5 years) result in almost 50% increase in mean population in Sebastian from 2030 to 2080 (black line) compared to releases alone (red line). Mean gene diversity and inbreeding in Sebastian are similar with or without habitat restoration, and extinction risk is <1% in both scenarios. The inland subpopulation (not shown) shows modest benefit in increased mean population size as well due to inter-population dispersal from a larger dune subpopulation. The mixed-strategy of releases every 5 years plus habitat restoration quickly to pre-storm K results in a projected viable metapopulation on SIPI to 2080 (Figure 87), with no extinction risk, gene diversity > 90%, and inbreeding < 10%. The projected rapid loss of inland habitat due to SLR between 2080 and 2100, in concert with accelerated SLR along the coast, results in rapid decline in projected SEBM numbers that jeopardize beach mouse populations in this habitat if SLR continues beyond projected levels for 2100.

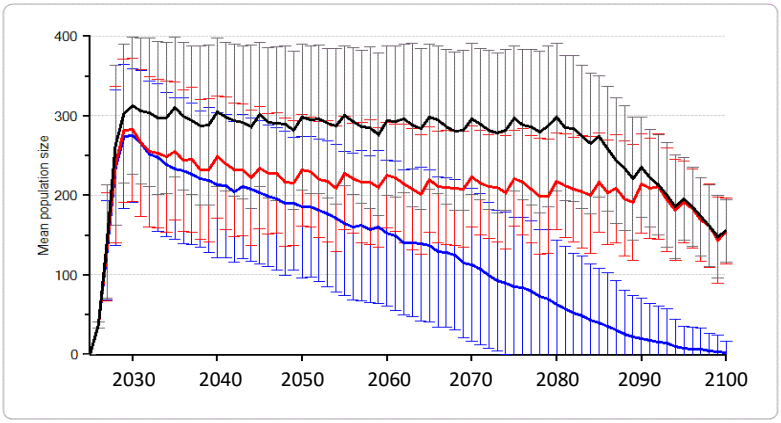


Fig. 86. Mean population size in December over 75 years for dune habitat (Sebastian) reintroduced population, with and without subsequent augmentation and habitat restoration. Bars represent ± 1 SD.

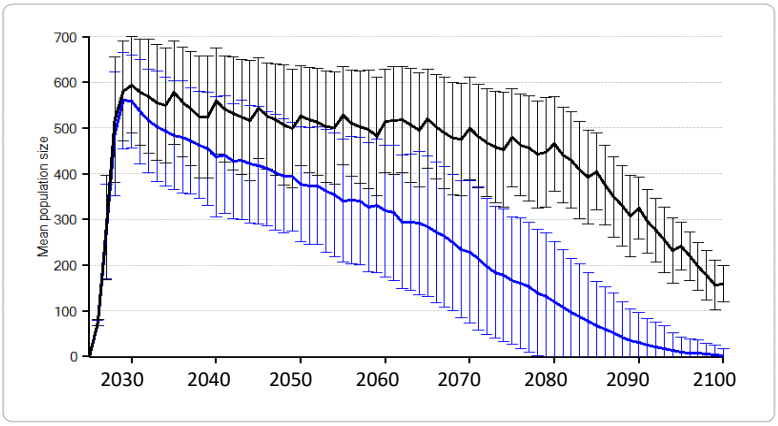


Fig. 87. Mean population size in December over 75 years for the reintroduced SIPI metapopulation, with and without subsequent augmentation and habitat restoration. Bars represent ± 1 SD.

Implications for reintroduction on Orchid Island

Orchid Island represents former SEBM habitat that may be suitable for the establishment of a SEBM subpopulation, at least in the short-term, but may require periodic genetic augmentation and habitat restoration. Several significant areas of uncertainty affect short- and long-term viability projections for a reintroduced population in this habitat.

