

Species Status Assessment Report for the Southern Sea Otter (*Enhydra lutris nereis*)

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EXECUTIVE SUMMARY

We, the U.S. Fish and Wildlife Service (Service) concluded in our last 5-year review of the southern sea otter (*Enhydra lutris nereis*) (Service 2015) that a change in status under the Endangered Species Act of 1973, as amended (Act) was not warranted for the following reasons: the southern sea otter remained restricted to a small fraction of its historic range; it remained possible that a large oil spill could affect a large proportion of the population; high levels of shark-bite mortality were affecting significant portions of the mainland range and preventing range expansion; food limitation, disease/biotoxin intoxication, mortality in fishing gear, and recreation-related harassment had emerged since listing as factors affecting or potentially affecting the population; limited genetic diversity indicated the subspecies may not have the capacity to adapt to novel pathogens or new risks associated with climate change; and the population index remained below the threshold for delisting consideration (3,090) as identified in the recovery plan (Service 2003). In 2018, the southern sea otter population index exceeded 3,090 for the third consecutive year, meeting the threshold for delisting consideration, although the following year it declined to 2,962 (Hatfield et al. 2018, 2019). Also in 2018, a study found that assumptions made in the recovery plan regarding the relationship between effective population size and actual population size, which serve as the basis for the criteria, are not accurate (Gagne et al. 2018). Gagne et al. (2018) recommended an alternate approach to evaluating the status of the species, such as conducting population viability analyses that can incorporate genetic and demographic factors to determine extinction risks. We announced our initiation of this status review in 2019 (84 FR 36116; July 26, 2019).

In the current analysis, we assessed the southern sea otter's needs (ecology), status under current conditions, and status under future conditions in terms of resiliency, redundancy, and representation (Service 2016; Smith et al. 2018, p. 306). Resiliency describes the ability of a population to withstand stochastic disturbance; it is positively related to population size and growth rate and may be influenced by connectivity among populations. Redundancy spreads risk among multiple populations or areas to minimize the damage due to large-scale, high-impact (i.e., catastrophic) events. Representation describes the ability of a species to adapt to changing environmental conditions over time and is characterized by the breadth of genetic and environmental diversity within and among populations.

Because sea otter populations are structured on relatively small spatial scales (on the order of tens of km) due to the high site fidelity and limited movements of individuals, particularly adult females (Gorbics and Bodkin 2001; Tinker et al. 2006; 2008b; 2019a; 2019b, pp. 63–85; Tarjan and Tinker 2016; Breed et al. 2017), we followed Tinker et al. (2021a) in dividing the historical and current range into coastal areas representing semi-distinct subpopulations. Although we

recognize the precise number and geographic boundaries of historical subpopulations can never be known with any certainty, we use the term *subpopulation* throughout this document as shorthand for “the sea otters that exist along or could be supported by the habitat in a coastal area.” We grouped these subpopulations into four ecoregions (Mendocinian, Montereyan, Southern Californian, and Magdalenian) based on biogeographic provinces defined by Fenberg et al. (2015).

We used historical information and an integrated population model (IPM) that incorporates all available data on southern sea otter trends, vital rates, and mortality risk factors (Tinker et al. 2021a) to examine the past and current influences of environmental and mortality risk factors on the resiliency of subpopulations and on the redundancy and representation at the species level. The IPM has a web tool that allows users to modify risk factors, either directly by adjusting hazard rates, or indirectly by adjusting environmental conditions. In response to peer reviewer comments on the draft SSA, we commissioned additional modeling work to address oil spill risk in the IPM (Appendix). The updated web tool allows users to add one or more oil spills to the simulations. For a baseline scenario, we used the IPM to project current conditions 50 years into the future from 2021. We based three plausible future scenarios on Representative Concentration Pathways to project differing rates of climate change, RCP 4.5 (low-intermediate rate), RCP 8.5 (high rate), and RCP 8.5 with a large oil spill, respectively.

The results of population projections based on current conditions and the three plausible future scenarios indicate that meaningful improvements in southern sea otter resiliency, redundancy, and representation are unlikely to occur on their own. Under current conditions and future scenarios, resiliency was expected to remain compromised by high levels of shark bite mortality on the outer coast, which maintained low sea otter abundance relative to estimated carrying capacity in subpopulations in the northern and southern portions of the current southern sea otter range along the central California coast and slowed or prevented range expansion to the north and south of these areas. Resiliency was further reduced by changes related to climate under the three plausible future scenarios, which intensified some known hazards, including mortality from shark bites, resulting in smaller mean subpopulation sizes and detrimentally affecting the ability of southern sea otter subpopulations to withstand and bounce back from stochastic events. Although there were a number of resilient subpopulations after 50 years, they were not sufficiently populated or distributed throughout the southern sea otter’s ecological settings or geographical range to reduce the risk of long-term impacts from ongoing threats or one or more catastrophic events. Adaptive capacity under current conditions and future scenarios remained compromised because genetic diversity and environmental diversity both remained limited. The southern sea otter’s diminished evolutionary potential poses an extinction risk because species with low evolutionary potential are more likely to produce maladaptive phenotypes, and thus to have reduced fitness, when confronted with a changing environment and suite of stressors.

We review the recovery criteria established in the sea otter recovery plan (Service 2003). Gagne et al. (2018) found that assumptions made in the recovery plan regarding the relationship between effective population size and an actual population size, which serve as the basis for the criteria, are not accurate. Therefore, we conclude that the 2003 criteria for endangered, threatened, and delisted status are not appropriate metrics against which to gauge the recovery of the southern sea otter.

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1.0 INTRODUCTION

1.1 Listing History

Original listing

FR notice: 42 FR 2965

Date listed: January 14, 1977

Entity listed: southern sea otter (*Enhydra lutris nereis*)

Classification: threatened

1.2 Species Basics

The sea otter (*Enhydra lutris*) [L. 1758] is the only strictly marine extant species of the Order Carnivora. The second-largest member of the family Mustelidae, the sea otter is also the smallest marine mammal. Living in a cold marine environment but lacking the blubber of most other marine mammals, sea otters rely on their dense pelage and a metabolism 2.4–3 times higher than that of land mammals of equal size to maintain homeostasis (Costa and Kooyman 1982, p. 2765; Costa and Kooyman 1984, p. 199). As a result, sea otters consume an amount of food (mostly benthic invertebrates) equivalent to 22–25% of their body weight each day (Costa 1982; Morrison et al. 1974). The sea otter's diet and caloric requirements are the underlying driver of the species' keystone role in nearshore marine ecosystems. Exerting a strong limiting influence on prey populations, sea otters have large-scale community effects disproportionate to their abundance and biomass, initiating trophic cascades in kelp forest (Estes and Palmisano 1974) and estuarine seagrass (Hughes et al. 2013) ecosystems.

Sea otters once ranged along the rim of the North Pacific Ocean from the northern Japanese islands to mid-Baja California, Mexico, but they were brought to near-extinction by the maritime fur trade of the 18th and 19th centuries. Three subspecies are recognized based on morphology (Wilson et al. 1991): the Asian sea otter (*E. l. lutris*); the northern sea otter (*E. l. kenyoni*); and the California or southern sea otter (*E. l. nereis*) (Figure 1). Contemporary sea otters are derived from 11 remnant populations, each numbering an estimated 10–100 individuals at their low point (Lensink 1962; Kenyon 1969, pp. 134, 189; Bodkin et al. 1999, p. 1380; Figure 2). Sea otter populations in Southeast Alaska, British Columbia, and Washington are the result of 13 separate translocations of 708 individuals from Amchitka Island and Prince William Sound to Southeast Alaska, British Columbia, and Washington between 1965 and 1972 (Jameson et al. 1982, p. 100). An additional translocation during the 1980s occurred at San Nicolas Island, California (Rathbun et al. 2000).

The southern sea otter historically ranged from Oregon, USA (which is thought to have been a transition zone between the northern and southern subspecies), to the species' southern range terminus near Punta Abreojos, Baja California, Mexico (Riedman and Estes 1990, p. 73; Lyman 1988; Wilson et al. 1991, Valentine et al. 2008; Larson et al. 2012, 2021; Wellman et al. 2018, 2020; see section 4.0). Currently, the subspecies occurs only in portions of California, USA: along roughly 500 km (310 mi) of the mainland coastline from San Mateo County to Santa Barbara County and in the waters surrounding San Nicolas Island, Ventura County, although occasionally individuals are documented in other areas. In 2019, the population index (the 3-year

average of combined sea otter counts from the mainland range and San Nicolas Island) was 2,962 (Hatfield et al. 2019).



Figure 1. Southern sea otter in Morro Bay with sand star (*Astropecten armatus*). Photo Lilian Carswell/USFWS.

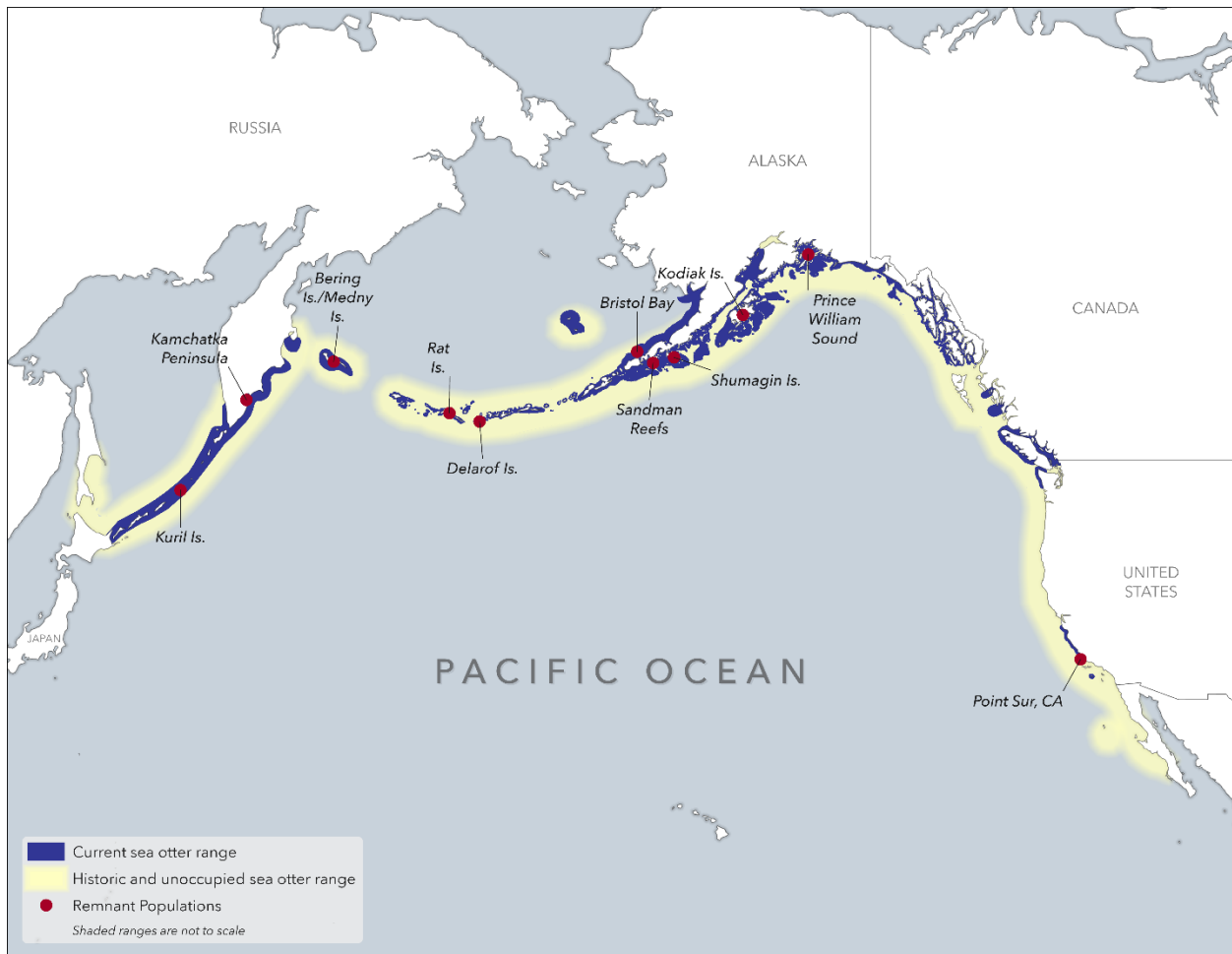


Figure 2. Historical global range of the sea otter (yellow) and currently occupied range (blue). Also shown are the 11 isolated remnant groups of sea otters that survived the maritime fur trade, likely numbering fewer than 1,000–2,000 animals total (red dots). All sea otters today (three recognized subspecies) are descended from these remnant populations.

1.3 Purpose

We, the U.S. Fish and Wildlife Service (Service), completed a 5-year review for the southern sea otter in 2015 (Service 2015) and found that a change in status under the Endangered Species Act of 1973, as amended (Act) was not warranted for the following reasons: the southern sea otter remained restricted to a small fraction of its historical range; it remained possible that a large oil spill could affect a large proportion of the population; high levels of shark-bite mortality were affecting sea otter abundance in substantial portions of the mainland range and preventing range expansion; food limitation, disease/biotoxin intoxication, mortality in fishing gear, and recreation-related harassment had emerged since listing as factors affecting or potentially affecting the population; limited genetic diversity indicated the subspecies may not have the capacity to adapt to novel pathogens or new risks associated with climate change; and the population index remained below the threshold for delisting consideration (3,090) as identified in the recovery plan (Service 2003). In 2018, the southern sea otter population index exceeded 3,090 for the third consecutive year, meeting the threshold for delisting consideration, although the following year it declined to 2,962 (Hatfield et al. 2018, 2019), after which surveys could not be completed due to the COVID-19 pandemic and plane availability. Also in 2018, a study found that assumptions made in the recovery plan regarding the relationship between effective population size and an actual population size, which serve as the basis for the criteria, are not accurate (Gagne et al. 2018). Gagne et al. (2018) recommended an alternate approach to evaluating the status of the species, such as conducting population viability analyses that can incorporate genetic and demographic factors to determine extinction risks. We announced our initiation of this status review in 2019 (84 FR 36116; July 26, 2019) and produced a draft species status assessment (SSA) in 2020. In response to peer reviewer comments on the draft SSA, we postponed revisions to the draft until the IPM was published (Tinker et al. 2021a) and commissioned additional modeling work to address oil spill risk in the IPM (Appendix).

On March 10, 2021, we received a November 2020 petition from the Pacific Legal Foundation, counsel for California Sea Urchin Commission and Commercial Fishermen of Santa Barbara, requesting that the southern sea otter be removed from the list of endangered and threatened wildlife (“delisted”) because the species does not meet the definition of an endangered or threatened species. We found that the petition presented substantial scientific or commercial information indicating that delisting the southern sea otter (*Enhydra lutris nereis*) may be warranted (87 FR 51635; August 23, 2022).

The SSA framework (Service 2016) is an analytical approach intended to provide an in-depth review of the species’ biology and threats, an evaluation of its biological status, and an assessment of the resources and conditions needed to maintain long-term viability. The SSA report is not a decision document; rather, it provides a review of available information related to the biological status of a species to aid decision makers, who must use the best available scientific information to make policy-guided decisions. This SSA summarizes our analysis of the best available scientific and commercial information¹ on the southern sea otters’ ability to sustain populations in the wild over time. We will use this SSA in concert with a consideration of

¹ The reference to “commercial data” is intended to allow for consideration of “trade data,” which refers to commercial harvest data on the abundance and catch levels of species that are, or were, traded. See H.R. Rep. 97-657 (H.R. Rep. No. 567, 97TH Cong., 2ND Sess. 1982, 1982 U.S.C.A.N. 2807, 1982 WL 25083) at 20.

all relevant laws, regulations, and policies to inform a potential reclassification decision for the southern sea otter.

2.0 METHODOLOGY

The SSA entails three iterative stages assessing the species’² needs (ecology), the species’ current condition, and the species’ future conditions (Service 2016; Smith et al. 2018).

2.1 Species Ecology

The SSA begins with a compilation of the best available biological information on the species’ life history and habitat and its ecological needs at the individual, population, and species levels.

- Individual level: A species’ life history, including its trophic niches, reproductive strategies, biological interactions, and habitat requirements, determines how individuals at each life stage respond to natural and anthropogenic influences. We developed a life history profile to describe what variables influence the successful completion of each life stage of the southern sea otter and to document characteristics that make the species sensitive to or resilient to particular natural or anthropogenic influences.
- Population level: *Resiliency* describes the ability of a population to withstand stochastic disturbance. Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. Because sea otter populations are structured on relatively small spatial scales (on the order of tens of km) due to the high site fidelity and limited movements of individuals, particularly adult females (Gorbics and Bodkin 2001; Tinker et al. 2006; 2008b; 2019a; 2019b, pp. 63–85; Tarjan and Tinker 2016; Breed et al. 2017), we followed Tinker et al. (2021a) in dividing the southern sea otter population into coastal areas representing semi-distinct subpopulations. These subpopulations may be considered demographically homogeneous but are connected within the larger population by the dispersal of animals between areas. Although we recognize the precise number and geographic boundaries of extirpated historical subpopulations can never be known with any certainty, we use the term *subpopulation* throughout this document as shorthand for “the sea otters that exist along or could be supported by the habitat in a coastal area.” We describe the resources, circumstances, and demographics that most influence the resiliency of each subpopulation of the southern sea otter across different ecological settings.
- Species level: *Redundancy* spreads risk among multiple populations or areas to minimize the damage due to large-scale, high-impact (i.e., catastrophic) events. Redundancy is characterized by having multiple resilient populations distributed within the species’ ecological settings and across the species’ range. It can be measured by population number, resiliency, spatial extent, and degree of connectivity. *Representation* describes the ability of a species to adapt to changing environmental conditions over time. It is

² The Act defines “species” as including any subspecies of fish or wildlife or plants, and any distinct population segment (DPS) of any species of vertebrate wildlife. Although taxonomically the southern sea otter is a subspecies, we generally refer to it as a species, in this legal sense, throughout the remainder of this document.

characterized by the breadth of genetic and environmental diversity within and among populations. Measures may include the number of varied niches occupied or genetic diversity, as assessed by heterozygosity or alleles per locus. We explored what factors influence redundancy and representation for the southern sea otter by examining its evolutionary history and historical distribution to understand how it functioned to maintain populations across its range.

We researched and evaluated the best available scientific and commercial information on the southern sea otter's life history and habitat and its ecological needs at the individual, population, and species levels. We consulted an extensive body of biological, ecological, anthropological, and historical research in peer-reviewed journal articles, academic theses and dissertations, books, and government reports.

2.2 Current Species Conditions

The SSA describes the current conditions of the species' habitat and demographics and the probable explanations for past and ongoing changes in abundance and distribution within the species' ecological settings (i.e., areas representative of the geographic, genetic, or life history variation across the range). We describe the current (a) population structure, distribution, abundance, demographic rates, ecological and genetic diversity, and habitat of the southern sea otter; (b) changes from historical to current conditions (i.e., distribution and abundance) of the southern sea otter; and (c) explanations or hypotheses of the causes and effects of stressors and conservation efforts that resulted in the current conditions of the southern sea otter.

We grouped the subpopulations of southern sea otters into four ecoregions: Mendocinian, Montereyan, Southern Californian, and Magdalenian. We based these ecoregions on biogeographic provinces defined by Fenberg et al. (2015) through their analysis of 406 species of macroinvertebrates and algae at rocky intertidal field sites within the California Current (Figure 3). Although sea otters are most often found in subtidal (not intertidal) waters, we believe these biogeographic provinces are appropriate for our purposes because they primarily reflect variables related to upwelling—such as sea surface temperature, nutrient concentrations, and the magnitude of the seasonal switch between upwelling and downwelling at each site (Fenberg et al. 2015, p. 90)—which affect the entire nearshore marine ecosystem.

We used an analysis of estimated density at carrying capacity (K) at the scale of coastal areas (Tinker et al. 2021b; see section 5.2) to evaluate the capacity of unoccupied habitat to support sea otter subpopulations. We used historical information and an integrated population model (IPM), which incorporates all available data on southern sea otter trends, vital rates, and mortality risk factors (Tinker et al. 2021a), to examine the past and current influences of environmental features and mortality risk factors on the resiliency of subpopulations. An implicit assumption of the IPM is that any health-based and/or demographic effects of low genetic diversity (i.e., inbreeding depression) are accounted for in the fitted model because the loss of genetic diversity occurred historically when abundance was very low and there is no evidence of continued loss of diversity at current population sizes and levels of connectivity (see section 3.2). The IPM also does not explicitly account for demographic stochasticity because the current abundance (~3000) and degree of connectivity between sub-populations via dispersal is expected

to swamp any such effects. The IPM has a web tool that allows users to modify risk factors (hazards), either directly or indirectly by adjusting environmental conditions, and to simulate a range of oil spill scenarios. The model dynamics are calculated in terms of independent (non-pup) sea otters; an estimate of abundance that includes pups is obtained by multiplying abundance estimates by a factor of 1.17 (which corresponds to the observed long-term average ratio of pups to independents of 17%; Tinker et al. 2021b, p. 16). Model outputs presented in this document include pups. Based on these results and additional information on distribution and genetic diversity, we then evaluated redundancy and representation at the species level.

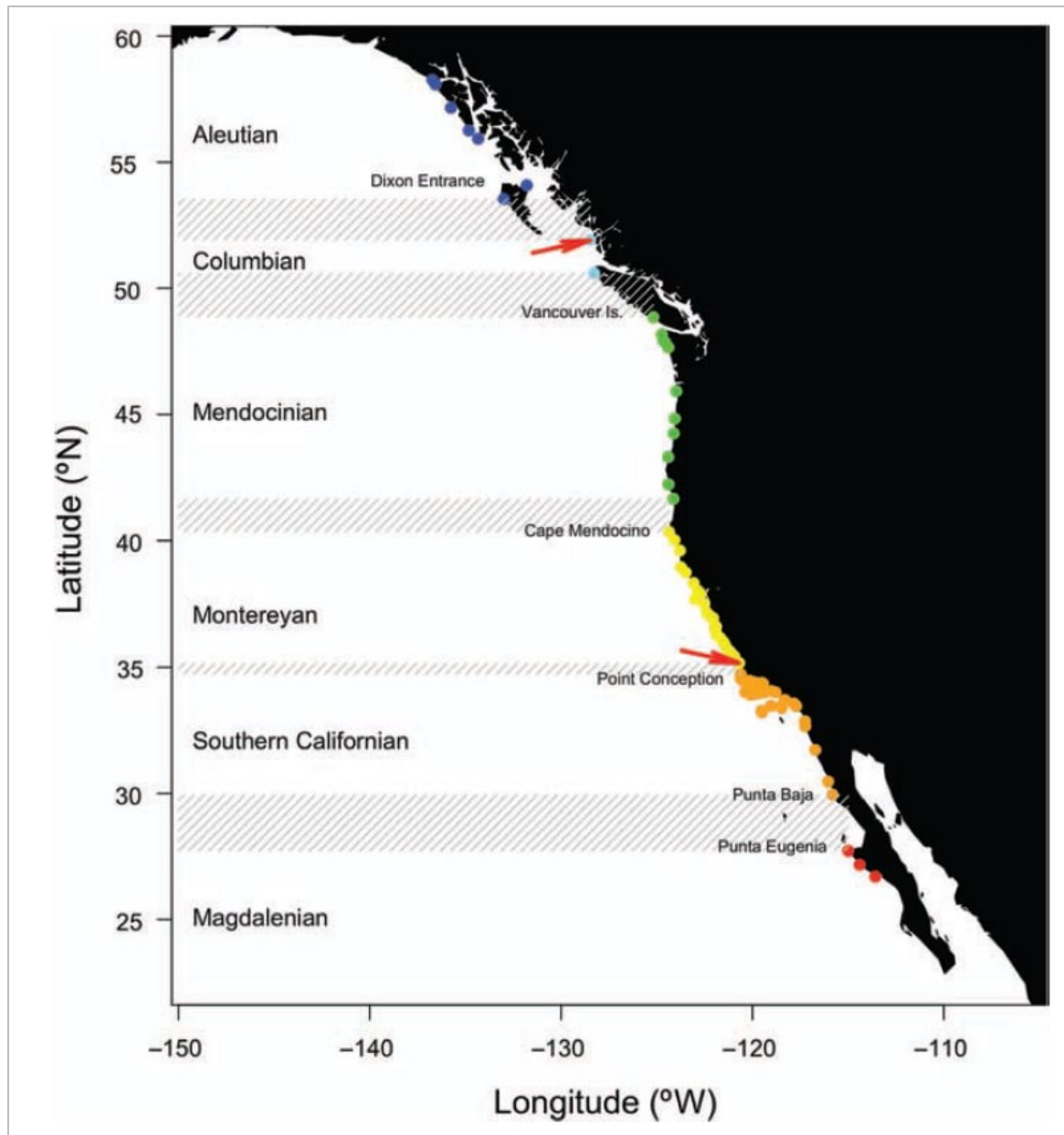


Figure 3. Biogeographic provinces defined by Fenberg et al. (2015) based on biogeographic structure of rocky intertidal organisms along the northeastern Pacific. Light-gray areas indicate unsampled regions. Reproduced from Fenberg et al. (2015, p. 88), doi: 10.1111/ecog.00880, ©2014 by The Authors. Licensed under CC BY.

2.3 Future Species Conditions

SSAs characterize a species' ability to sustain populations in the wild over time (its viability) based on the best scientific understanding of current and future abundance and distribution within the species' ecological settings. SSAs forecast a species' response to a range of plausible future scenarios designed to explore the response of the species to changing environmental conditions.

The Fifth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC 2014) established Representative Concentration Pathways (RCP) to project differing rates of climate change by the year 2100. The report uses the concept of *radiative forcing* (RF) to quantify atmospheric conditions that are the result in part, but not entirely, of emissions of greenhouse gases (GHGs). With increasing RF, global temperatures rise and other changes in climate occur. The RCP scenarios, RCP 2.6, 4.5, 6.0, and 8.5, expressed in units of watts per meter squared (W m^{-2}), span the range of values for RF in the scientific literature—from low to intermediate to high—and describe the different pathways by which various RF conditions might be reached. The RCPs have been widely used throughout the scientific community for modeling a broad range of possible future climate conditions. The IPCC is now in its sixth assessment cycle, and it is scheduled to approve the Synthesis Report to the Sixth Assessment Report in late 2022 or early 2023 (<https://www.ipcc.ch/report/sixth-assessment-report-cycle/>). The nine scenarios prepared for the sixth assessment, called Shared Socioeconomic Pathways (SSPs), span a greater range of climate outcomes ($1.9\text{--}8.5 \text{ W m}^{-2}$), but four of the new SSPs provide continuity with the RCPs (Gidden et al. 2019). Because the studies we use to anticipate environmental changes relevant to southern sea otters under different climate scenarios reference the RCPs, we refer to RCPs, not SSPs, throughout the remainder of this document.

For the southern sea otter SSA, we developed a baseline scenario projecting current conditions and three other projections based on the RCP scenarios. In the baseline scenario, we used the southern sea otter IPM to project current conditions into the future. Although this projection is not realistic because environmental conditions are expected to change, it serves as a basis for comparison with other scenarios and demonstrates the effects of changes in hazard rates under these other scenarios. We based three plausible future scenarios on RCP 4.5, RCP 8.5, and RCP 8.5 with a large oil spill, respectively. The scenarios were designed to explore the response of the species to environmental stressors and represent the range of plausible scenarios given the information currently available. RCP 4.5 is at the low end of the intermediate range of conditions and represents a situation under which key atmospheric conditions would stabilize at a moderate level shortly after 2100. RCP 8.5 is at the high end of such conditions and is consistent with a future in which there are no significant global efforts to limit or reduce emissions. We did not use RCP 2.6 because numerous scientific papers show that although it is theoretically possible to achieve this pathway and outcome, key assumptions underlying it (including a very rapid reduction in GHG emissions) have already not been met, and other future activities it relies upon are highly speculative.

Under each scenario, we used the southern sea otter IPM (Tinker et al. 2021a) to model abundance and distribution 50 years into the future from 2021, using census data from 2019, the most recent year for which range-wide survey data are currently available. We limited our

projections to 50 years because numerous indirect effects and complex ecosystem interactions will influence southern sea otter viability, and available information does not allow us to reliably anticipate trends beyond this time horizon. Even within this time horizon, reliability declines over the time, as reflected in the widening credible intervals. Because the IPM is not tied to particular RCP scenarios but rather allows for the spatially explicit adjustment of risk factors known to affect southern sea otters, we researched the probable effects of different levels of RF on the risk factors known to affect sea otters and adjusted those risk factors accordingly to obtain projections of the distribution and abundance of southern sea otters. It is important to note that although the IPM output is quantitative, the adjustments to risk factors are based on a qualitative understanding of the relationship between modeled climate scenarios and the risk factors affecting southern sea otters. However, in each case we explicitly state which risk factors we adjusted and the magnitude and rationale for the adjustment. Some climate-related changes under future RCP scenarios, such as the effects of ocean acidification on prey availability, were incorporated into IPM projections by adjusting carrying capacity (K) downward, which in turn affected the risk factors associated with density dependence.

Using the IPM output under the different scenarios and independent genetic modeling (Beichman et al. 2022), we evaluated the ability of southern sea otters to sustain resilient populations in natural coastal ecosystems over time. Using the SSA framework, we considered what the southern sea otter needs to maintain viability by characterizing the status of the species in terms of its resiliency, redundancy, and representation (Service 2016; Smith et al. 2018). We evaluated resiliency—the species’ ability to withstand stochastic events—by assessing each subpopulation’s size, growth rate, and connectivity. We evaluated redundancy—the species’ ability to withstand catastrophic events—by assessing the number and distribution of resilient subpopulations across the range of the species in relation to plausible catastrophic events. We evaluated representation—the species’ ability to adapt to changing physical and biological conditions—by assessing genetic diversity and the number of different environmental settings in which resilient subpopulations occur.

3.0 SPECIES BACKGROUND

Although southern sea otters are genetically and morphologically distinct (see section 3.2), the biology, ecology, and behavior of sea otters are generally consistent across the three recognized subspecies (Estes and Bodkin 2002). In the sections that follow, we reference studies that were conducted on northern sea otters as well as those that were conducted on southern sea otters, though we make distinctions as appropriate.

3.1 Physical Description

Southern sea otters are medium-sized furred mammals. They reach adult length at 4–6 years of age, with females averaging 118 centimeters (cm) (46.5 inches [in]) and males averaging 127 cm (50 in). Females reach adult weight at around this same age, averaging 21 kilograms (kg) (46 pounds [lbs]), whereas males continue to gain muscle mass until about 8 years of age, when they weigh an average of 29 kg (64 lbs) (Tinker et al. 2019b, pp. 138–151). Their short forelegs and highly sensitive padded paws (Strobel et al. 2018) are used for grooming and for locating, retrieving, and manipulating prey. The hind feet are webbed, with the longest digit on the outside

of the foot. The tail is long relative to those of pinnipeds, somewhat flattened, and of uniform width. They have powerful jaws and blunt molars and premolars suited to grinding and crushing prey. Unlike other marine mammals, sea otters do not have a blubber layer for insulation or energy storage but instead rely on their pelage and a high metabolic rate to maintain homeostasis (Costa and Kooyman 1984; Yeates et al. 2007). The pelage consists of primary and secondary guard hairs and an undercoat (Liwanag et al. 2012), with up to 164,662 hairs/cm² (1,062,070/in²) in the densest locations on the forelimb (Williams et al. 1992). The pelage color ranges from buff or brown at birth to darker shades of brown or nearly black in adults. In some individuals the guard hairs may lose pigmentation as the animal ages, resulting in a grizzled appearance—a light or nearly white coloration that progresses from the head and neck to chest and forelimbs (Riedman and Estes 1990, p. 21). The sea otter's skin is very loose, enabling all areas of the fur to be pulled to the mouth for grooming. There is also a loose pouch-like area under each forearm where prey items or a tool retrieved on a foraging dive may be stored during ascent and while other items are being consumed at the surface (Kenyon 1969, pp. 17, 111–112).

3.2 Genetics

Numerous studies have used genetic data to detect spatial structuring among sea otter populations (Rotterman 1992; Sanchez 1992; Cheney 1995; Cronin et al. 1996), but these studies do not cleanly support the number and distribution of subspecies proposed by Wilson et al. (1991) based on skull morphometric data. Cronin et al. (1996) analyzed samples from eight locations throughout the sea otter's range, including the Kuril Islands and Medny Island (in the Commander Islands), and identified four major population groupings based on mtDNA haplotypes: the Kuril Islands; Kodiak-Adak-Amchitka-Attu-Medny islands; Prince William Sound; and California.³ Although Cronin et al. (1996) generally confirmed the taxonomy of Wilson et al. (1991), they found more structuring among subspecies based on mtDNA than was evident from the morphometric data: *E. l. nereis* had unique haplotypes and was distinct from other populations, but populations considered to be *E. l. lutris* in the Kuril and Medny islands differed notably in their frequencies of haplotypes, as did populations considered to be *E. l. kenyoni* in Prince William Sound and other areas of Alaska.

Using a genomic approach, Beichman et al. (2022) detected broad-scale population structure across the sea otter's range and fine-scale population structure within northern and Asian populations. They identified five geographically driven groupings (Kuril Islands; Commander Islands; western Aleutian Islands; south central Alaska; and California) that existed before the fur trade and persisted despite fur trade bottlenecks because remnant populations survived in each of these areas. Of these groupings, the southern sea otter in California was the most distinct, having diverged from other sea otters to the north about 28,000 years ago (about the time of the Last Glacial Maximum), before the northern and Asian populations differentiated from each other (Beichman et al. 2022). Beichman et al. (2022) propose that ice cover along much of the British Columbian and Alaskan coast may have isolated the southern sea otter from these other populations, with only limited gene flow after the ice retreated.

³ Contemporary populations in Southeast Alaska, British Columbia and Washington are the result of translocations from Amchitka Island and Prince William Sound, Alaska (Jameson et al. 1982).

By 1911, fur-trade-era hunting had reduced sea otter numbers by more than 99%, causing a substantial loss of genetic diversity in remnant populations (Cronin et al. 1996; Bodkin et al. 1999; Larson et al. 2002a, 2002b, 2012; Aguilar et al. 2008; Gagne et al. 2018; Beichman et al. 2019, 2022). Remnant populations suffered from small population sizes and a lack of gene flow due to their geographic isolation (Larson et al. 2015). Based on a study of microsatellite diversity, Larson et al. (2012) determined that modern sea otter populations have lost an average 33% of their pre-fur-trade heterozygosity and 69% of their pre-fur-trade alleles (from 19.8 to 6.2 alleles per locus). Genomic analysis has also revealed extremely low heterozygosity, evidence of recent inbreeding (which likely occurred in the small remnant populations after the post-fur trade bottleneck), and an elevated burden of potentially deleterious alleles in both the northern sea otter and the southern sea otter (Beichman et al. 2019, 2022). Genetic variation in the southern sea otter, in particular, is among the lowest observed for any mammal and is similar to that seen in other species that have undergone population bottlenecks or persistent population declines, such as the northern elephant seal (*Mirounga angustirostris*) and the Mediterranean monk seal (*Monachus monachus*), though with very different subsequent population trajectories (Larson et al. 2002a; Aguilar et al. 2008; Gagne et al. 2018).

To assess genetic diversity within the southern sea otter, Gagne et al. (2018) analyzed samples from 1,006 southern sea otters at 38 microsatellite loci and found little evidence for population genetic structure within the remnant population in California; however, there was some evidence of genetic isolation by distance north and south from Big Sur, the location from which the contemporary population expanded its range as it grew. Notably, they were unable to obtain consistent estimates of the effective population size (N_e), which may be thought of in simplified terms as the number of breeding individuals (those contributing genetic material) in a population (Gagne et al. 2018, pp. 2, 5). Based on a genetic estimator (linkage disequilibrium), N_e for the southern sea otter population as a whole was 341 (95% CI 287–410), whereas the demographically estimated N_e was 1,230 (95% CI 1,087–1,272) (Gagne et al. p. 8). Frankham et al. (2014, pp. 58–59) suggest that an effective population size of $\geq 1,000$ is needed to maintain evolutionary potential in perpetuity (i.e., the loss of genetic variation through random genetic drift is expected to be balanced or exceeded by the gains of mutation). The number of actual individuals in a population required to achieve the desired effective population size varies with the genetic diversity in the population (i.e., populations with higher genetic diversity will require fewer actual individuals to achieve a target effective population size; Frankel and Soulé 1981). While genetic and demographic estimates of effective population size (N_e) in the southern sea otter did not agree at the scale of the entire occupied range, they did agree at the scale of approximately Monterey County, suggesting that this subregional scale is more appropriate for considering effective population sizes in this species (Gagne et al. 2018).