The first area of uncertainty surrounds the current habitat quality for beach mice and any existing threats, including those that may have contributed to previous SEBM extirpation, such as disease, inadequate food or nesting resources, competing species, natural or invasive predators, or human disturbance. Prior land use on Pelican Island may have impacts on habitat quality for beach mice. Any reintroduction program should follow the IUCN SSC *Guidelines for Reintroductions and Other Conservation Translocations* (IUCN/SSC, 2013), which recommends efforts to confirm that the threat(s) that caused extirpation have been identified and removed or sufficiently reduced.

Poor post-release survival or reproduction may require larger or more frequent releases to establish the subpopulation than modeled here. Population growth in the initial years of reintroduction are important in capturing gene diversity and reducing inbreeding, suggesting value in short-term management efforts to support mice and habitat quality. Monitoring of beach mice to understand population status and support effective responsive management will be a valuable and recommended component of any reintroduction efforts.

Connectivity and movement of beach mice throughout the reintroduction area and across habitats is likely vital to the viability of a reintroduced population. PVA model assumptions include a fair amount of uncertainty and may be optimistic, as the model does not include fragmentation and assumes panmictic breeding subpopulations. The location of initial releases should be considered to facilitate occupation of dune habitat in both the north and south, as feasible. **Management that promotes connectivity across the habitat (along the coast as well as between dune and inland habitat on Pelican island) may be important for viability**, whether it is achieved through natural movement, facilitated corridors, or translocation. Limited connectivity may necessitate more frequent genetic augmentation.

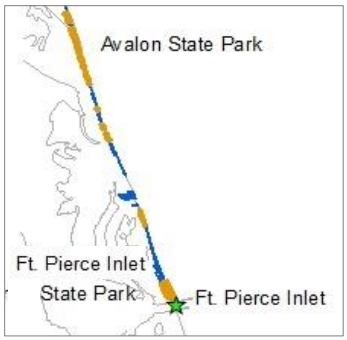
Model assumptions regarding the loss of SEBM habitat due to SLR and to storms drives the long-term viability of this SEBM reintroduced population. While based on the best currently available information, there is uncertainty surrounding these projections**. If SLR accelerates as projected, this limits long-term viability for SEBM on Orchid Island.** Storm impacts on the habitat, and post-storm habitat recovery, are also uncertain. Any factors that affect the carrying capacity of these areas for SEBM ultimately may drive the potential for a viable subpopulation.

Periodic releases of SEBM from Beach2 (or other sources) have the potential to counteract genetic drift and inbreeding. Modeling suggests that the degree of translocation effort required to establish and maintain a viable subpopulation is feasible, provided sufficient habitat is available and there is good connectivity. The size, number and schedule of translocations is impacted by uncertainty both related to SEBM and to the habitat. Habitat management efforts to improve or restore habitat quality for SEBM will promote larger SEBM numbers and minimize the need for additional augmentation.

Avalon State Park to Fort Pierce Inlet State Park (AFP)

This area of North Hutchinson Island is anchored by Avalon State Park to the north and Fort Pierce Inlet State Park to the south (shaded in yellow) connected by private lands (shaded in blue), as depicted in Figure 88 taken from the 2020 SSA (USFWS 2020). This historical SEBM habitat consists primarily of narrow coastal dunes (81ha) with an additional 4ha of inland habitat. Few if any SEBM are believed to inhabit this area, which has no connectivity with other SEBM subpopulations. The PVA team estimated the average carrying capacity of this habitat to be similar to that for the CNS subpopulation (i.e., 5 mice/ha). Estimated potential density results in an estimated initial K of 425 mice based on 85 ha. These scenarios assume that there are no SEBM in this area prior to the first translocation.

Fig. 88. North Hutchinson Island habitat.



Inbreeding impacts

Inbreeding depression was set at the same level as the Beach2 source subpopulation (4.72 LE). Computational constraints necessitated applying inbreeding depression in relatively large SEBM subpopulations (Beach2, Inland, HC) only via recessive lethal alleles (100%). Since the maximum population size (K) is relatively small for AFP, inbreeding impacts can be applied using 50% due to recessive lethal alleles and 50% due to reduced survival of inbred juveniles (as for CNS and SIPI).

Impact of sea level rise (SLR)

Expected future loss of SEBM habitat in AFP due to sea level rise and high tide flooding was estimated based on 2022 projections using High SLR and Moderate HTF scenarios for Trident Pier and applying the same methodology as was used for the other SEBM subpopulations. **By 2100, about 52% of SEBM habitat in AFP is estimated to be lost** (Figure 89). There is little projected loss from 2020 to 2080 but then a rapid decline in habitat due to SLR. This projection does not include habitat loss due to storms.

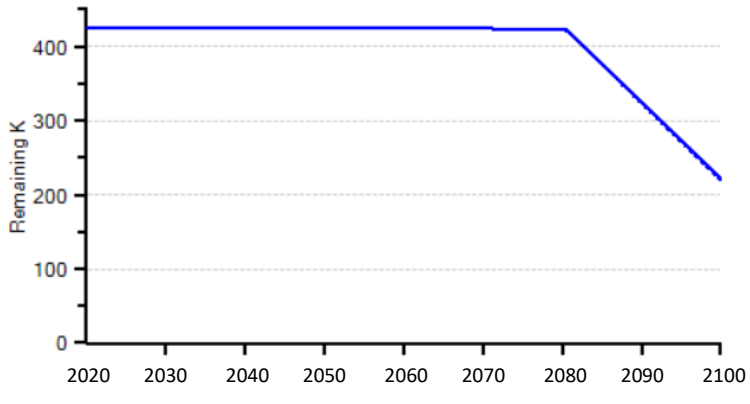


Fig. 89. Estimated K for SEBM habitat in AFP over 80 years based on HIGH SLR & Moderate HTF scenario projections by NOAA (2022) for years 2050, 2080, 2110.

Impacts of storms

AFP is estimated to have the same risk of experiencing major storms as other SEBM subpopulations. There is a 4% annual risk of hurricanes (occurring in summer and fall) and 4% risk of Nor'Easters (occurring in March-April), with the following demographic impacts if a storm occurs: 1) 30% mortality of mice during the initial timestep; and 2) no successful reproduction during the initial or next timestep. These are the same demographic impacts as used for the CNS PVA (similar narrow coastal habitat). In

addition, impacts on SEBM habitat were applied based on inputs used for SDP PVA: 1) loss of habitat (K) for SEBM to 40% of pre-storm conditions; and 2) recovery of the habitat (K for SEBM) to 70% of pre-storm conditions in 5 years, with no further recovery. The model does not allow the habitat (K) to recover above that available due to SLR and HTF.

Model Results

Reintroduction scenarios were developed that involved multiple releases of 50 juvenile mice. This release number was found to be effective in modeling of the SDP subpopulation and was also suggested by the PVA team as logistically feasible. The first translocations are modeled as occurring in October 2025, but these general results should be applicable to translocation projects that begin later than 2025, with one caveat. There is essentially no change in K due to SLR projected between 2020 and 2080; however, if any major storms occur before reintroduction begins, the associated loss of habitat (K) could limit population growth and reduce the retention of founder gene diversity in the subpopulation.

The first set of scenarios involve releases of 50 mice every October for five consecutive years (total of 250 mice released). Figure 90 illustrates mean population size (and SD) for the five years of releases and subsequent 5 years with no additional releases (January 2025-December 2034). Projected mean population size peaks about 4 years after releases begin at ~80% K (determined by SLR) and then starts to slowly decline (blue line). If storms are removed from the model (red line), mean population size reaches ~94% of K. The base projection with storms is lower due to some iterations that experienced major storms, which reduce K by 60% in the model. The actual realized population growth after reintroduction may be determined, in part, on whether a major storm occurs in the first few years of the reintroduction effort and its impact on SEBM carrying capacity.