The loss of genetic variation in sea otter populations is consistent with the extreme population bottleneck caused by the fur trade (Larson et al. 2002b; Larson et al. 2012). However, Aguilar et al. (2008) suggested that the decline in effective population size of southern sea otters may have begun up to 550 years ago, potentially as a result of a sustained small breeding population or a severe reduction in population size that predates the fur trade, and Larson et al. (2012) also detected evidence of pre-fur-trade bottlenecks. Genomic analysis has further indicated that population bottlenecks—or other changes in population structure, population mixing, or

migration—occurred in both northern and southern sea otters sometime around 35,000–45,000 years ago, and that an additional more recent (but still pre-fur-trade) bottleneck may have occurred in southern sea otters 1,000–3,000 years ago (Beichman et al. 2019). The authors speculated that the more recent pre-fur-trade decline may reflect exploitation by indigenous people (Aguilar et al. 2008, p. 42; Larson et al. 2012, p. 9 and Beichman et al. 2019, p. 16) and that ancient declines may reflect changes in sea level resulting from climate change around the last glacial maximum (Beichman et al. 2019, p. 16). Beichman et al. (2022) also estimated small ancestral effective population sizes relative to historical census sizes in all sea otter populations even before the fur trade bottleneck occurred and proposed they could be the result of limited resource availability, high variance in reproductive success due to strong polygyny, local structure caused by linear coastline geography, or hunting by aboriginal peoples. Regardless of cause, historical bottlenecks, along with more recent bottlenecks, have contributed to the current low levels of genetic variation in all sea otters, but especially southern sea otters.

The southern sea otter population has increased to a size ($\approx 3,000$ individuals) where it is no longer inbreeding and losing genetic variation (Gagne et al. 2018, pp. 4–5). Additionally, its fecundity and survival rates (Gerber et al. 2004, pp. 1558–1559) are comparable to those of northern sea otter populations (after controlling for density-dependent effects) and adequate to sustain population growth where extrinsic factors (e.g., prey availability or shark-bite mortality) are not limiting. However, the potentially deleterious alleles and low levels of genetic variation detected in southern sea otters are a concern for management because inbred animals are generally less resilient to stress than outbred ones, and populations with low genetic diversity are less able to adapt to environmental change. Simulations conducted by Beichman et al. (2022) demonstrated that, without intervention, recessive genetic load (an excessively high frequency of recessive deleterious alleles in homozygous state) in southern sea otters did not return to pre-fur-trade levels for 400 generations or $\approx 2,800$ years, though it was unlikely to cause extinction on its own.

If southern and northern sea otters came into contact and interbred, it could potentially improve the health and adaptive ability of southern sea otters (Gagne et al. 2018, p. 9; Larson et al. 2021; Beichman et al. 2022, p. 13). Genomic data are revealing that hybridization between populations, subspecies, and even species is a normal occurrence in both animals and plants and often provides an important source of new genetic variation upon which natural selection can act (vonHoldt et al. 2018; Taylor and Larson 2019). The potential for increased gene flow and decreased genetic load through heterosis (outbreeding vigor) has been proposed as a contributing rationale for reintroduction of sea otters (Bodkin et al. 1999, pp. 1383–1384; Davis et al. 2019, p. 8; Larson et al. 2021; Beichman et al. 2022, p. 13; Service 2022, pp. 36–37).

3.3 Life History

Sea otters exhibit many of the life history traits typical of *K*-strategists, including density-dependent mortality, uncertain juvenile survival, population sizes near carrying capacity (*K*) on average, slow physical development and delayed age of reproduction, large body size, iteroparity, low birth rates, high parental investment, small clutch or litter size, low reproductive effort, and longer time between generations (Estes 1979, Monson and Bowen 2015). *K* was defined by Odum (1953, p. 122) as the “upper level beyond which no major increase can occur

(assuming no major changes in environment).” In other words, it is the greatest population abundance that can be sustained over the long term in a particular environment relative to available resources, such as food, water, and space. As sea otter populations reach K , recognizable patterns of changes in feeding behavior, diet, activity budgets, morphology, survival, and reproductive success occur (Estes et al. 1996; Bodkin et al. 2004; Tinker et al. 2021a, p. 18; Figure 4). These changes include an increase in percent time spent foraging; a dietary shift to include smaller and lower-value prey; a decrease in energy recovery rates; a reduction in body size and condition; an increase in the mortality of pups, juveniles, and aged animals; and an increase in adult female mortality due to end-lactation syndrome (ELS), a case presentation in which females exhibit moderate to severe emaciation (independent of any concurrent disease process) and die at or shortly after the time of pup weaning (Chinn et al. 2016, 2018; Estes 1990; Monson et al. 2000b; Tinker et al. 2008a, 2017, 2018, 2019b; Thometz et al. 2016a, 2016b).

Sea otters are polygynous. Males reach breeding age when they are about 5 years old, at which time they may attempt to establish a breeding territory in a female area or become roaming or satellite males looking for mating opportunities on the periphery of male territories (Tarjan 2016, pp. 89–95). Most females reach sexual maturity at 3–4 years of age. All females are mature by age 5 and show little to no evidence of reproductive senescence before they die (Loughlin et al. 1981; Jameson and Johnson 1993; Riedman et al. 1994; Monson et al. 2000b).

Mating and pupping occur throughout the year. Peak pupping periods were formerly reported as January–March, with a secondary peak from late summer to early fall (Riedman and Estes, p. 59), though the peak based on more recent studies is from October to January (Chinn et al. 2016, p. 314). The gestation period lasts approximately 6 months, consisting of an unimplanted phase of 2–3 months during which the embryo remains unattached to the uterine wall (delayed

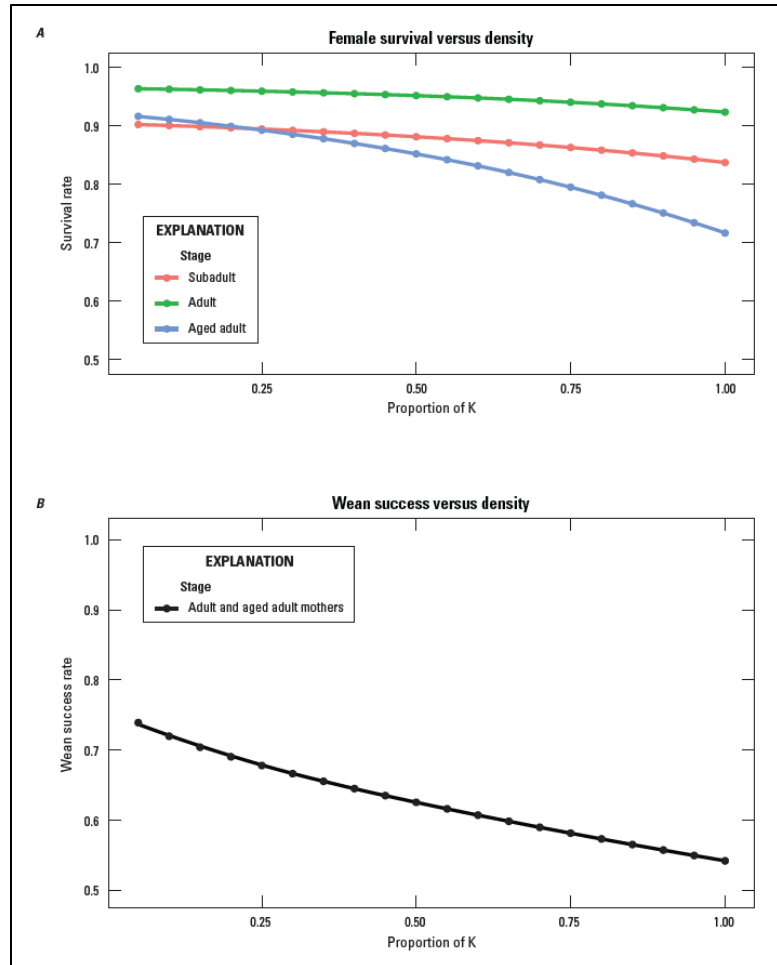


Figure 4. Female survival and weaning success relative to population density (proportional abundance relative to local carrying capacity, or K). Effects of population density on age-specific survival rates of males (not pictured) were similar. Reproduced from Tinker et al. (2021a), p. 18.

implantation) and an implanted phase of 4 months (Jameson and Johnson 1993). Females typically give birth to a single pup annually and provide sole care of their pup for approximately 6 months until weaning (Riedman and Estes 1990, pp. 66–68; Jameson and Johnson 1993). Although birth rates are generally invariant (Riedman and Estes, p. 82), the successful weaning of pups is strongly dependent on the body condition of the mother (Jameson and Johnson 1993; Monson et al. 2000b; Staedler 2011). Pup rearing and provisioning impose high energetic costs on females, with energy demands on the female more than doubling by the time a pup is near weaning (Thometz et al. 2014; 2016a; 2016b). Increased energy demands require females to increase foraging effort during the pup dependency period and may leave them highly susceptible to stressors they encounter when they come into estrus after weaning, such as parasite infections or aggression by males (Tinker et al. 2019b, pp. 152–178). These effects are especially evident in areas where per-capita prey abundance is limiting and sometimes lead to ELS (Chinn et al. 2016; Chinn et al. 2018). ELS is the name given to a case presentation in which females exhibiting moderate to severe emaciation (independent of any concurrent disease process) die at or shortly after the time of pup weaning (Chinn et al. 2016).

Pup and juvenile survival are particularly sensitive to the population's local status with respect to K (Monson et al. 2000b). Females often use geographic separation to facilitate weaning (i.e., they travel to a new location with their pup and leave it there when they return to their home range) (Breed et al. 2017). Shortly after weaning, juveniles disperse, with juvenile males moving further from natal groups than juvenile females. Territorial males exclude juvenile and subordinate males from their territories, so the longer dispersal distances of juvenile males may be partly due to the aggressive behavior exhibited by breeding males (Ralls et al. 1996; Jameson 1998). After dispersing from their mother's home range, juvenile males congregate in areas with few females. These male areas are typically occupied by non-territorial males and subadult animals of both sexes and only rarely by adult females and pups (Tinker et al. 2008b). They tend to occur at the periphery of expanding populations (Wild and Ames 1974, p. 44; Garshelis et al. 1984; Lafferty and Tinker 2014) but in long-established populations, or where the range peripheries are distant, they can also occur in other geographically isolated locations. Within the southern sea otter range, male areas often occur in sandy and soft-bottom habitats (particularly those in Monterey Bay, Estero Bay, and from Pismo Beach to Point Sal). Many males formerly migrated to the range peripheries during the winter and early spring, apparently to take advantage of more abundant prey resources there, but then returned to the range center in search of estrous females during the period when most breeding occurs (Jameson 1989; Ralls et al. 1996; Tinker et al. 2008b). However, this pattern has not been observed for more than a decade. Rocky, kelp-dominated areas are occupied primarily by females, dependent pups, and territorial males (Tinker et al. 2008b; Nicholson et al. 2018).

The home ranges of southern sea otters appear to reflect coastal bathymetry and the distribution of resources, as well as reproductive strategy. For instance, a study found that although absolute home range sizes did not differ between Big Sur and Monterey, the coastline extent of home ranges in Big Sur was greater, reflecting the distribution of prey resources within depths accessible to sea otters along the narrow coastal shelf off Big Sur relative to the wide continental shelf off Monterey (Tarjan and Tinker 2016; Tinker et al. 2019b, pp. 71, 74). Males exhibit two distinct home range strategies that reflect their reproductive status: territorial males maintain strong site fidelity to a small home range consisting of a single center of use; males that are non-

territorial (or territorial only during certain parts of the year) move between multiple range centers over a larger total area and over a longer span of coastline (Tarjan and Tinker 2016; Tinker et al. 2019b, pp. 63–85). Female home range characteristics fall between those of the two groups of males (Tinker et al. 2019b, pp. 63–85). Compared to males, which may travel long distances, most females are more sedentary (Ralls et al. 1996; Tinker et al. 2006, p. 53). Adult females rarely disperse more than 20 km (12 mi) within a 1-year period (Ralls et al. 1996; Tinker et al. 2019b, pp. 63–85, Breed et al. 2017); however, some adult females will travel longer distances up to 40–50 km (25–31 mi) (Tinker et al. 2006), and subadult females are more likely to make occasional long-distance movements than adult females (Ralls et al. 1996, Tinker et al. 2019b, pp. 63–85). All females move freely across male territories.

Typical life spans for wild southern sea otters are 12–18 years for females and 10–15 years for males (USGS unpublished data), although individuals may live longer. One female sea otter translocated to San Nicolas Island in 1987 as a juvenile was documented in 2006 to have reached at least 19 years of age in the wild (USGS unpublished data). Sea otters of all subspecies transition through life stages, from pup (0–6 months) to subadult (6 months–3 years) to adult (4–10 years) to aged adult (>10 years) (Figure 5).

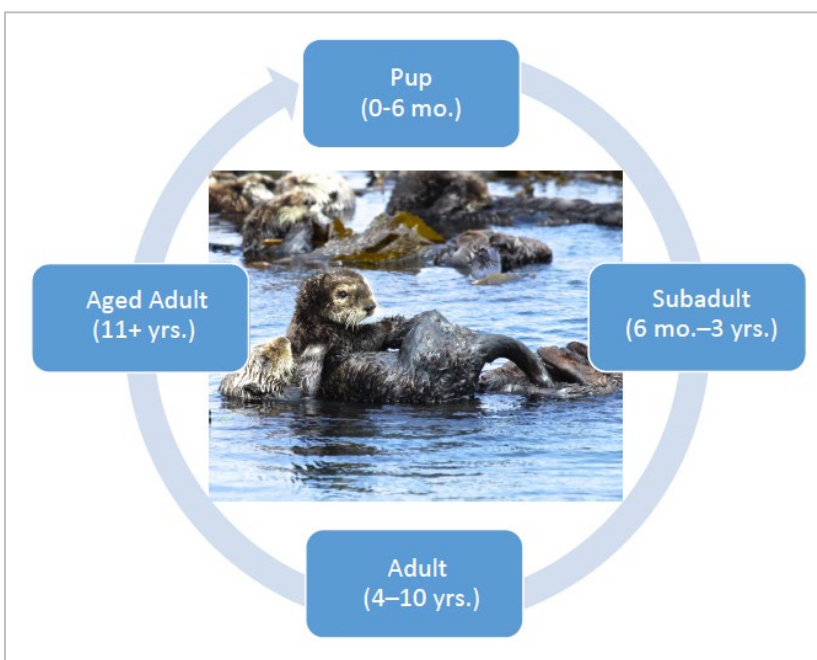


Figure 5. Life cycle of the sea otter.

3.4 Habitat

Sea otters, including southern sea otters, occupy a variety of coastal marine habitats, including rocky exposed coastline, sandy embayments, and estuaries. Sea otter habitat in California is typically defined by the 40 m (131 ft) (Riedman and Estes 1990, p. 31; Laidre et al. 2001) or 60 m (197 ft) depth contour (Tinker et al. 2021b). Depending on local bathymetry, most sea otters in California reside within 2 km (1.2 mi) of shore. Foraging occurs in both rocky and soft-sediment communities in water depths 25 m (82 ft) or less, although some animals utilize deeper waters. Sea otters occasionally make dives of up to 100 m (328 ft), but the vast majority of feeding dives (about 95%) occur in waters less than 40 m (131 ft) in depth (Bodkin et al. 2004; Thometz et al. 2016b, p. 1556; Tinker et al. 2006, pp. 138–142; 2019b, p. 90). Dive depth and dive pattern vary by sex (males tend to make dives greater than 25 m (82 ft) more frequently than females; Bodkin et al. 2004), geographic location, and diet specialization (Tinker et al. 2006, pp. 138–142; Tinker et al. 2007), as well as age and reproductive status (Thometz et al. 2016b; Tinker et al. 2019b, pp. 86–107).

Rocky habitats that are topographically heterogeneous and support forests of giant kelp (*Macrocystis pyrifera*) or bull kelp (*Nereocystis luetkeana*) provide the greatest diversity and abundance of sea otter prey, including abalone, rock crabs, sea urchins, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, and chitons. Canopy-forming kelp dampens swells, provides a means of anchoring to allow sea otters to rest without repeatedly repositioning, serves as important nursery habitat (Riedman and Estes 1990, pp. 23, 62), and may provide some protection from fatal bites by white sharks (*Carcharodon carcharias*) (Nicholson et al. 2018). In soft-sediment areas, benthic invertebrate assemblages are less diverse, and sea otters typically consume burrowing infaunal species, such as clams and marine worms, and epifaunal invertebrates, such as sand dollars and crabs (Kvitek et al. 1988; Newsome et al. 2015). In these areas canopy-forming kelp is absent because of the lack of adequate substrate to anchor holdfasts, and sea otters may rest in open water or protected embayments. In the Elkhorn Slough estuary, the only estuarine habitat within their historical range that southern sea otters have fully recolonized, foraging occurs mostly in the subtidal main channel, where sea otters feed on large clams (*Tresus nuttallii* and *Saxidomus nuttalli*), worms (*Urechis caupo*) and crabs (*Cancer* sp.). Some foraging on shore crabs (*Pachygrapsus* spp.) also occurs within tidal creeks and salt marsh, especially at high tides. Resting occurs mostly in eelgrass beds, salt marsh tidal creeks, and salt marsh areas that are protected from human disturbance, where sea otters often haul out onto the banks, using areas to at least 50 m, and as much as 200 m, from the water's edge (Eby et al. 2017; Tinker et al. 2018).

Because of the relatively small home ranges of sea otters in general and of adult females in particular (Tarjan and Tinker 2016), equilibrium densities are determined at small spatial scales (tens of km) (Gorbics and Bodkin 2001; Tinker et al. 2019a; Tinker et al. 2021a, 2021b). The most important limiting resource is high-quality prey, although coastal bathymetry and water depth (Thometz et al. 2016b), kelp canopy cover (Nicholson et al. 2018), benthic substrate complexity and composition (Stewart et al. 2015; Tinker et al. 2017) and ocean productivity (Davis et al. 2019, pp. 5–7) also affect sea otter abundance and foraging success (Tinker et al. 2021b). Estimated southern sea otter densities at K range from >10 sea otters per km^2 around the Monterey Peninsula and in Elkhorn Slough estuary to <1.5 sea otters per km^2 in sandy habitats such as northern Monterey Bay or Point Sal (Laidre et al. 2001; Tinker et al. 2021b).

Sea otters exhibit density-dependent variation in their diets (Ostfeld 1982; Tinker et al. 2007; 2008a; 2009; 2012; Newsome et al. 2010; 2015). In areas of low sea otter density relative to K —for example, where sea otters recolonize portions of their historical range—sea otters forage almost exclusively on large, abundant, calorically rich invertebrates such as sea urchins, large clams and crabs, or abalone. As these prey types become depleted over time, the diet of the population diversifies to include smaller, lower-calorie prey. In areas of high sea otter density relative to K , sea otters exhibit high levels of intraspecific variation in diet, with some individuals specializing on snails, some on mussels and urchins, others on large crabs and abalone, and others adopting a generalist diet (Estes et al. 2003; Tinker et al. 2008a).

Because of their size-selective consumption of large quantities of marine invertebrates, sea otters can profoundly influence their habitat. In rocky exposed habitats, sea otters limit populations of herbivorous invertebrates that can otherwise limit kelp and other macroalgae (Estes and

Palmisano 1974; Van Blaricom and Estes 1988; Estes and Duggins 1995). In the estuarine habitat of Elkhorn Slough, sea otters promote eelgrass abundance and distribution by reducing the abundance and size of crabs, which consume mesograzers, primarily an isopod (*Idotea resicata*) and a sea slug (*Phyllaplysia taylori*). These mesograzers feed on epiphytic algae, which can harm eelgrass by blocking the sunlight essential for photosynthesis (Hughes et al. 2013).

3.5 Individual Needs

We assessed the best available information to identify the physical and biological needs to support individual fitness at all life stages for the southern sea otter. For the purposes of this SSA, we considered the most important elements to be benthic invertebrate prey, coastal marine waters <40 m in depth, canopy-forming kelp, shallow protected waters (e.g., estuaries), haulout areas, and maternal care. Some of these resources are essential and non-exchangeable, whereas others are exchangeable, in that sea otters' needs can potentially be satisfied by any one of several resource types, depending on location or context. Specifically, benthic invertebrate prey, coastal marine waters <40 m in depth, and maternal care are essential. Canopy-forming kelp, shallow protected waters, and haulout areas are exchangeable because each of these habitat features can fulfill the need for sheltered resting habitat. Table 1 summarizes the essential (E) and exchangeable (X) individual resource needs of southern sea otters by life stage.

TABLE 1. INDIVIDUAL RESOURCE NEEDS OF SOUTHERN SEA OTTERS, DEFINED AS EITHER ESSENTIAL (E) OR EXCHANGEABLE (X), BY LIFE STAGE.

INDIVIDUAL NEEDS	LIFE HISTORY STAGE			
	PUP	JUVENILE	SUBADULT	ADULT & AGED ADULT
Benthic invertebrate prey	E	E	E	E
Coastal marine waters <40 m in depth	E	E	E	E
Canopy-forming kelp	X	X	X	X
Shallow protected waters (e.g., estuaries)	X	X	X	X
Haulout areas	X	X	X	X
Maternal care	E			

3.5.1 Benthic invertebrate prey

Southern sea otters consume a broad variety (≈ 75 species) of benthic invertebrate prey, including abalone, sea urchins, rock crabs, kelp crabs, clams, turban snails, mussels, octopus, barnacles, scallops, sea stars, chitons, clams, marine worms, and sand dollars (Riedman and Estes 1990, 34–45; Tinker et al. 2019b, 108–137). When the population density is low relative to local K , individuals consume similar diets of large, abundant, calorically rich invertebrates (e.g., sea urchins, large clams, abalone). When population density is high relative to local K , individuals specialize on a particular diet type (e.g., snails, crabs and abalone, or generalist) while the breadth of the diet of the population as a whole increases (Estes et al. 2003; Tinker et al. 2008a). Benthic invertebrate prey is important to individuals of all life stages, although pups 0–6 weeks of age mostly suckle (Payne and Jameson 1984). Pups become progressively more dependent on prey obtained by their mother until the last month before weaning, when they suckle only rarely

and subsist mostly on prey obtained from their mother or on their own dives (Payne and Jameson 1984).

3.5.2 Coastal marine waters <40 m in depth

Sea otters use a variety of coastal marine habitats, the usable depths of which are limited by their dive capacity. About 95% of feeding dives occur in waters less than 40 m (131 ft) in depth (Bodkin et al. 2004; Thometz et al. 2016b, p. 1556; Tinker et al. 2006, pp. 138–142; 2019b, p. 90). Males tend to make dives greater than 25 m (82 ft) more frequently than females. Adult females tend to be limited to shallower waters within the diving range of dependent pups (Tinker et al. 2019b, pp. 86–107).

3.5.3 Canopy-forming kelp

Kelp canopy is important resting habitat on the outer coast because it reduces tidal energy and provides a means of anchoring, removing the need for repeated repositioning when sea otters are resting. It also serves as important nursery habitat for adult females and their pups (Riedman and Estes 1990, pp. 23, 62). Kelp cover appears to provide some protection from fatal bites by white sharks. At a spatial scale of 5 km (3.1 mi), a 10% increase in kelp cover has been associated with a 99% reduction in the probability of shark bite (Nicholson et al. 2018). Kelp forests also provide food and habitat for many prey species. Although kelp canopy is important, it is not essential to sea otters, as other habitat features can serve many of the same purposes in other habitat types.

3.5.4 Shallow protected waters (e.g., estuaries)

Estuarine habitat is not essential (most southern sea otters alive today will never encounter an estuary), but sea otters occupy estuarine systems wherever they co-occur. Southern sea otters have fully recolonized only one estuary within their historical range, Elkhorn Slough, though increasing numbers of sea otters have also been detected in Morro Bay over the past decade. Our understanding of estuarine use by southern sea otters comes primarily from a study conducted in Elkhorn Slough (Tinker et al. 2018), where about 3% of the population resides. From 2013–2018, sea otter density in Elkhorn Slough was 22 sea otters/km², about twice as high as any other area along the coast (Tinker et al. 2018, Hughes et al. 2019). These high densities resulted, in part, from the release of surrogate-reared juvenile sea otters into Elkhorn Slough (Mayer et al. 2019, Becker et al. 2020). Regardless of the source of sea otters, the estuary's capacity to support high densities for a sustained period indicates that it is an important habitat type. The shallow protected waters of Elkhorn Slough can provide all the resources needed, such that all female study animals remained exclusively within the slough and never ventured to the open coast for the duration of the study (Tinker et al. 2018). Males and females used the habitat somewhat differently. While males and females both used the main channel most often for foraging and eelgrass habitats for resting and other non-foraging behaviors, males were found more often in the harbor mouth, main channel, and upper reaches of the slough, whereas females used salt marsh tidal creeks more extensively than males. Habitat use was similar for females whether they had a dependent pup or not. Land habitats such as salt marsh and salt mud were used less often, but areas within 50 m (164 ft) of water were important resting areas, especially for females with pups. The likelihood of use declined with increased distance from water, but sea otters used salt

mud habitats up to 150 m (492 ft) from water and salt marsh habitats up to 200 m (656 ft) from water. During high tides sea otters used the flooded salt marsh for foraging (Tinker et al. 2018). The shallow waters of Elkhorn Slough also appear to provide refuge from shark bites (Nicholson et al. 2018, Mayer et al. 2019).

3.5.5 Haulout areas

Hauling out is a behavioral strategy that conserves energy because of the lower thermoregulatory costs for sea otters on land (Costa and Kooyman 1982; Faurot 1985; Yeates et al. 2007; Thometz et al. 2014). Sea otters sometimes haul out to rest or to recover from sickness or injury. Hauling out is likely especially important in times of nutritional stress, such as for end-lactation-stage females in areas approaching K , who may be in poor body condition and have minimal energy reserves (Tinker et al. 2019b, pp. 86–107). Estuaries provide abundant opportunities for hauling out because land is easy to access at most tidal heights. Sea otters in Elkhorn Slough, particularly females with pups, frequently haul out on salt marsh banks where regulations ensure minimal or no human disturbance (Eby et al. 2017; Tinker et al. 2018, pp. 16–40). Low offshore rocks and protected beaches may also be used for hauling out. Ogden (1941, p. 8) noted that according to hunting records “in all localities the animals came ashore much more frequently in former days before extensive hunting was pursued.” Although haulout areas are important, they are not essential to sea otters, as other habitat features can serve the same purpose.

3.5.6 Maternal care

Sea otter pups are entirely dependent on maternal care and provisioning from birth to weaning at around 6 months of age. Earlier weaning results in higher pup mortality (Monson et al. 2000b; Staedler 2011).

3.6 Population Needs

Resiliency refers to the ability of a population to withstand stochastic disturbance events. As such it is related to the demographic ability to absorb and bounce back from disturbance and persist at the population or metapopulation scale (Smith et al. 2018). Resiliency is positively related to population size and growth rate and may be influenced by connectivity among populations. We assess the resources, circumstances, and demographics that most influence the resiliency of southern sea otter subpopulations. Stochastic events that affect southern sea otters include storms and warm, nutrient-poor water associated with El Niño-Southern Oscillation (ENSO) events, which can increase pup mortality through mother-pup separation, rip out kelp beds, and delay kelp regrowth (Bell et al. 2015; Edwards 2004; 2019; Nicholson et al. 2018); shark-bites (Tinker et al. 2016); disease outbreaks that can alter benthic invertebrate communities, such as sea star wasting disease (Menge et al. 2016; Miner et al. 2018); and prey recruitment events. A variety of factors regulate southern sea otter subpopulations. These factors may be density-dependent (e.g., per-capita prey abundance, ELS) or density-independent [e.g., shark-bite mortality, harmful algal bloom (HAB) intoxication, protozoal infections, and direct human-caused mortality] (Tinker et al. 2021a). We consider the following population needs in this SSA: abundance, survival, recruitment, and dispersal.

3.7 Species Needs

Redundancy spreads risk among multiple populations or areas to minimize the risk due to large-scale, high-impact (i.e., catastrophic) events, and *representation* is the ability of a species to adapt to changing environmental conditions (Smith et al. 2018). In this analysis, we quantitatively evaluate the possible impacts of one type of catastrophic event, a large-scale oil spill, though other catastrophic events (such as the rapid spread of a novel devastating disease in sea otters or their prey, or dramatic negative system-wide impacts resulting from the crossing of climate-induced tipping points) are also possible. We assess the representation and redundancy of the southern sea otter population as a whole to characterize the viability of the species. We define viability as the ability of the southern sea otter to sustain subpopulations in the wild over time.

4.0 HISTORICAL DISTRIBUTION AND ABUNDANCE

The southern sea otter's historical range limit in the south is thought to have been approximately Punta Abreojos, Baja California, Mexico (Figure 6), because this location marks the southern limit of extensive rocky habitat and coastal upwelling along the eastern North Pacific (Riedman and Estes 1990, p. 73). The extension of warm water northward during ENSO events or the extension of cold, nutrient-rich waters southward during years when the California Current was particularly strong may have influenced this range limit accordingly (Riedman and Estes 1990, p. 73). Sea otters were definitely present at Morro Hermoso (Ogden 1941, pp. 143–145) and Natividad and Cedros Islands (Anthony 1925, pp. 303–304), Baja California, Mexico.

The northern range limit of the southern sea otter has not been fully resolved, as genetic and morphological analyses have produced mixed results regarding whether historical samples from Oregon are more closely related to today's northern sea otters or southern sea otters. Valentine et al. (2008) determined, based on mtDNA haplotypes, that ancient sea otter bones from middens just south of Newport, Oregon, were more closely related to modern southern sea otters than northern sea otters, although they detected some samples with the typical northern sea otter haplotype. In contrast, using nuclear markers, Larson et al. (2012) detected more similarities between ancient Oregon samples and ancient Washington samples than between ancient Oregon samples and ancient California samples. Larson et al. (2012) suggested these conflicting findings may indicate that females in Oregon were derived from southern animals but mated primarily with males dispersing from the north. Wellman et al. (2020) compared mitochondrial genomes of archaeological (older than 1150 BP) and historical (19th century pre-extirpation) Oregon sea otters with those of extant and historical populations from other areas of the species' range and found they clustered primarily with samples from Washington, British Columbia, and Alaska, not California. However, the archaeological samples (n = 20) used by Wellman et al. (2020) were all from sites in northern Oregon, and only one of the historical samples from Oregon (n = 2) was linked to a specific location in Oregon. Interestingly, the historical sample collected from Port Orford in southern Oregon did not cluster closely with these other samples, potentially supporting the idea of a latitudinal cline (Wellman et al. 2020, p. 6).

The idea of a latitudinal cline is also supported by morphological evidence. Lyman (1988) examined sea otter teeth from archaeological sites in Oregon and found they were not



Figure 6. Current and historical range of the southern sea otter. Map Z. Cravens, USFWS.

significantly different in size from the teeth of modern sea otters in California and were generally smaller than the teeth of contemporary sea otters in Alaska. Using skull morphometrics and discriminant function analysis, Wilson et al. (1991) found that ancient sea otters from Oregon were intermediate between the northern and southern subspecies but more similar to northern sea otters. Using larger sample sizes than Lyman (1988), Wellman (2018) examined teeth and long bones (femora and humeri) from modern Alaska, ancient Oregon, and modern California sea otters and found that the Oregon samples shared phenotypic characteristics with both populations, although with several significant differences from modern California.

For the purposes of this SSA, we define the northern range limit as Seal Rock, near present-day Newport, Oregon, as its central location serves as a convenient dividing line between the northern sites and central/southern sites from which ancient/archaeological and historical samples have been obtained and analyzed (Figure 6). However, current evidence suggests that there was likely no distinct boundary. Additional research, including finer-scale sampling, may further resolve this question (Larson et al. 2012, p. 9).

Historical sea otter numbers in present-day Oregon, California, and Baja California are unknown because of the incompleteness of hunting records from the maritime fur trade in these areas, which Ogden (1941) dated from 1784–1848. Evermann (1923, pp. 524–526) estimated that at least 200,000 pelts were taken from California and Baja California from 1786–1868 but did not specify the records on which this estimate was based. However, a sense of the former abundance of sea otters can be obtained from records of the large numbers killed over short periods of time. Ogden (1941, pp. 140–141) reported that one vessel alone took 1,800 pelts in 1803, 4,819 in 1806–1807, and 3,952 in 1810–1811. In a single year at the height of the California trade (1811) more than 9,356 pelts were taken (Ogden 1941, p. 140).

The hunting records also give some idea of the relative abundance of sea otters throughout the historical range. Ogden (1941, p. 6) noted that trade records indicate the existence of a partial break in sea otter habitat between the Strait of Juan de Fuca, at the northern boundary of Washington, and Trinidad, California (encompassing coastal Washington and Oregon and a small portion of northern California), where relatively few sea otters were found. Elliott (1875, p. 56, cited in Ogden 1941, pp. 6–7) reported that 50–100 sea otters were killed each year off a 20-mile stretch of beach near Grays Harbor in central Washington, but that this was more than twice the number obtained annually from the remainder of the Washington and Oregon coastline. In northern California, sea otters were abundant in Trinidad Harbor (Ogden 1941, p. 7). Within San Francisco Bay, sea otters were also abundant and “not only swam around in the bay but frequented the numerous estuaries and even hauled up on the shore” (Ogden 1941, p. 7). They were found near Point San Quentin, the mouths of Petaluma and Sonoma creeks, and in the San Jose, San Mateo, and San Bruno estuaries. According to Choris (1913, pp. 16–18), writing in 1816, “sea otters abound in the harbor [San Francisco Bay] and in the neighboring waters,” enabling Russians established at Bodega Bay to kill “almost two thousand otters every year without trouble.” Ogden (1941, p. 141) notes that in 1814, 100 sea otters were killed in San Francisco Bay in one day, and that from 1823–1824 about 1,200 were killed in the bay, 455 of which were killed within a span of two weeks. Hunting records also frequently mentioned sea otters along the central California coast: off Pillar Point, Point Año Nuevo, Santa Cruz, Point Sur, Cooper’s Point, San Simeon, the coast opposite San Luis Obispo (presumably Morro Bay to

Avila Beach), and Point Conception. Whereas there were relatively few sea otters along the mainland coastline of southern California, there were great numbers off the northern Channel Islands and, to a lesser extent, off Santa Catalina and San Clemente islands. Sea otters off Mexico were even more numerous than in California and were centered in seven bays, with their associated headlands and islands: Todos Santos Bay and Todos Santos Island; Santo Tomás anchorage; Colón Bay; San Quintín Bay; Rosario Bay and San Jerónimo Island; Santa Rosalía Bay; and Sebastián Vizcaíno Bay and Natividad, Cedros, and San Benito Islands. Sea otters were also found around Guadalupe Island and Morro Hermoso (Ogden 1941, pp. 7–8).

Efforts to reconstruct the historical abundance of southern sea otters based on estimates of carrying capacity have been conducted for California and Oregon but not Baja California. Based on rocky- and sandy-bottom areas in California and assumed maximum densities of sea otters in these habitat types, with an added number to account for bays, the California Department of Fish and Game roughly estimated the historical population size as about 16,000 animals (CDFG 1976, p. 100). DeMaster et al. (1996) used a similar method, though based on three benthic habitat types (rocky, sandy, and mixed), to obtain a carrying capacity estimate of 13,513. Laidre et al. (2001) again used a similar approach but with updated equilibrium densities for each benthic habitat type and a Geographic Information Systems (GIS) program to estimate the amount of habitat of each type available to obtain a carrying capacity estimate in California of 15,941 (95% CI 13,538–18,577). Tinker et al. (2021b) used a hierarchical Bayesian state-space model fit to time series of survey data in order to estimate the parameters of a theta-logistic growth model (including K), while accounting for the functional relationships between local abundance and a suite of biotic and abiotic habitat variables. This habitat-specific K analysis resulted in a carrying capacity estimate of 17,226 sea otters in California (95% CrI 9,739–30,087). Kone et al. (2021) applied similar methods to Oregon and estimated carrying capacity as 4,538 (95% CrI 1,742–8,976), with higher abundance and densities expected within the southern region.

5.0 CURRENT CONDITIONS

5.1 Distribution and Abundance

All present-day southern sea otters descended from a small remnant population that survived the fur trade near Bixby Creek in Monterey County, California. The population probably numbered about 50 animals in 1914 (Bryant 1915). Since receiving protection under the International Fur Seal Treaty in 1911 and from the State of California in 1913, southern sea otters have gradually expanded northward and southward along the central California coast. Data on range extent by year through the mid-1980s are summarized by Lubina and Levin (1988). At the time of listing in 1977, the southern sea otter range spanned 293 linear km (182 linear mi) of coastline from Soquel Point near the city of Santa Cruz, Santa Cruz County, to Point San Luis, near Avila Beach, San Luis Obispo County. By 2009, the mainland sea otter range extended about 523 km (325 mi), from the Tunitas Creek mouth near Half Moon Bay, San Mateo County, to Coal Oil Point, Santa Barbara County.⁴ After 2009, the range ends retracted and have remained virtually

⁴ Due to variability in the location of the terminal groups that typically mark the ends of the range, a range limit definition was adopted in 2008 to standardize annual reporting. The definition is based on units along the ATOS (“As The Otter Swims”) line, a linear axis described as a series of points spaced at 500-m intervals along the 10-m

unchanged since. At the time of the 5-year review in 2015, the northern boundary was approximately 2.5 km (1.5 mi) southeast of Pigeon Point, and the southern boundary was 5 km (3 mi) west of Gaviota State Beach (Service 2015, p. 7). In 2018, the northern range extent was still 2.5 km (1.5 mi) southeast of Pigeon Point (Hatfield et al. 2018, p. 9). In 2019, the northern range extent was not calculated due to the lack of aerial surveys north of Pigeon Point. However, the southern boundary was about 4.5 km (2.8 mi) west of Gaviota State Beach (Hatfield et al. 2019, p. 10). A geographically distinct subpopulation resulting from translocation efforts from 1988–1990 exists at San Nicolas Island, Ventura County (Figure 6). Southern sea otters are occasionally found well beyond the limits of the established range. Two dead sea otters found stranded in Humboldt County in northern California were genotyped; one of these, found at Gold Bluffs Beach in 2012, was determined to be a southern sea otter, and the other, found in 2014, was determined to be a northern sea otter (Larson 2021). Individual southern sea otters have also been found as far south as Baja California, Mexico (Schramm et al. 2014, Beichman et al. 2022).