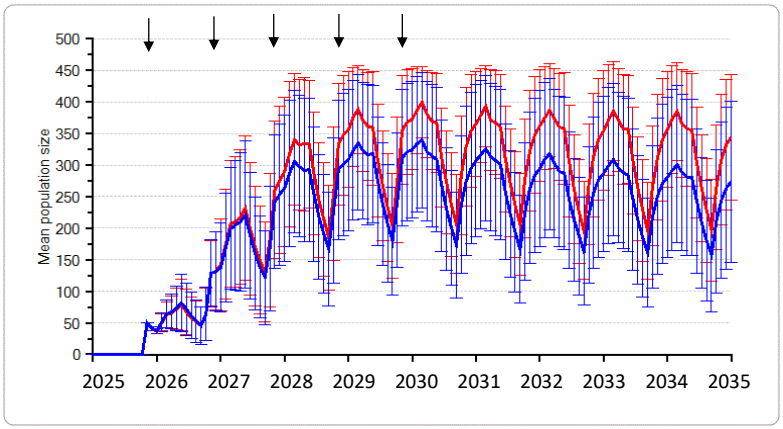


Fig. 90. Mean population size for the five release years and subsequent five years. Bars represent ± 1 SD. Arrows indicate timesteps for releases.

Long-term viability of this scenario (i.e., release of 50 mice for 5 consecutive years, with no further releases) to establish the AFP reintroduced subpopulation is poor, with declining mean population size and gene diversity, and eventual extinction. Driving factors for viability in the PVA model are similar to the isolated SDP subpopulation of similar size, with both loss of K due to storms and inbreeding depression as the primary contributors (Figure 91; also see SDP PVA for additional analysis). While there is significant loss of K due to SLR after 2080, given model assumptions this is overshadowed by loss of K due to storms, in the absence of additional habitat restoration. This situation mirrors that emerging from the SDP PVA. Both habitat restoration and genetic supplementation are likely be valuable and potentially necessary strategies for maintaining a viable SEBM subpopulation in AFP.

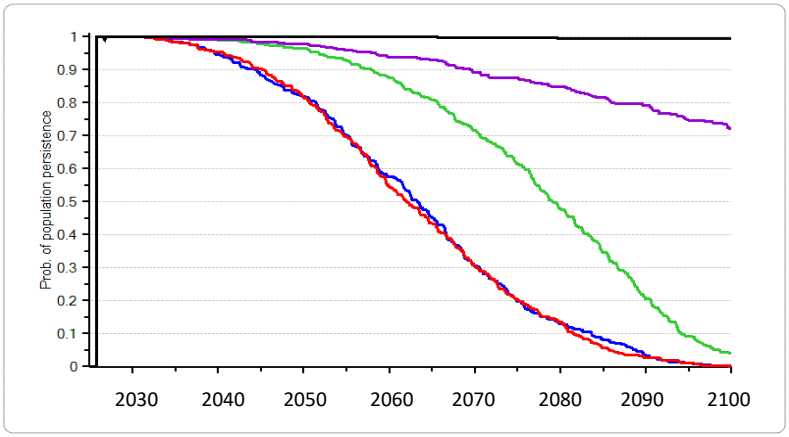


Fig. 91. Probability of population persistence over 75 years for AFP reintroduced population, with 5 annual releases and no subsequent augmentation, with exclusion of different threats (SLR, storm K loss, inbreeding depression).

Inbreeding depression

Exploration of inbreeding impacts for AFP follow those for SDP, in that higher impacts (LE) lead to lower mean population size and higher extinction risk. If purging of recessive lethal alleles has already occurred in the Beach2 source population, this may lessen inbreeding impacts for AFP, but given the relatively large number of SEBM on the Cape, past purging is uncertain. These results suggest that, like the isolated SDP subpopulation, establishment of a viable AFP subpopulation may require additional periodic releases for genetic augmentation.

Genetic augmentation and habitat restoration

Exploratory scenarios were tested with additional releases of 50 mice every 5 or 10 years, similar to scenarios explored for SDP. These augmentation schedules were tested both with and without habitat restoration efforts following storms, with SEBM K restored to 100% of pre-storm values within 5 years.

Periodic augmentation alone improves viability, but is still associated with population decline and risk of extirpation between releases (Figure 92; red and green lines). Mean population size is about the same for both augmentation schedules, maintaining on average fewer than 100 mice by 2100 (Figure 93), due to the projected loss of K due to major storms and contributing to extinction risk between releases. Augmentation improves genetic status, with releases every 5 years performing better (green line) and projected to retain good gene diversity (Figure 94) and keep inbreeding below 20% (Figure 95).

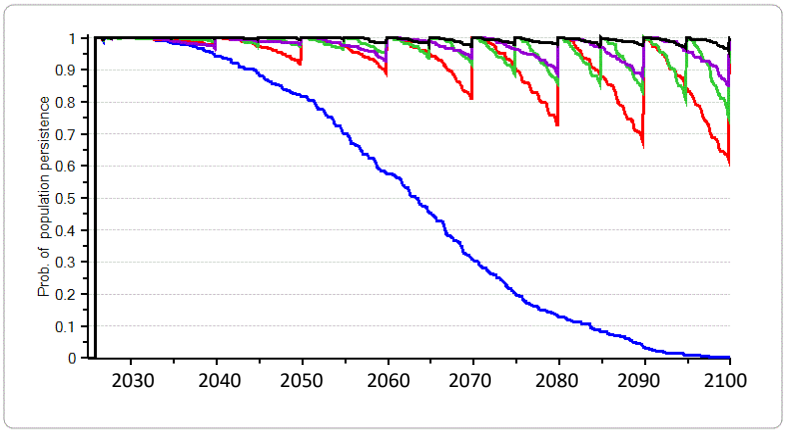


Fig. 92. Probability of population persistence over 75 years for AFP reintroduced population, with and without subsequent augmentation and habitat restoration.

When post-storm habitat restoration is added, the resulting **two-prolonged management strategy supports a larger SEBM mean population size** (Figure 93; purple and black lines), **which in turn slows genetic drift and inbreeding and reduces extinction risk**. The release of 50 mice every 5 years, in combination with habitation restoration, keeps the risk of extinction before the next release to <4% by 2100, maintains 82-90% GD, and limits inbreeding to ≤15%. These management strategies will become more important as population size declines after SLR begins to accelerate (~2080).

Given the uncertainties in model inputs, these scenarios do not provide precise projections but can serve as a tool to better understand how **continued management may be required to establish and maintain a viable AFP SEBM subpopulation** in this relatively small and vulnerable habitat.

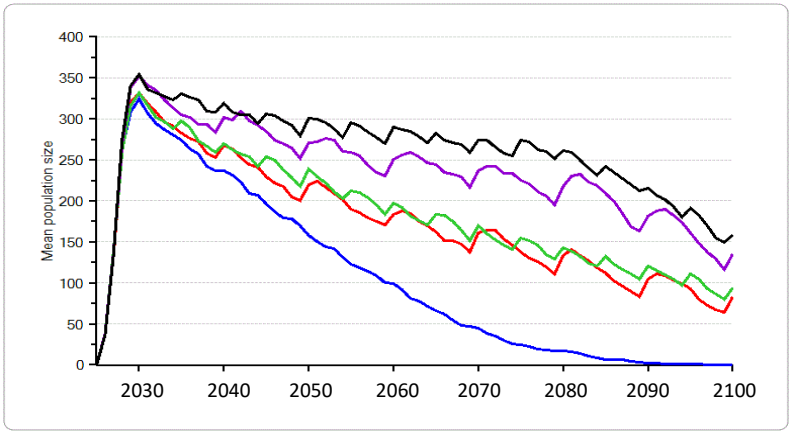


Fig. 93. Mean population size in December over 75 years for AFP reintroduced population, with and without subsequent augmentation and habitat restoration.