Data on population size have been gathered for more than 50 years. Sea otter numbers increased consistently for most of the 20th century at a rate of about 5–6% per year, except during two periods of decline: the late 1970s and early 1980s due to incidental mortality in a set-net fishery, and the late 1990s, the cause of which was not definitively determined (Bodkin et al. 1999; Estes et al. 2003; Service 2003, p. 4). In 1982, a standardized survey technique was adopted to ensure that subsequent counts were comparable (Estes and Jameson 1988). This survey method involves a shore-based census of approximately 60 percent of the range, with the remainder surveyed from the air. Counts of the mainland range are typically conducted each spring, though they were not completed in 2011 (due to weather) or 2020–2022 (due to COVID-19 restrictions and plane availability). At San Nicolas Island, counts are typically conducted from shore quarterly, with the spring count taken as the official count for the year, though COVID-19 restrictions also limited the number of surveys completed there in recent years. Although the annual count is typically referred to as a “census,” it is acknowledged that not every sea otter is counted; thus, the census is a standardized index of abundance slightly lower than true abundance (Estes and Jameson 1988, Henkel et al. 2014).

The recovery plan recommends using the 3-year running average of total counts as the official metric for monitoring trends to reduce the influence of anomalously high or low counts from any particular year (Service 2003, p. 4). Since termination of the experimental status of the San Nicolas Island sea otter population in 2012 (77 FR 75266; December 19, 2012), the island and mainland counts have been combined to arrive at an annual range-wide index of abundance, which consists of the 3-year running average of the combined spring counts (Figure 7). In 2019, the population index (independent animals plus pups) was 2,962 (2,863 along the mainland and 99 at San Nicolas Island) (Hatfield et al. 2019, pp. 3–5). The 5-year average annual growth rate for the southern sea otter range-wide was essentially flat: 0.12% (-0.13% along the mainland and 9.58% at San Nicolas Island) (Hatfield et al. 2019, p. 3).

depth contour. A value of 0 is arbitrarily assigned to the ATOS point at the southern tip of the Golden Gate Bridge in San Francisco, and then ATOS values increase as one moves south along the coast, with ATOS 1111 corresponding to Point Conception. The mainland range boundaries are defined by combining counts of independent sea otters within a moving window of 10-km stretches of coastline and taking the northern and southern ATOS values, respectively, of the northernmost and southernmost stretches in which at least 5 sea otters were counted for at least 2 consecutive spring surveys during the last 3 years (Hatfield et al. 2019, p. 10).

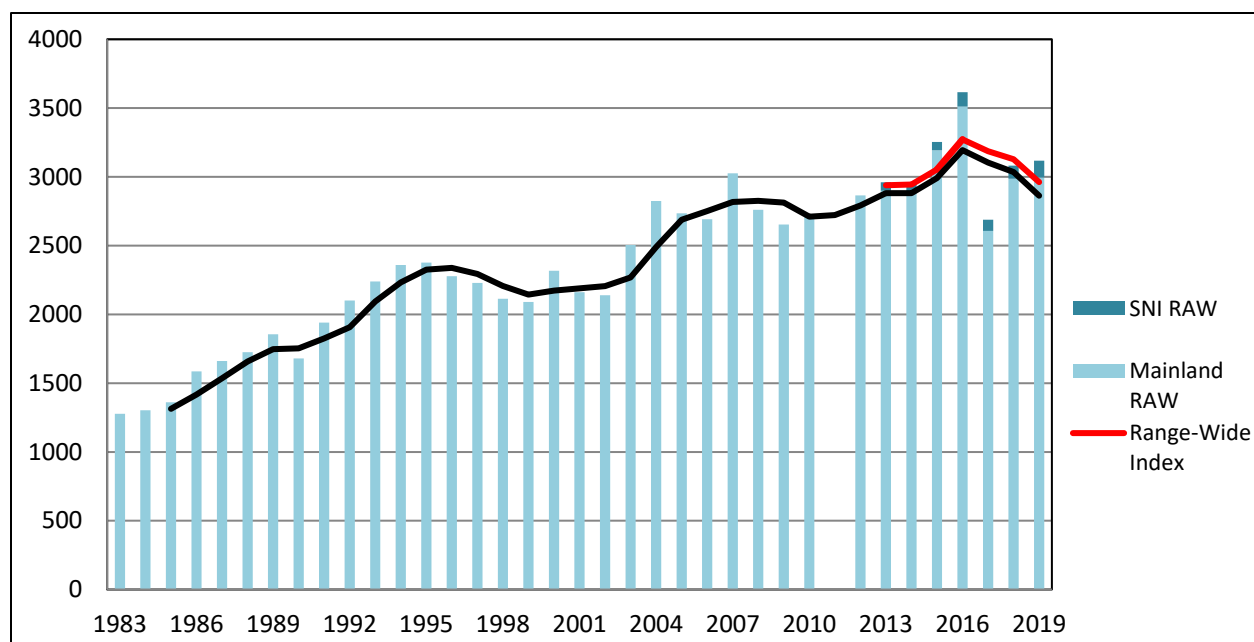


Figure 7. Southern sea otter counts 1983–2019. Bars show raw counts for each year for the central California mainland and San Nicolas Island (SNI), whereas lines represent 3-year running averages. The annual census was not completed in 2011 (due to weather) or 2020–2022 (due to COVID-19 restrictions and plane availability).

Excluding declines that occurred during periods of unusually elevated mortality (such as those caused by gill-net entanglements and white sharks increasingly during the past 20 years; Tinker et al. 2016, Moxley et al. 2019) the difference between the theoretical maximum growth rate of sea otters (between 0.20 and 0.25; Estes 1990, Tinker 2015, Tinker et al. 2019a) and the observed maximum growth rate in California can be explained by two factors: (1) the narrow, linear configuration of habitat within sea otters' usable depth range along the mainland coast of California and (2) the high degree of spatial structuring of sea otter populations (i.e., the short expected dispersal distances of sea otters, especially reproductive females; Gorbics and Bodkin 2001, Tinker et al. 2008b, Tinker 2015). In combination, these factors result in slower range expansion and thus slower overall population growth in California than in other portions of the species' range, such as Alaska and British Columbia, where the habitat consists of bays, islands, and complex matrices of inland channels, or Washington, which is characterized by numerous emergent offshore rocks (in the north) and a broad, shallow sandy shelf (in the south). The narrow, linear configuration of habitat along the California mainland means that only sea otters at the terminal ends have unoccupied habitat within dispersal range, and thus a larger proportion of the population becomes resource limited sooner (Tinker 2015). This difference in habitat configuration results in very different expected population growth rates over the long term (Tinker 2015).

5.2 Current Conditions by Subpopulation

For the purposes of this SSA, we followed Tinker et al. (2021a) in dividing the coastline into discrete areas, which we term *subpopulations* regardless of whether they are extant or extirpated (Figure 8; see p. 8 for our use of the term *subpopulation* in this document). Because habitat analyses were only recently completed for the portions of the historical range in Oregon (Kone et al. 2021) and have not been conducted for Baja California, they were not included in the IPM.

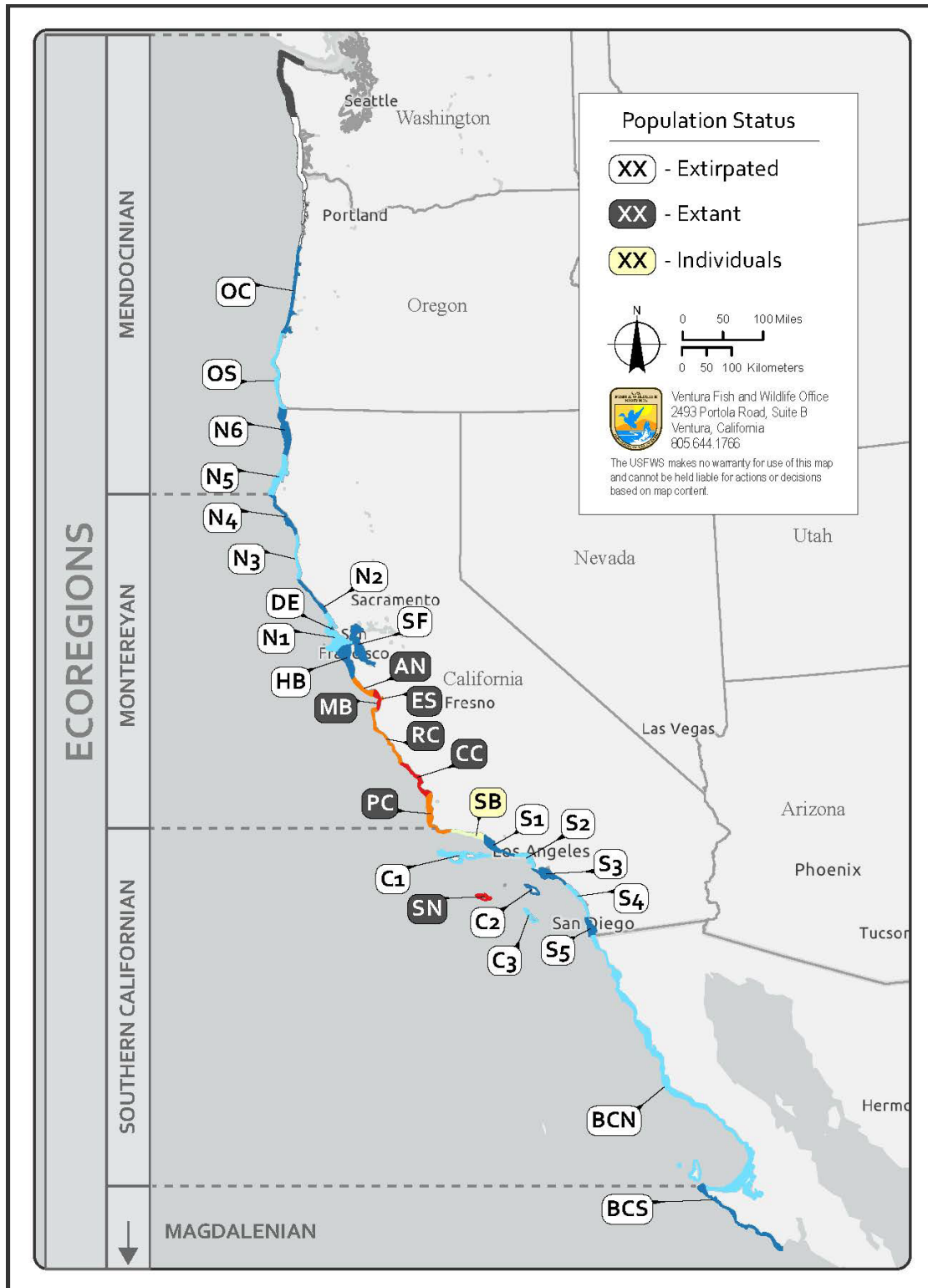


Figure 8. Southern sea otter subpopulations, extant (orange/red polygons, gray labels), extirpated (blue polygons, white labels), and individuals (yellow polygon, yellow label). The Mendocinian ecoregion includes subpopulations OC, OS, and N5–N6. The Montereyan ecoregion includes N1–N4, DE, SF, HB, AN, MB, ES, RC, CC, and PC. The Southern Californian ecoregion includes SB, S1–5, C1–C3, SN, and BCN. The Magdalenian region includes the BCS subpopulation. The occupied portion of the northern sea otter's range in Washington (dark gray polygon) is shown for reference. Map M. Metevier, USFWS.

For completeness, however, we include available information for these areas. We grouped the subpopulations into four ecoregions—Mendocinian, Montereyan, Southern Californian, and Magdalenian—based on the overlap of historical southern sea otter range and biogeographic provinces in the northeast Pacific as identified in Fenberg et al. (2015; see Figure 3). Abundance estimates for extant subpopulations are from the annual census (see section 5.1 for an explanation of how these numbers are obtained). We summarize information for all subpopulations in Table 2.

5.2.1 Mendocinian Ecoregion

The Mendocinian Ecoregion includes the northernmost portions of the southern sea otters' historical range, from about Newport, Oregon, in the north to Cape Mendocino, California, in the south. Jameson (1975) and Kone et al. (2021) divided Oregon into three areas of sea otter habitat, which we identify as subpopulations: North (from the Columbia River to Lincoln City), Central (from Lincoln City to Coos Bay), and South (from Coos Bay to the Oregon-California border). Because we consider the northern range limit of the southern sea otter to be approximately Newport, Oregon, we discuss only the Central and South Oregon subpopulations. All four subpopulations within the Mendocinian Ecoregion are extirpated.

5.2.1.1 Oregon Central (*Extirpated*)

The Oregon Central sea otter subpopulation (coastal segment OC, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or California have been observed off the central Oregon coast. From 2008–2019, 19 sea otter sightings were reported in this region, 16 of which were sightings of a live sea otter (Rice 2019). Two dead sea otters have stranded and been genotyped within the last 10 years. Both were determined to be northern sea otters that had apparently dispersed from the reintroduced population in Washington, although one of these had a very strong Vancouver Island/Prince William Sound genetic signature (Lynch 2020), suggesting possible ancestry in the reintroduced population at Vancouver Island. Available habitat along the outer coast within the 40 m (131 ft) depth contour is 1,175 km² (454 mi²) (Kone et al. 2021, p. 167). Available habitat within estuaries is 78 km² (30 mi²) (Kone et al. 2021, p. 167). The estimated carrying capacity of this coastal segment is 997 (95% CrI 383–1972) (outer coast) and 290 (95% CrI 111–574) (estuaries) sea otters (Kone et al. 2021, p. 167). The combined carrying capacity estimate for this coastal segment is 1,287 (95% CrI 494–2,600; Tinker 2022).

5.2.1.2 Oregon South (*Extirpated*)

The Oregon South sea otter subpopulation (coastal segment OS, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or California have been observed off the southern Oregon coast. From 2008–2019, six sea otter sightings were reported in this region, all of which were sightings of a live sea otter (Rice 2019). Available habitat along the outer coast within the 40 m (131 ft) depth contour is 1,005 km² (388 mi²) (Kone et al. 2021, p. 167). Available habitat within southern Oregon estuaries is 63 km² (24 mi²) (Kone et al. 2021, p. 167). The estimated carrying capacity of this coastal segment is 1551 (95% CrI 595–3068) (outer coast) and 234 (95% CrI 90–

462) (estuaries) sea otters (Kone et al. 2021, p. 167). The combined carrying capacity estimate for this coastal segment is 1,785 (95% CrI 686–3,606; Tinker 2022).

5.2.1.3 Del Norte (Extirpated)

The sea otter subpopulation off Del Norte County (coastal segment N6, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or other areas of California likely transit the Del Norte County coast, but from 2008–2019, no sea otter sightings were reported in this region. Available habitat within the 40 m (131 ft) depth contour is 681 km² (263 mi²). The estimated carrying capacity of this coastal segment is 428 (95% CrI 136–1043) sea otters.

5.2.1.4 Humboldt (Extirpated)

The sea otter population off Humboldt County (coastal segment N5, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or other areas of California have been observed off the Humboldt County coast. From 2008–2019, three sea otter sightings were reported in this region (two live, one dead). As noted in section 5.1, one of these dead sea otters was determined to be a northern sea otter, whereas the other was determined to be a southern sea otter. The live sea otter was not genotyped. Available habitat within the 40 m (131 ft) depth contour is 559 km² (215 mi²). The estimated carrying capacity of this coastal segment is 469 (95% CrI 186–987) sea otters.

5.2.2 Montereyan Ecoregion

The Montereyan Ecoregion includes the area from Cape Mendocino, California, in the north to Point Conception, California, in the south. Of 13 subpopulations, 6 are extant.

5.2.2.1 Mendocino North (Extirpated)

The sea otter subpopulation off northern Mendocino County (coastal segment N4, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or other areas of California have been observed off the northern Mendocino County coast. From 2008–2019, one sighting of a sea otter (live) was reported in this region. Available habitat within the 40 m (131 ft) depth contour is 296 km² (114 mi²). The estimated carrying capacity of this coastal segment is 118 (95% CrI 37–285) sea otters.

5.2.2.2 Mendocino South (Extirpated)

The sea otter subpopulation off southern Mendocino County (coastal segment N3, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from Washington or occupied areas of California have been observed off the southern Mendocino County coast. From 2008–2019, one sighting of a sea otter (live) was reported in this region. Available habitat within the 40 m (131 ft) depth contour is 163

km² (63 mi²). The estimated carrying capacity of this coastal segment is 155 (95% CrI 50–370) sea otters.

5.2.2.3 Sonoma (Extirpated)

The sea otter subpopulation off Sonoma County (coastal segment N2, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters, likely dispersing from occupied areas of California, have been observed off the Sonoma County coast. From 2008–2019, five sea otter sightings were reported in this region (three live, of which two sightings were almost certainly of the same animal, and two dead). Available habitat within the 40 m (131 ft) depth contour is 164 km² (63 mi²). The estimated carrying capacity of this coastal segment is 186 (95% CrI 60–447) sea otters.

5.2.2.4 Marin (Extirpated)

The sea otter subpopulation off Marin County (coastal segment N1, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters, likely dispersing from occupied areas of California, have been observed off the Marin County coast. From 2008–2019, 14 sea otter sightings were reported in this region (9 live, 5 dead). Available habitat within the 40 m (131 ft) depth contour is 513 km² (198 mi²). The estimated carrying capacity of this coastal segment is 742 (95% CrI 257–1709) sea otters.

5.2.2.5 Drake's Estero (Extirpated)

The sea otter subpopulation in Drake's Estero (coastal segment DE, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Dispersing sea otters from occupied areas of California likely enter Drake's Estero occasionally, but from 2008–2019, no sea otter sightings were reported in the estuary (one sea otter seen in Drake's Bay is included in totals for the Marin coastal segment). Available habitat within the 40 m (131 ft) depth contour is 9 km² (3.5 mi²). The estimated carrying capacity of this coastal segment is 62 (95% CrI 40–114) sea otters.