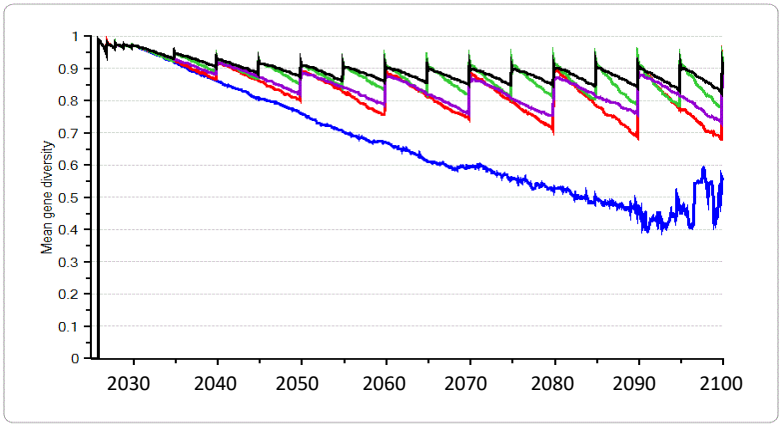


Fig. 94. Mean gene diversity over 75 years for AFP reintroduced population, with and without subsequent augmentation and habitat restoration.

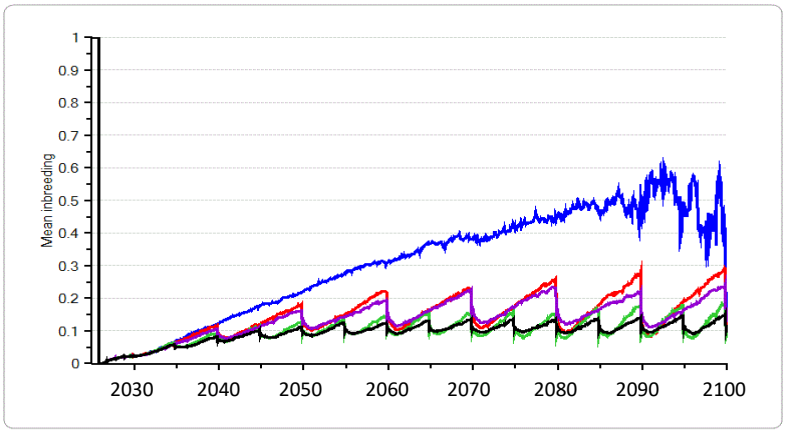


Fig. 95. Mean inbreeding over 75 years for AFP reintroduced population, with and without subsequent augmentation and habitat restoration.

Implications for reintroduction on North Hutchinson Island

North Hutchinson Island represents former SEBM habitat that may be suitable for the establishment of a small SEBM subpopulation that likely will require periodic genetic augmentation and habitat restoration. Uncertainty in several model inputs can affect projected SEBM viability in this habitat.

Poor post-release survival or reproduction may require larger or more frequent releases to establish the subpopulation than modeled here. Population growth in the initial years of reintroduction are important in capturing gene diversity and reducing inbreeding, suggesting value in short-term management efforts to support mice and habitat quality. Monitoring of beach mice to understand population status and support effective responsive management will be a valuable and recommended component of any reintroduction efforts.

Model projections assume the reintroduced population to be panmictic throughout the available habitat and that there is no fragmentation of this coastal habitat; thus, PVA projections may be optimistic. Connectivity and movement of beach mice throughout the reintroduction area is likely vital to the viability of a reintroduced population. The location of releases should be considered to facilitate occupation of dune habitat along the length of the coast, as feasible. **Management that promotes connectivity across the habitat may be important for viability,** whether it is achieved through natural movement, facilitated corridors, or translocation. Limited connectivity may necessitate more frequent genetic augmentation.

While based on the best currently available information, there is uncertainty surrounding model assumptions regarding the loss of SEBM habitat due to storms and SLR. Any factors that affect the carrying capacity of this area for SEBM may drive the potential for a viable subpopulation.

Periodic releases of SEBM from Beach2 (or other sources) have the potential to counteract genetic drift and inbreeding. Modeling suggests that the degree of translocation effort required to establish and maintain a viable subpopulation is feasible, provided sufficient habitat is available. The size, number and schedule of translocations is impacted by uncertainty both related to SEBM and to the habitat. **Habitat management efforts to improve or restore habitat quality for SEBM** will promote larger SEBM numbers and minimize the need for additional augmentation.

Finally, it is **important to ascertain if there is some threat or factor present that is not accounted for in this PVA**, such as disease, inadequate food or nesting resources, competing species, natural or invasive predators, or human disturbance. Any reintroduction program should follow the IUCN SSC *Guidelines for Reintroductions and Other Conservation Translocations* (IUCN/SSC, 2013), which recommends efforts to confirm that the threat(s) that caused extirpation have been identified and removed or sufficiently reduced.

Summary of Reintroduction Scenarios

Given the potential strong growth rate of beach mice in an environment with ample resources and free from threats, PVA modeling supports the feasibility of re-establishing extirpated SEBM populations in suitable, secure habitat. Three primary factors influence the viability of reintroduced SEBM populations:

Habitat carrying capacity: Reintroductions are likely to be most successful in large, stable areas that can support large SEBM populations. Any factors that reduce habitat (or the ability of the habitat to support SEBM) can reduce population viability, especially for smaller areas, and may require more management or intervention. PVA models focused on sea level rise and major storms, but other factors that reduce K for SEBM or cause habitat fragmentation could be relevant.

Genetic isolation: Genetic diversity is important for adaptive potential to new challenges and is negatively correlated with inbreeding, which can lower individual and therefore population viability. Most reintroduced SEBM populations may be genetically isolated and, if small enough to experience genetic drift and inbreeding depression, may benefit from or potentially require genetic augmentation. Alternatively, restoring habitat connectivity (if feasible) to other SEBM subpopulations may provide genetic rescue.

Additional threats: The prior loss of SEBM subpopulations from these potential reintroduction sites may have been due to limited habitat and carrying capacity, habitat fragmentation, and loss of genetic diversity and damaging inbreeding, as described above. However, there may be other factors or threats that were not included in this PVA. This could include threats that directly impact SEBM survival or reproduction, such as disease, competitors, invasive species, or intense predation. The 2005 PVA for the Alabama beach mouse concluded that predation by a small number of domestic cats has the potential to devastate small beach mouse populations, particularly when mouse numbers are low following a major storm (Traylor-Holzer *et al*., 2005). It will be important to investigate why SEBM were extirpated from an area so that any existing threats can be addressed before reintroduction begins.

Different potential reintroduction sites may pose different challenges. In general, the smaller the site (and SEBM K), the more intensive management efforts may need to be to maintain a viable SEBM population. Given the model assumptions, priority for initial reintroduction efforts might be focused on Happy Creek, given its large size and projected carrying capacity for SEBM, as well as the potential option to connect this area to existing SEBM subpopulations. This might provide the opportunity to refine trapping and release protocols to improve translocation and post-release success before moving to smaller sites. However, there are other factors not included in this PVA, including biological issues, political concerns, and public support, that may impact reintroduction priorities among potential sites. Regardless of reintroduction site, monitoring SEBM reintroduced populations and habitat will help to address uncertainties and provide a basis for adapting management efforts to meet project needs. Any reintroduction efforts should review the guidance provided by the IUCN reintroduction guidelines (IUCN/SSC 2013).

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Appendix A. SEBM PVA Team members

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Appendix B. Model inputs

Table B1. Inputs for base demographic model. F=# adult females; M=# adult males; PARITY=# of previous offspring; PARITY>0 indicates multiparous female. Note that the timestep (ts) used in the model is 28 days (e.g., 3 ts = 84 days). All rates are adjusted to a 28-day ts.