5.2.2.6 San Francisco Bay (Extirpated)

The sea otter subpopulation in San Francisco Bay (coastal segment SF, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters, likely dispersing from occupied areas of California, have been observed in San Francisco Bay. From 2008–2019, eight sea otter sightings were reported in this region (all live, of which one subsequently died and was recovered for necropsy, and of which two were almost certainly the same animal). Available habitat within the 40 m (131 ft) depth contour is 977 km² (377 mi²). The estimated carrying capacity of this coastal segment is 3,503 (95% CrI 689–9913) sea otters.

5.2.2.7 Half Moon Bay (Extirpated)

The sea otter subpopulation off San Mateo County (coastal segment HB, Figure 8), which includes Half Moon Bay, was extirpated during the maritime fur trade, and there is currently no

resident subpopulation. Occasional sea otters, likely dispersing from occupied areas of California, have been observed off the San Mateo County coast. From 2008–2019, 16 sea otter sightings were reported in this region (6 live, 10 dead). Available habitat within the 40 m (131 ft) depth contour is 663 km² (256 mi²). The estimated carrying capacity of this coastal segment is 744 (95% CrI 222–1869) sea otters.

5.2.2.8 Año Nuevo (Extant)

The sea otter subpopulation off Año Nuevo (coastal segment AN, Figure 8) numbers 162 sea otters (3-year average) as of 2019. The 5-year trend is 0.78% per year. Available habitat within the 40 m (131 ft) depth contour is 187 km² (72 mi²). The estimated carrying capacity of this coastal segment is 440 (95% CrI 288–651) sea otters.

5.2.2.9 Monterey Bay (Extant)

The sea otter subpopulation in Monterey Bay (coastal segment MB, Figure 8) numbers 81 sea otters (3-year average) as of 2019. The 5-year trend is strongly negative, an average of -34.4% per year, and likely represents the redistribution (emigration) of sea otters into an adjacent subpopulation. Available habitat within the 40 m (131 ft) depth contour is 161 km² (62 mi²). The estimated carrying capacity of this coastal segment is 248 (95% CrI 156–375) sea otters.

5.2.2.10 Elkhorn Slough (Extant)

The sea otter subpopulation in Elkhorn Slough (coastal segment ES, Figure 8) numbers 104 sea otters (3-year average) as of 2019. The 5-year trend is negative at an average -11.2% per year. The downward trend may reflect the effect of the population reaching local carrying capacity and/or the redistribution of sea otters. Available habitat within the 40 m (131 ft) depth contour is 4.4 km² (1.7 mi²). The estimated carrying capacity of this coastal segment is 115 (95% CrI 35–246) sea otters.

5.2.2.11 Range Center (Extant)

The sea otter subpopulation off the range center (coastal segment RC, Figure 8) numbers 1,642 sea otters (3-year average) as of 2019. The 5-year trend is negative at an average -1.5% per year. Sea otters in this region have been at or near carrying capacity for many years. They benefited from a prey subsidy over the past several years because the onset of wasting disease in 2013 eliminated predatory controls by sea stars on sea urchins and mussels, increasing the availability of these prey types to sea otters; however, the effects of the prey subsidy now appear to be diminishing (Hatfield et al. 2019, p. 3). Available habitat within the 40 m (131 ft) depth contour is 221 km² (85 mi²). The estimated carrying capacity of this coastal segment is 1,189 (95% CrI 864–1584) sea otters.

5.2.2.12 Central Coast (Extant)

The sea otter subpopulation off the central California coast (coastal segment CC, Figure 8) numbers 694 sea otters (3-year average) as of 2019. The 5-year trend is negative at an average -

3.9% per year. Available habitat within the 40 m (131 ft) depth contour is 293 km² (113 mi²). The estimated carrying capacity of this coastal segment is 1,064 (95% CrI 755–1450) sea otters.

5.2.2.13 Point Conception (Extant)

The sea otter subpopulation off northern Santa Barbara County and Point Conception (coastal segment PC, Figure 8) numbers 180 sea otters (3-year average) as of 2019. The 5-year trend is negative at an average -5.5% per year. Available habitat within the 40 m (131 ft) depth contour is 470 km² (181 mi²). The estimated carrying capacity of this coastal segment is 810 (95% CrI 572–1147) sea otters.

5.2.3 Southern Californian Ecoregion

The Southern Californian Ecoregion includes the area from approximately Santa Barbara, California, in the north to Punta Eugenia, Baja California, Mexico, in the south. The number of subpopulations in this region has not been determined because no detailed sea otter habitat analysis has been conducted for portions of the historical range in Mexico. However, 10 subpopulations have been identified within the U.S. portion of this ecoregion. We collapse the northern Baja California portion into one subpopulation, recognizing it must have consisted of many subpopulations historically. We therefore refer to the number of subpopulations in this ecoregion as 11+ to indicate that this number is a minimum. Only 1 of 11+ subpopulations in this ecoregion is extant: the translocated population at San Nicolas Island.

5.2.3.1 Santa Barbara (Individuals)

The sea otter subpopulation off southern Santa Barbara County (coastal section SB, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. However, the 3-year average as of 2019 is 2 sea otters. These are extra-limital sea otters that are considered to be outside the established range. The 5-year trend is 0% per year. Numerous sea otters dispersing from occupied areas of California have been observed off the Santa Barbara County coast. From 2008–2019, approximately 300 sea otter sightings were reported in this region, mostly live, with some groups of approximately 20–30 sea otters reported in the same areas on many consecutive days. The largest numbers of sea otters were seen in 2008 at Coal Oil Point and Naples Reef. The continued presence of multiple sea otters in these areas resulted in a temporary extension of the official range boundary to Coal Oil Point in 2008 (Hatfield and Tinker 2008, p. 2) before it retracted again in 2010 to the vicinity of Gaviota State Beach (Hatfield and Tinker 2010, p. 2), where it has remained since. Available habitat within the 40 m (131 ft) depth contour is 187 km² (72 mi²). The estimated carrying capacity of this coastal segment is 165 (95% CrI 53–398) sea otters.

5.2.3.2 Ventura (Extirpated)

The sea otter subpopulation off Ventura County (coastal section S1, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed off the Ventura coast. From 2008–2019, five sea otter sightings were reported in this region (all live). Available habitat

within the 40 m (131 ft) depth contour is 393 km² (152 mi²). The estimated carrying capacity of this coastal segment is 418 (95% CrI 138–959) sea otters.

5.2.3.3 Los Angeles (Extirpated)

The sea otter subpopulation off Los Angeles County (coastal section S2, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed off the Los Angeles County coast. From 2008–2019, eight sea otter sightings were reported in this region (seven live, one dead). Available habitat within the 40 m (131 ft) depth contour is 176 km² (68 mi²). The estimated carrying capacity of this coastal segment is 200 (95% CrI 67–461) sea otters.

5.2.3.4 Orange (Extirpated)

The sea otter subpopulation off Orange County (coastal section S3, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed off the Orange County coast. From 2008–2019, 13 sea otter sightings were reported in this region (11 live, of which 3 were likely repeat sightings of already-reported sea otters, and 2 dead). Available habitat within the 40 m (131 ft) depth contour is 414 km² (160 mi²). The estimated carrying capacity of this coastal segment is 749 (95% CrI 234–1842) sea otters.

5.2.3.5 San Diego North (Extirpated)

The sea otter subpopulation off northern San Diego County (coastal section S4, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed off the northern San Diego coast. From 2008–2019, one sea otter sighting was reported in this region (dead). Available habitat within the 40 m (131 ft) depth contour is 264 km² (102 mi²). The estimated carrying capacity of this coastal segment is 320 (95% CrI 107–736) sea otters.

5.2.3.6 San Diego South (Extirpated)

The sea otter subpopulation off southern San Diego County (coastal section S5, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed off the southern San Diego coast. From 2008–2019, two sea otter sightings were reported in this region (live). Available habitat within the 40 m (131 ft) depth contour is 281 km² (108 mi²). The estimated carrying capacity of this coastal segment is 701 (95% CrI 223–1684) sea otters.

5.2.3.7 Northern Channel Islands (Extirpated)

The sea otter subpopulation around the northern Channel Islands (coastal section C1, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed in the northern Channel Islands. From 2008–2019, 14 sea otter sightings were reported in this region

(12 live, 2 dead). Available habitat within the 40 m (131 ft) depth contour is 463 km² (179 mi²). The estimated carrying capacity of this coastal segment is 958 (95% CrI 320–2,194) sea otters.

5.2.3.8 Santa Catalina Island (Extirpated)

The sea otter subpopulation around Santa Catalina Island (coastal section C2, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed at Santa Catalina Island. From 2008–2019, one sea otter sighting was reported in this region (live). Available habitat within the 40 m (131 ft) depth contour is 34 km² (13 mi²). The estimated carrying capacity of this coastal segment is 82 (95% CrI 24–201) sea otters.

5.2.3.9 San Clemente Island (Extirpated)

The sea otter subpopulation around San Clemente Island (coastal section C3, Figure 8) was extirpated during the maritime fur trade, and there is currently no resident subpopulation. Occasional sea otters dispersing from occupied areas of California have been observed at San Clemente Island, but from 2008–2019, no sea otter sightings were reported in this region. Available habitat within the 40 m (131 ft) depth contour is 53 km² (20 mi²). The estimated carrying capacity of this coastal segment is 365 (95% CrI 119–857) sea otters.

5.2.3.10 San Nicolas Island (Extant–Translocated)

The sea otter subpopulation at San Nicolas Island (coastal segment SN, Figure 8) numbers 99 sea otters (3-year average) as of 2019. The 5-year trend is positive at 9.6% per year. Available habitat within the 40 m (131 ft) depth contour is 96 km² (37 mi²). The estimated carrying capacity of this coastal segment is 380 (95% CrI 184–694) sea otters.

5.2.3.11 Baja California North (Extirpated)

The sea otter subpopulations off northern Baja California, Mexico (from the USA-Mexico border to Punta Eugenia, Baja California, Mexico; coastal segment BCN, Figure 8), were extirpated during the maritime fur trade, and there are currently no resident subpopulations. We refer to this area as a single subpopulation for simplicity, even though (based on sea otter home range size and the extent of habitat in this area) it must have consisted of many subpopulations historically. Occasional sea otters dispersing from California have been observed off the northern Baja California coastline or around its islands, with several sightings at San Jeronimo Island (Rodríguez-Jaramillo and Gendron 1996; Gallo-Reynoso and Rathbun 1997; Schramm et al. 2014). Two sea otter carcasses from northwest Baja California (a male recovered in 2008 and a female recovered in 2011) were genotyped and are consistent with southern sea otters from California (Schramm et al. 2014, pp. 1266–1268). Available habitat within the 40 m (131 ft) depth contour is 6,521 km² (2,518 mi²). A carrying capacity estimate for the waters off northern Baja California does not currently exist.

5.2.4 Magdalenian Ecoregion

The Magdalenian Ecoregion includes the area from approximately Punta Eugenia, Baja California, Mexico, in the north to the species' range terminus at approximately Punta Abreojos, Baja California, Mexico, in the south. The number of subpopulations in this ecoregion has not been determined because no detailed sea otter habitat analysis has been conducted for portions of the historical range in Mexico. We collapse the southern Baja California portion into one subpopulation, recognizing it likely consisted of more than one subpopulation historically. We therefore refer to the number of subpopulations in this ecoregion as 1+ to indicate that this number is a minimum. Of the 1+ subpopulations in this ecoregion, none is extant.

5.2.4.1 Baja California South (Extirpated)

The sea otter subpopulation off southern Baja California, Mexico (from Punta Eugenia, Baja California, Mexico, in the north to the species' range terminus at approximately Punta Abreojos, Baja California, Mexico, in the south; coastal segment BCS, Figure 8), was extirpated during the maritime fur trade, and there is currently no resident subpopulation. No published reports of sea otter sightings in this portion of the range exist for recent years. However, one private citizen provided a credible report of a sea otter in Estero Coyote, a mangrove lagoon near Punta Abreojos, in 2014 (Haskell 2014). Available habitat within the 40 m (131 ft) depth contour is 865 km² (334 mi²). A carrying capacity estimate for the waters off southern Baja California does not currently exist.

TABLE 2. SOUTHERN SEA OTTER SUBPOPULATIONS. CREDIBLE INTERVALS ARE GIVEN IN PARENTHESES BELOW MEAN ESTIMATED ABUNDANCE. SUBPOPULATIONS WITH FEWER THAN 10 INDIVIDUALS ARE LABELED "INDIVIDUALS."

SUBPOPULATION	ECOREGION	STATUS	HABITAT AREA	ABUNDANCE (2019 3-YEAR AVERAGE)	ESTIMATED CARRYING CAPACITY (95% CRI)
Oregon Central	Mendocinian	Extirpated	1,253 km ² (484 mi ²)	—	1,287 (494–2,600)
Oregon South	Mendocinian	Extirpated	1,068 km ² (412 mi ²)	—	1,785 (686–3,606)
Del Norte (N6)	Mendocinian	Extirpated	681 km ² (263 mi ²)	—	428 (136–1043)
Humboldt (N5)	Mendocinian	Extirpated	559 km ² (215 mi ²)	—	469 (186–987)
Mendocino North (N4)	Montereyan	Extirpated	296 km ² (114 mi ²)	—	118 (37–285)
Mendocino South (N3)	Montereyan	Extirpated	163 km ² (63 mi ²)	—	155 (50–370)
Sonoma (N2)	Montereyan	Extirpated	164 km ² (63 mi ²)	—	186 (60–447)
Marin (N1)	Montereyan	Extirpated	513 km ² (198 mi ²)	—	742 (257–1709)
Drake's Estero (DE)	Montereyan	Extirpated	9 km ² (3.5 mi ²)	—	62 (40–114)

San Francisco Bay (SF)	Montereyan	Extirpated	977 km ² (377 mi ²)	—	3,503 (689–9913)
Half Moon Bay (HB)	Montereyan	Extirpated	663 km ² (256 mi ²)	—	744 (222–1869)
Año Nuevo (AN)	Montereyan	Extant	187 km ² (72 mi ²)	162	440 (288–651)
Monterey Bay (MB)	Montereyan	Extant	161 km ² (62 mi ²)	81	248 (156–375)
Elkhorn Slough (ES)	Montereyan	Extant	4.4 km ² (1.7 mi ²)	104	115 (35–246)
Range Center (RC)	Montereyan	Extant	221 km ² (85 mi ²)	1,642	1,189 (864–1584)
Central Coast (CC)	Montereyan	Extant	293 km ² (113 mi ²)	694	1,064 (755–1450)
Point Conception (PC)	Montereyan	Extant	470 km ² (181 mi ²)	180	810 (572–1147)
Santa Barbara (SB)	Southern Californian	Individuals	187 km ² (72 mi ²)	2	165 (53–398)
Ventura (S1)	Southern Californian	Extirpated	393 km ² (152 mi ²)	—	418 (138–959)
Los Angeles (S2)	Southern Californian	Extirpated	176 km ² (68 mi ²)	—	200 (67–461)
Orange (S3)	Southern Californian	Extirpated	414 km ² (160 mi ²)	—	749 (234–1842)
San Diego North (S4)	Southern Californian	Extirpated	264 km ² (102 mi ²)	—	320 (107–736)
San Diego South (S5)	Southern Californian	Extirpated	281 km ² (108 mi ²)	—	701 (223–1684)
Northern Channel Islands (C1)	Southern Californian	Extirpated	463 km ² (179 mi ²)	—	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	Extirpated	34 km ² (13 mi ²)	—	82 (24–201)
San Clemente Island (C3)	Southern Californian	Extirpated	53 km ² (20 mi ²)	—	365 (119–857)
San Nicolas Island (SN)	Southern Californian	Extant	96 km ² (37 mi ²)	99	380 (184–694)
Baja California North	Southern Californian	Extirpated	6,521 km ² (2,518 mi ²)	—	Undetermined
Baja California South	Magdalenian	Extirpated	865 km ² (334 mi ²)	—	Undetermined

5.3 Current Conditions—Factors Influencing Viability

We use the term “threat” to refer in general to actions or conditions that are known to, or are reasonably likely to, negatively affect individuals of a species. The term “threat” includes actions or conditions that have a direct impact on individuals (direct impacts), as well as those that affect individuals through alteration of their habitat or required resources (stressors). The term “threat”

may encompass—either together or separately—the source of the action or condition or the action or condition itself.

At the time of listing, recognized threats to the southern sea otter included curtailment of its range as a consequence of the maritime fur trade, the threat of a major oil spill, the inadequacy of existing regulatory mechanisms to protect habitat, possible threats from pollution and competition with humans, and a possible loss of genetic diversity (42 FR 2965; January 14, 1977). The primary threat recognized by the first recovery plan (Service 1982, pp. 32–33) was the sea otter's limited range and hence susceptibility to significant population declines due to oil spills, combined with the increased threat of an oil spill in connection with offshore development and the production and transfer of petroleum products. Oil exposure destroys the insulative properties of the pelage and can lead rapidly to hypothermia and death (Costa and Kooyman 1982; Siniff et al. 1982). A lumber spill in central California (which traveled at a rate similar to that of oil slicks observed elsewhere) suggested that an oil spill could spread through most of the range of the southern sea otter in a few weeks (VanBlaricom et al. 1982). Secondary concerns included accidental drownings in monofilament gill nets, intentional killing or injury, contamination of sea otters or their habitat from sources other than oil, destruction and degradation of the sea otter habitat as a result of coastal zone development or other activities, the likelihood of increased conflict with commercial and recreational fisheries, and the lack of precise knowledge concerning the numerical and functional relationships between sea otters, shellfish, finfish, kelp, and other components of nearshore marine communities (Service 1982, pp. 32–33).

The main threats recognized by the revised recovery plan (Service 2003) were habitat degradation, including oil spills and other environmental contaminants, and human take, including shooting, entanglement in fishing gear, and harassment. The plan expressed concern that oil spills, which could occur at any time, could decimate the sea otter population, and that a decline in sea otter abundance was occurring for unknown reasons (possibly infectious disease resulting from increased immune deficiencies or elevated parasite and pathogen exposure, incidental mortality caused by commercial fishing activities, or food resource limitation) (Service 2003, p. viii).

The 5-year review (Service 2015) found that delisting was not warranted because previously recognized threats still existed, and new ones had emerged. These threats included the following: the southern sea otter remained restricted to a small fraction of its historical range; it remained possible that a large oil spill could affect a large proportion of the population; high levels of shark-bite mortality were affecting significant portions of the mainland range and preventing range expansion; food limitation, disease/biotoxin intoxication, mortality in fishing gear, and recreation-related harassment had emerged since listing as factors affecting or potentially affecting the population; and limited genetic diversity indicated the subspecies may not have the capacity to adapt to novel pathogens or new risks associated with climate change.