		Winter	Spring	Summer	Fall
Parameter	***VORTEX* input**	**Mean (EV)**	**Mean (EV)**	**Mean (EV)**	**Mean (EV)**
Mating system	Long-term polygyny				
Maximum # of female mates	2; entered as this function: =1+(F/M>1.22)				
Age of first reproduction	3 ts				
Maximum age of reproduction	26 ts				
Maximum age	26 ts				
% males in breeding pool	100				
Maximum # litters per timestep	1				
Mean # offspring per litter	Entered as this function: =POISSON(((PARITY>0)*0.77)+3.56)				
Sex ratio at birth	50:50				
Seasonal vital rates					
% adult females breeding		58.7 (21.9)	35.4 (25.3)	38.6 (19.6)	83.7 (3.6)
% mortality: Age 0 to 1 ts **inputs after adjusting for dam deaths*		57.8 **53.1 (13.7)*	45.6 **36.0 (11.8)*	80.2 **73.8 (9.5)*	51.8 **42.7 (0.8)*
% mortality: Age 1 ts to 2 ts		10.9 (5.6)	16.5 (10.8)	27.0 (6.9)	17.5 (0.8)
% mortality per ts (Age 2+)		9.9 (5.1)	15.0 (9.8)	24.5 (6.3)	15.9 (0.7)

Table B2. Inputs for Status Quo Scenarios, by subpopulation.

Parameter	Smyrna Dunes	Beach1 (CNS)	Beach2 (Cape)	Inland
Habitat type and area	81.34ha (63ha of dune habitat plus 18ha spoils)	96.4ha of dune habitat; eroding	1238.3ha of dune habitat; not homogenous	3786.4ha of inland habitat; possible variable quality
Density estimate for K	5.86 mice/ha across 81.34 ha	Test 5mice/ha and 7mice/ha	9 mice/ha	4 mice/ha
Carrying capacity (K) (Total K = 27,250-27,443)	477	482 & 675	11145 (before impact zones)	15146 (before impact zones)
Initial pop size (N) on Jan 1 (Initial N = 23,162-23,326) (before impact zones)	405 Set at 85% of K w/SAD	410 & 574 Set at 85% of K w/SAD	9473 (before impact zones) Set at 85% of K w/SAD	12874 (before impact zones) Set at 85% of K w/SAD
Connectivity	Isolated	Connected to Beach2 until 2080	Connected to CNS until 2080 Connected to Inland	Connected to Beach2
Dispersal between subpopulations	n/a	Both sexes; Age1 (first month of burrow); same across all boundaries; 100% survival		
Demography	Base model demographic inputs			
Truncation to K	Juveniles only			
Genetics	2.36 LE; 0% lethal Initial kinships = 0.087	4.72 LE; 50% lethal Initial kinships = 0	4.72 LE; 100% lethal Initial kinships = 0	4.72 LE; 100% lethal Initial kinships = 0
Hurricane risk	4% annual risk; occurs mid-June thru Nov (6 ts), w/higher risk in Sept (1%, vs 0.6% for other 5 ts)			
Hurricane impacts on SEBM	20% mortality; no reproduction for 2 ts	30% mortality; no reproduction for 2 ts	10% mortality; no reproduction for 1 ts	10% mortality; 50% reproduction for 1 ts
Hurricane impacts on K and recovery	Loss to 40% K (recovers to 70% pre-storm K in 5 yrs, and then stable)	Loss to 40% K (recovers to 50% in pre-storm K in 5 yrs and then stable); 50% risk of fragmenting into 2 areas	Some loss (erosion) and gain (accretion), with no net change; increased dispersal with Inland for 2 ts	No impact on SEBM habitat
Recent hurricane impacts	None included	K impacts of hurricane in Nov 2022 & October 2024), with same rate of recovery	None included	None included
Nor'Easter risk/impacts on SEBM & K	Same risk as hurricanes except occurs in March-April (2 ts); also same impacts on mortality and reproduction; also same impact on K but no fragmentation			No risk or impacts
SLR impacts	Estimated from 2022 projections using High SLR and Moderate High HTF scenarios			
K loss by 2100 due to SLR	31% loss (K_{2100}=329)	37% loss (K_{2100}=303 or 424)	55% loss (K_{2100}=5015)	10% loss (K_{2100}=13,631)
Development impacts	None included	None included	Impact zones around pads (¼ mile and ½ mile) with 50% density and with no mice; subtracted from K	
K loss due to impact zones	None	None	0.7%-9.2%	2.4%-13%