Factors influencing the survival of southern sea otters vary with sex, age, location, and time. Our evaluation of the threats currently affecting the southern sea otter (Figure 9) is based on a spatially explicit IPM, in which survival rates for a given year and location within California are

computed as the net outcome of multiple competing hazards (Breslow, 1975; Gelfand et al.; Tinker et al. 2021a). The IPM was fit using Bayesian methods to multiple data sets, including a time series of rangewide survey counts (Hatfield et al. 2019), estimated survival rates of tagged animals from various multi-year, telemetry-based population studies (Tinker et al. 2017; Tinker et al. 2019b), and cause-of-death data from comprehensive necropsies conducted for beach-cast carcasses collected from locations throughout the mainland range from 1998 to 2012 (Miller et al. 2020). The result is a stage-structured, metapopulation projection model in which semi-distinct subpopulations are linked by dispersal, and in which annual demographic transitions account for density dependence, environmental stochasticity, and multiple spatially explicit and temporally variable hazards (Tinker et al. 2021a). The IPM allows quantitative

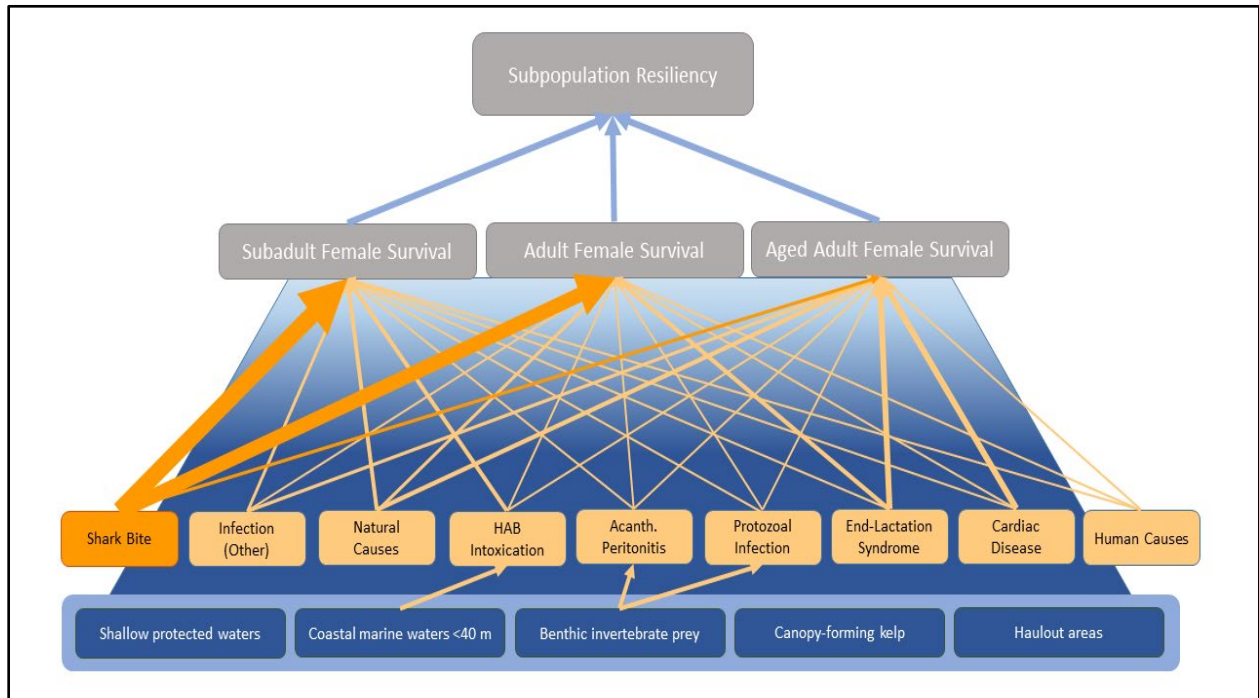


Figure 9. Influence diagram for the southern sea otter. Because female demographics drive population trends, only female age classes are pictured here. Orange indicates a negative influence, whereas blue indicates a positive influence. The result is a stage-structured, metapopulation projection model in which semi-distinct subpopulations are linked by dispersal, and in which annual demographic transitions account for density dependence, environmental stochasticity, and multiple spatially explicit and temporally variable hazards (Tinker et al. 2021a)

evaluation of the population-level impacts of specific hazards currently affecting southern sea otters, as well as projections of future trends, assuming that spatiotemporal variation and covariation among hazards exhibits similar patterns and magnitude in the future as has been observed in the past (Tinker et al. 2021a). The model was fit to multiple datasets, including cause-of-death data for carcasses collected between 1998 and 2012 (Miller et al. 2020; Tinker et al. 2021a, p. 2). Because fresh carcasses are rarely recovered from San Nicolas Island, the hazard analysis does not explicitly address the SN subpopulation. However, the growth rate of the subpopulation and its status with respect to local carrying capacity allow for inferences about whether specific hazards are likely to be important factors influencing population outcomes there. Spatial variation in the relative degree of exposure to different types of hazards can be represented in units corresponding to the log of proportional change in the log hazard ratio

relative to the overall average for that hazard type (Tinker et al. 2021a, pp. 42–50). Positive values represent higher than average levels of exposure to a given hazard. We binned the mean hazard exposure for each area in categories from 1–3, with 1 being highest (mean hazard exposure ≥ 0.1), 2 being moderate ($-0.1 < \text{mean hazard exposure} < 0.1$), and 3 being lowest (mean hazard exposure ≤ -0.1) (Table 3). We describe each of the factors currently affecting subpopulation resiliency below.

5.3.1 Shark Bite

Evidence of attacks by white sharks (*Carcharodon carcharias*) has been documented since the southern sea otter began to recover in the early 20th century (Ames and Morejohn 1980), but the probability of shark-related mortality in sea otters has tripled range-wide in recent decades and increased eightfold in the southern portion of the mainland range (Tinker et al. 2016; Nicholson et al. 2018). White sharks do not appear to target sea otters as prey, and all available evidence suggests they do not consume the sea otters they bite (Tinker et al. 2016, pp. 310, 322; Moxley et al. 2019, p. 2). However, the resulting injuries due to mistaken attacks or investigatory bites are usually fatal to a sea otter.

The reasons for the increase in shark-related mortality of sea otters are unknown and remain the subject of investigation. Potential explanations include an increase in white shark numbers as a result of increases in their preferred marine mammal prey, including elephant seals (*Mirounga angustirostris*) and California sea lions (*Zalophus californianus*); changes in the encounter rate between immature white sharks and sea otters as a result of increases in the immature stage-class following the banning of fishing for white sharks in 1990 and gill-net restrictions put in place in southern California in 1994; and changes in the encounter rate between immature white sharks and sea otters due to episodes of ocean warming, which allow immature white sharks to venture further north for greater portions of the year (Tinker et al. 2016; Moxley et al. 2019).

Regardless of the cause, shark attacks have increased over the last two decades and are now believed to be a barrier to range expansion (Tinker et al. 2016, pp. 322–323). Of all hazards currently affecting southern sea otters, shark-bite mortality has, by far, the greatest population impacts (Tinker et al. 2021a). This hazard affects all age and sex classes, although its relative impact on survival is greatest for subadult females and males, followed closely by adult males, probably because of the tendency of otherwise healthy animals in these age and sex classes to travel longer distances and thus to leave the relative safety of nearshore kelp beds (Nicholson et al. 2018). Hazard rates from shark bite show no relationship to population density (i.e., no density-dependence). Relative exposure to this hazard among subpopulations is highest in AN and PC, followed by MB/ES and CC, and lowest RC (Tinker et al. 2021a; see Figure 8 for subpopulation locations). Note that although the MB and ES subpopulations were combined for the initial hazard analysis, they were subsequently separated in the IPM to reflect the strongly divergent patterns of shark bite risk in these two areas: substantially increasing shark-bite risk in northern Monterey Bay (Tinker et al. 2016; Hatfield et al. 2019, p. 3), as opposed to high sea otter survival rates and no known shark-bite risk in Elkhorn Slough (Tinker et al. 2018, p. 51; Hughes et al. 2019, p. 6). Because of increases in shark-bite risk in northern Monterey Bay subsequent to the period on which the initial hazard analysis was based (1998–2012; Miller et al. 2020), we characterize the hazard rate for the combined MB/ES subpopulation as moderate (as

opposed to low). There is currently no evidence to indicate that shark-bite mortality is an important factor in ES or SN.

5.3.2 End-Lactation Syndrome

By the time a pup is nearing weaning age, daily energy demands on the female are more than twice as high as during times when she is without a pup (Yeates et al. 2007; Thometz et al. 2014; 2016a; 2016b). In food-limited habitats (i.e., areas where sea otters are at or near carrying capacity), such as off Big Sur or the Monterey Peninsula, sea otters spend up to 50% of their time foraging (Tinker et al. 2019b, p. 101). In this situation, females may be physically unable to increase their foraging effort to compensate fully for the increased energy demands imposed by pup-rearing. As a result, they may become emaciated and in poor body condition by the time a pup weans, a condition that increases their susceptibility to other hazards and sometimes leads to death. This condition is now termed end-lactation syndrome (ELS) (Chinn et al. 2016, 2018). This hazard affects aged adult (see section 3.3 for definitions of age classes) females the most, followed by adult females, and shows a strong relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in RC, lower in CC, still lower in PC and AN, and lowest in MB/ES (Tinker et al. 2021a). Because the size of the SN subpopulation is low relative to local carrying capacity, ELS is not expected to be an important factor there.

5.3.3 HAB Intoxication

Harmful algal or cyanobacterial blooms (HAB) can cause acute, subacute, or chronic effects in exposed sea otters (Kreuder et al. 2003; Miller et al. 2010b; 2020). Anthropogenic inputs of nitrogen or phosphorus into coastal watersheds and rising water temperatures may exacerbate these blooms (Mos 2001; Vezie et al. 2002; Kudela et al. 2008). Marine diatoms of the genus *Pseudo-nitzschia* produce domoic acid, a neurotoxin. Domoic acid binds to receptors in the brain and other tissues and can cause memory impairment, problems with spatial navigation, and temporal lobe epilepsy (Miller et al. 2021, p. 2). Domoic acid intoxication of sea otters is also associated with increased risk of cardiac disease, which is more common in older sea otters and can cause death due to heart failure (Kreuder et al. 2003; Kreuder et al. 2005; Miller et al. 2020, p. 19). Freshwater cyanobacteria of the genus *Microcystis* produce another biotoxin, microcystin. Microcystin was been implicated as either a primary or contributing cause in the deaths of 21 sea otters that stranded between 1999 and 2008 (Miller et al. 2010b). This hazard affects adult males most strongly, followed by adult females, and shows no relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in CC, slightly lower in PC, still lower in AN and MB/ES, and lowest in CC (Tinker et al. 2021a). It is not known whether HAB intoxication is an important factor in SN.

5.3.4 Cardiac Disease

As noted under “HAB Intoxication,” cardiac disease is more common in older sea otters and can cause death by heart failure (Kreuder et al. 2003; Kreuder et al. 2005; Miller et al. 2020, pp. 17 and 19). Cardiomyopathy is broadly defined as any myocardial disease that results in cardiac dysfunction, while myocarditis is subtype of cardiomyopathy that is characterized by a significant inflammatory component (Richardson et al. 1996). Risk factors for cardiomyopathy

in sea otters include domoic acid toxicosis, toxoplasmosis and sarcocystosis (see section 5.3.5) (Kreuder et al. 2005; Miller et al. 2020, pp. 17 and 19). Dilated cardiomyopathy (DCM) is associated with dilation and impaired contraction of the ventricles (Richardson et al. 1996). Myocarditis or other chronic or recurrent cardiac pathology is a predisposing factor for the development of DCM, which is considered to be an end-stage expression of chronic cardiac disease (D'Ambrosio et al. 2001; Miller et al. 2020, pp. 17 and 19). Severe, repeated or prolonged domoic acid toxicosis in sea otters is significantly associated with death due to cardiomyopathy, with or without the development of DCM; sea otters with probable DA intoxication were 4.64 times more likely to have died of cardiomyopathy than animals that were not considered to be DA cases ($P < 0.001$) (Miller et al. 2020, p. 17). Sea otters with *Toxoplasma gondii* or *Sarcocystis neurona* infection (see section 5.3.5) were 2.31 and 1.91 times more likely to have fatal cardiomyopathy, respectively (Miller et al. 2020, p. 17). This hazard affects aged adult females the most, followed closely by aged adult males, and shows a moderate relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in CC, slightly lower in RC, still lower in PC and AN, and lowest in MB/ES (Tinker et al. 2021a). It is not known whether cardiac disease is an important factor in SN.

5.3.5 Protozoal Infection

The protozoal parasites *T. gondii* and *S. neurona* can cause severe neurological disease in sea otters. Effects range from subclinical to fatal, including fetal abortion and possible synergistic interactions with other disease processes (Miller et al. 2020, pp. 17–19, 20). In the case of *T. gondii*, felids (wild and domestic cats) are the only definitive or egg-shedding hosts (Dubey et al. 1970; Miller et al. 2002, 2004, 2008). *T. gondii* infection appears to be lifelong, and periods of immune suppression can cause latent infections to reactivate (Pittman and Knoll 2015, pp. 388–389). Environmentally-resistant “eggs” (sporocysts) of the closely related parasite *S. neurona* are shed in the feces of opossums (*Didelphis virginiana* and *D. albiventris*), the only known definitive hosts (Kreuder et al. 2003; Miller et al. 2010a; Miller et al. 2020, p. 18). For both *T. gondii* and *S. neurona*, sea otters are intermediate hosts that are likely infected through consumption of marine invertebrates that have concentrated oocysts or sporocysts from infected water without becoming infected themselves (Arkush et al. 2003; Miller et al. 2008; Miller et al. 2020, p. 18). This hazard affects all age and sex classes, but with a slightly higher impact on subadult females and males, and shows a low relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in CC, slightly lower in PC and AN, still lower in MB/ES, and lowest in RC (Tinker et al. 2021a). It is highly unlikely that protozoal infection is an important factor in SN because neither opossums nor wild or domestic cats currently occur at San Nicolas Island.

5.3.6 Acanthocephalan Peritonitis

Acanthocephalans (thorny-headed worms) infect the intestinal tract of sea otters. Two types of acanthocephalan parasites typically infect sea otters in California. *Corynosoma enhydri* rarely causes disease, whereas *Profilicollis* sp. often burrow through the intestinal wall and enter the abdominal cavity, causing fatal infection (Mayer et al. 2003, Miller et al. 2020). Sea otters are exposed to these parasites by consuming sand crabs (*Emerita analoga*) and mole crabs (*Blepharipoda occidentalis*), which serve as intermediate hosts (Mayer et al. 2003). This hazard

most strongly affects subadult females and males and shows a moderate relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in RC, slightly lower in CC, and lowest in AN, MB/ES, and PC (Tinker et al. 2021a). It is unlikely that acanthocephalan peritonitis is an important factor in SN for the following reasons: the size of the SN subpopulation is low relative to local carrying capacity, making it unnecessary for sea otters to resort to less-preferred prey types such as sand crabs and mole crabs; the benthic substrate at San Nicolas Island is mostly rocky, making it less likely that sea otters would encounter these prey species than in sandy-bottom areas of the central California coast; and sea otters have not been observed consuming these prey species at San Nicolas Island (Tinker et al. 2008a).

5.3.7 Infection (Other)

The Infection (Other) hazard category includes the fungal disease coccidioidomycosis (known as valley fever in humans) as well as viral and bacterial infections other than *S. neurona* and *T. gondii* (Tinker et al. 2021a). Coccidioidomycosis is caused by *Coccidioides* spp. It is the most important fungal disease of southern sea otters and can lead to chronic debilitation and death (Miller et al. 2020, pp. 16). The main route of exposure is inhalation of aerosolized arthroconidia from the environment that can result from disturbed or aerosolized soil. Viral infections are poorly understood in sea otters, and their impacts are relatively unknown. Although they are rarely considered a primary or contributing cause of death, these pathogens can be very hard to detect, and it is certain that fatal cases are missed. As such, viral infections may cause disease or death, sometimes in synergism with other pathogens, but their broader population effects are currently uncharacterized (Miller et al. 2020, pp. 4, 16, 20). Bacterial infections are common, either as a primary cause of death or as a secondary result of trauma, such as injury to the female's nose during mating (Miller et al. 2020, pp. 3–4, 8–9). This hazard category affects all age and sex classes similarly, with potentially stronger effects on subadult and aged adult females than on adult females, and shows a moderate relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in PC, lower in RC and CC, still lower in AN, and lowest in MB/ES (Tinker et al. 2021a). It is unknown whether “Infection (Other)” is an important hazard category in SN.

5.3.8 Natural Causes (Other)

The Natural Causes (Other) hazard category includes a number of pathologies not covered by the other cause-of-death categories, including mating and fighting trauma, gastroenterocolitis, gastrointestinal impaction, neoplasia, and dental disease (Tinker et al. 2021a, p. 16). This set of pathologies most strongly affects subadults and aged adults of both sexes and shows a strong relationship to density dependence. Relative exposure to this set of pathologies among subpopulations is highest in AN, lower in RC, slightly lower in PC and CC, and lowest in MB/ES (Tinker et al. 2021a). It is unlikely that “Natural Causes (Other)” is an important hazard category in SN because the size of the SN subpopulation is low relative to local carrying capacity.

5.3.9 Human Causes (Shootings, Boat Strikes, Entanglements)

Human causes, such as shootings, boat strikes, and entanglements and entrapments in fishing gear or other debris, are a relatively low but persistent source of mortality (Service 2017). This hazard category primarily affects aged adult females and shows no relationship to density dependence. Relative exposure to this hazard among subpopulations is highest in RC, lower in CC, slightly lower in AN and MB/ES, and lowest in PC (Tinker et al. 2021a). It is unknown whether “Human Causes” is an important hazard category in SN. Whereas the number of boat strikes would be expected to be much lower at San Nicolas Island than within harbors along the mainland range because there is much less boat traffic, it is the only area within the southern sea otter’s current range where a spiny lobster fishery regularly occurs. This fishery is pursued throughout southern California waters and presents a risk of entrapment where sea otters co-occur (Hatfield et al. 2011).

5.3.10 Human Causes (Oil Spills)

Oil spills are a form of human-caused pollution resulting from the release of liquid petroleum hydrocarbon into the environment. Sea otters are particularly vulnerable to oil contamination. When sea otters come into contact with oil, it causes their fur to mat, which prevents it from insulating their bodies. Without this natural protection from the frigid water, sea otters can quickly die from hypothermia (Costa and Kooyman 1982, p. 2763–2766; Williams et al. 1988, pp. 2779–2780). The toxicity of oil can also be harmful to sea otters, causing liver and kidney failure as well as severe damage to their lungs and eyes (Kooyman and Costa 1979; Siniff et al. 1982; Lipscomb et al. 1993, 1994; Rebar et al. 1995, 1996). In addition to acute effects, oil can also have chronic and ecosystem effects: the initial exposure may result long-term health impacts, and sea otters may also be exposed repeatedly to residual oil while foraging for prey in contaminated sediments (Peterson et al. 2003, Ballachey et al. 2014, Esler et al. 2018).

The 1989 *Exxon Valdez* Oil Spill (EVOS) in Prince William Sound, Alaska, released 261,905 barrels (41,639,568 liters or 11,000,000 gallons) and killed up to several thousand sea otters (Garrott et al. 1993, Ballachey et al. 1994, Bodkin and Udevitz 1994, DeGange et al. 1994, Eberhardt and Garrott 1997, Garshelis 1997, Garshelis and Estes 1997). In addition to acute effects, mortality rates of prime-age (2–8 years old) individuals were elevated for about two decades following EVOS, with substantial evidence implicating long-term effects of the spill (Monson et al. 2000a, 2011; Bodkin et al. 2002, 2012; Dean et al. 2002; Peterson and Holland-Bartels 2002; Ballachey et al. 2003, 2013, 2014; Peterson et al. 2003, Esler et al. 2018).

A large oil spill has not occurred within the occupied range of the southern sea otter, though several have occurred just to the south and north. The 1969 blowout of Platform A, off Santa Barbara, released 80,000–100,000 barrels into the Santa Barbara Channel and remains the third largest spill in U.S. waters, behind Deepwater Horizon and EVOS (<http://www.countyofsb.org//energy//information//history.asp>). In 1971, two tankers collided under the Golden Gate Bridge, spilling 19,048 barrels (3,028,329 liters/800,000 gallons) of bunker fuel (U.S. Coast Guard 1971). Numerous safety improvements in tanker design and routing have since reduced the chance of such a spill occurring. The Oil Pollution Act of 1990 (33 U.S.C. 2706(b)) phased out single-hull tankers and tank barges in U.S. waters as of January

1, 2015 (though the double hull requirements do not apply to container ships, freighters, cruise ships, or other types of vessels; <https://response.restoration.noaa.gov/about/media/final-farewell-oil-tankers-single-hulls.html>), and a vessel traffic management system was put in place in 2000 (<http://montereybay.noaa.gov/vt/vtexec.html>), reducing the risk of collisions. Additionally offshore oil production is expected to decline gradually, with operators in the Bureau of Ocean Energy and Management (BOEM) Southern California Planning Area estimating they will continue to produce oil and gas for the next 20 years (BOEM 2019, p. 3). Despite these improvements, significant oil spills have continued to occur: the Cosco Busan spill released 219,554 liters (58,000 gallons) of Bunker C/Intermediate Fuel Oil/Heavy Fuel Oil into San Francisco Bay in 2007 (<https://darrp.noaa.gov/oil-spills/mv-cosco-busan>); the rupture of the Plains All-American pipeline released 466,469 liters (123,228 gallons) of crude oil into the Santa Barbara area in 2015, of which approximately 200,627 liters (53,000 gallons) reached the ocean (Refugio Beach Oil Spill Trustees 2021); and the Platform Elly pipeline to Long Beach released at least 112,270 liters (24,696 gallons) of crude oil in 2021 (<https://wildlife.ca.gov/OSPR/NRDA/Pipeline-P00547>).

California is a high-traffic shipping, oil producing, and refining region. Marine vessels (tankers and barges) have historically been and are currently still the primary mode of transporting oil along the west coast, and substantial volumes of crude oil and petroleum products continue to be transported off the California coast from Alaska, from foreign countries, and between California production sources. Large ship traffic increased fourfold from 1992–2012 (Tournadre 2014, p. 7,929). The trend was roughly a 6% increase per year from 1992–2002, and an increasing rate of 10% per year from 2002–2011, with the exception of 2008 to 2009, when the rate remained stable, possibly due to the economic downturn (Tournadre 2014, p. 7,929). Traffic density along the Pacific coast of the contiguous U.S. is highest in southern California along the Santa Barbara TSS and transiting between the Santa Barbara TSS and San Francisco (USCG 2021, p. 8). The Los Angeles/Long Beach and San Francisco Bay harbors include some of the highest volume oil importing ports and refining facilities in the United States (Legislative Analyst’s Office 2022, California Energy Commission 2023). Although these ports are outside the current range of the southern sea otter, collisions or ship groundings off the California coast, or within congested ports or harbor areas, have the potential to occur and to kill large numbers of sea otters if spilled oil enters the range. The largest tankers entering San Francisco Bay carry up to 1.2 million barrels of North Slope crude oil (USCG 2022, section 9000 140, p. 124), and thus the risk of a catastrophic spill still exists (see also Appendix, this document).

Despite significant advances in techniques for washing oiled sea otters made during the last 20 years at the CDFW’s Marine Wildlife Veterinary Care and Research Center, a spill of sufficient magnitude to cause population-level effects would overwhelm the capacity of rehabilitators to rescue sea otters and return them to the wild.

5.3.11 Summary of hazards by extant subpopulation

As noted above, the hazards affecting sea otters vary across the occupied range, with shark bites strongly affecting subpopulations at the northern and southern ends of the mainland range and a suite of other hazards affecting subpopulations in its center (Table 3). Some of these hazards (end-lactation syndrome and natural causes [other]) are strongly related to density dependence

(the status of the subpopulation relative to local carrying capacity), whereas others (cardiac disease, acanthocephalan peritonitis, and infection [other]) have a moderate relationship to density dependence. Still others (shark bite, HAB intoxication, protozoal infection, and human causes) have little or no relationship to density dependence (see Figure 6 in Tinker et al. 2021a, p. 19). Oil spills are infrequent but potentially catastrophic and could have impacts across multiple subpopulations.

Importantly, not all hazards have equivalent impacts on population growth. A perturbation analysis using the IPM in which each hazard was reduced by 20% (while holding constant all other cause-specific hazards) demonstrated that shark bite mortality had by far the greatest impact on population growth (see Figure 10 in Tinker et al. 2021a, p. 24).

TABLE 3. RELATIVE EXPOSURE TO HAZARDS/THREATS BY EXTANT SUBPOPULATION. ADAPTED FROM FIGURES 1.2A–1.2H IN TINKER ET AL. 2021A, PP. 42–50.

HAZARD/THREAT	EFFECT OF DENSITY DEPENDENCE ON HAZARD RATES	AN	MB/ES	RC	CC	PC	SN
Shark bite	N	1	2**	3	2	1	3*
End-lactation syndrome	Y	3	3	1	2	3	3*
HAB intoxication	N	2	2	3	1	2	3*
Cardiac disease	M	2	3	2	1	2	3*
Protozoal infection	N	2	2	3	1	2	3*
Acanthocephalan peritonitis	M	3	3	1	1	3	3*
Infection (other)	M	2	3	2	2	2	3*
Natural causes (other)	Y	1	3	2	2	2	3*
Human causes (shootings, entanglements, boat strikes)	N	2	2	1	2	3	3*
Human Causes (oil spills)	N	T	T	T	T	T	T
	Y=strong M=moderate N=none/low	1=highest (mean hazard exposure ≥ 0.1) 2=moderate ($-0.1 < \text{mean hazard exposure} < 0.1$) 3=lowest (mean hazard exposure ≤ -0.1) T=threat exists but is not currently affecting subpopulation **See explanation for ranking in section 5.3.1.					*no necropsy data available, but other data suggest mean hazard rates are low

5.4 Current Conditions—50-Year Projection

The IPM (Tinker et al. 2021a) allowed us to anticipate population growth and range expansion under current conditions (Table 4 and Figures 10–11). Although the continuation of current conditions is unlikely because environmental conditions are expected to change, this projection served as a basis for comparison with the plausible future scenarios presented in section 6.0. We limited our projections to 50 years (2022–2071) because numerous indirect effects and complex ecosystem interactions will influence southern sea otter viability, and available information does not allow us to reliably anticipate trends beyond this time horizon. We ran 1,000 iterations for this and each of the scenarios in section 6.0.

Under this scenario, the mean population size increased over the next 50 years from an estimated 2,975 sea otters in 2022 to 4,362 (80% CrI 2,287–6,699) in 2071 (Figures 10–11). Although the mean trajectory suggested population growth, the large credible interval indicated considerable uncertainty, with plausible outcomes including substantial growth as well as substantial declines. Some range expansion occurred over the 50-year period, resulting in the addition of HB, SB, S1, and C1 as extant subpopulations. In total, the current conditions projection resulted in the following extant subpopulations after 50 years: HB, AN, MB, ES, RC, CC, PC, SB, S1, C1, and SN (Table 4 and Figure 11; see Figure 8 for subpopulation locations).

The strong effect of shark-bite mortality on future population growth and distribution was evident when, as a heuristic exercise, shark-bite mortality rate was reduced by 20% but all other hazards remained at current levels (see Figures 9 and 10 in Tinker et al. 2021a, pp. 23–24). There is no reason to expect a 20% reduction in levels of shark-bite mortality, but the results—a 19% increase in projected abundance after 50 years relative to that based on current conditions—illustrated the strength of this hazard’s influence on population trends (Tinker et al. 2021a, p. 21).

TABLE 4. PROJECTED POPULATION SIZE AND DISTRIBUTION OF SEA OTTERS IN 50 YEARS BASED ON CURRENT CONDITIONS. CREDIBLE INTERVALS (CRI) ARE GIVEN IN PARENTHESES. SUBPOPULATIONS WITH A CRI THAT DOES NOT INCLUDE ZERO ARE CONSIDERED EXTANT. SUBPOPULATIONS WITH A MEAN ABUNDANCE GREATER THAN ZERO BUT WITH A CRI THAT INCLUDES ZERO ARE CONSIDERED UNKNOWN.

SUBPOPULATION	ECOREGION	STATUS	ABUNDANCE IN 2071 (80%CrI)	ESTIMATED CARRYING CAPACITY (95% CrI)
Oregon Central	Mendocinian	Extirpated	—	1,287 (494–2,600)
Oregon South	Mendocinian	Extirpated	—	1,785 (686–3,606)
Del Norte (N6)	Mendocinian	Extirpated	—	428 (136–1,043)
Humboldt (N5)	Mendocinian	Extirpated	—	469 (186–987)
Mendocino North (N4)	Montereyan	Extirpated	—	118 (37–285)
Mendocino South (N3)	Montereyan	Extirpated	—	155 (50–370)
Sonoma	Montereyan	Extirpated	—	186

(N2)				(60–447)
Marin (N1)	Montereyan	Unknown	68 (0–212)	742 (257–1,709)
Drake’s Estero (DE)	Montereyan	Unknown	21 (0–61)	62 (40–114)
San Francisco Bay (SF)	Montereyan	Unknown	232 (0–626)	3,503 (689–9,913)
Half Moon Bay (HB)	Montereyan	Extant	232 (3–506)	744 (222–1,869)
Año Nuevo (AN)	Montereyan	Extant	306 (191–409)	440 (288–651)
Monterey Bay (MB)	Montereyan	Extant	250 (194–300)	248 (156–375)
Elkhorn Slough (ES)	Montereyan	Extant	176 (153–199)	115 (35–246)
Range Center (RC)	Montereyan	Extant	1,170 (933–1,383)	1,189 (864–1,584)
Central Coast (CC)	Montereyan	Extant	523 (137–889)	1,064 (755–1,450)
Point Conception (PC)	Montereyan	Extant	486 (146–767)	810 (572–1,147)
Santa Barbara (SB)	Southern Californian	Extant	131 (17–229)	165 (53–398)
Ventura (S1)	Southern Californian	Extant	76 (12–206)	418 (138–959)
Los Angeles (S2)	Southern Californian	Unknown	2 (0–9)	200 (67–461)
Orange (S3)	Southern Californian	Extirpated	—	749 (234–1,842)
San Diego North (S4)	Southern Californian	Extirpated	—	320 (107–736)
San Diego South (S5)	Southern Californian	Extirpated	—	701 (223–1,684)
Northern Channel Islands (C1)	Southern Californian	Extant	253 (20–629)	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	Unknown	4 (0–11)	82 (24–201)
San Clemente Island (C3)	Southern Californian	Unknown	3 (0–9)	365 (119–857)
San Nicolas Island (SN)	Southern Californian	Extant	427 (372–483)	380 (184–694)
Baja California North	Southern Californian	Extirpated	—	Undetermined
Baja California South	Magdalenian	Extirpated	—	Undetermined
RANGE-WIDE TOTAL			4,362 (2,288–6,699)	21,764 (11,481–39,363) + Baja California

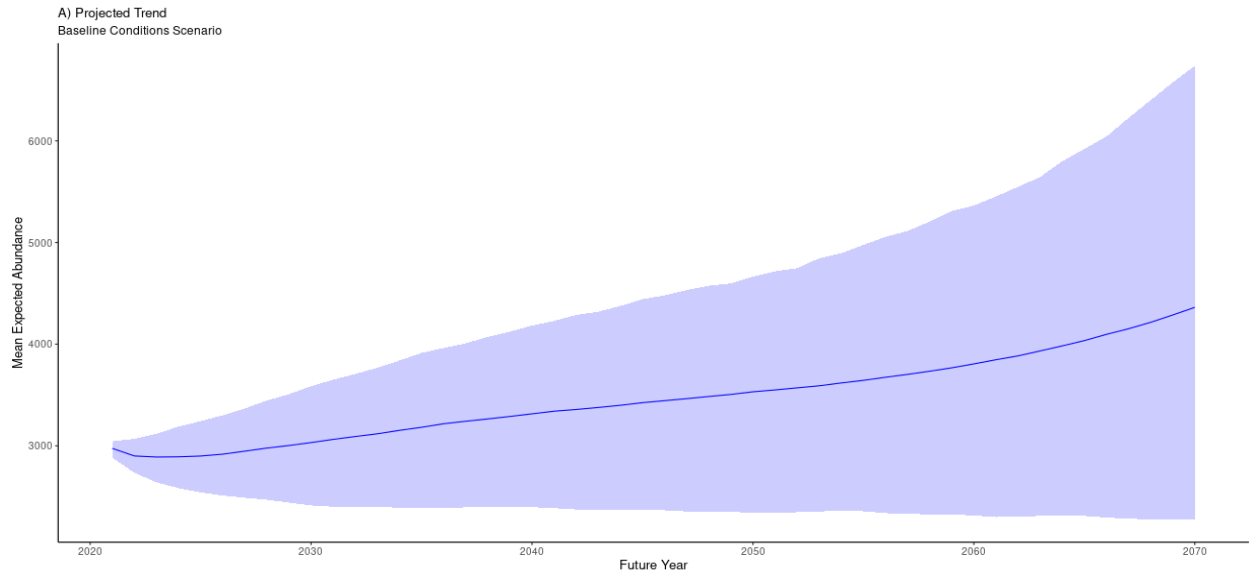


Figure 10. Projected population trend, current conditions, over 50 years. The blue line indicates mean expected abundance. Blue shading indicates the 80% credible interval.

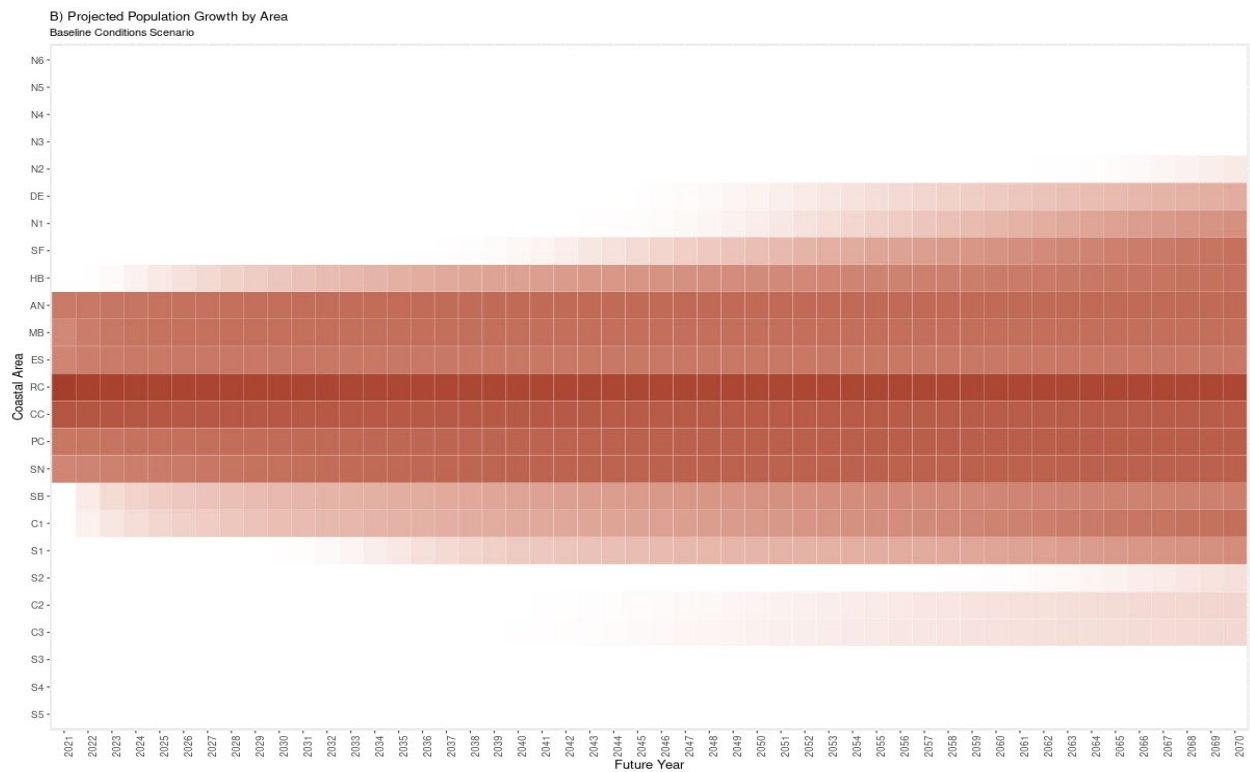


Figure 11. Projected population numbers by area, baseline conditions. Coastal areas within California, oriented north-south, are on the vertical axis. Shading indicates occupation of a coastal segment (darker red indicates higher numbers of sea otters). Sea otter numbers are on a log scale to increase the visibility of low sea otter numbers in newly occupied coastal segments.

5.5 Current Conditions—Summary

5.5.1 Factors Influencing Viability (Resiliency, Redundancy, and Representation)

Resiliency—During the most recent census, the population index was 2,962 (2,863 along the mainland and 99 at San Nicolas Island) (Hatfield et al. 2019, pp. 3–5). The 5-year average annual growth rate for the southern sea otter range-wide was essentially flat: 0.12% (-0.13% along the mainland and 9.58% at San Nicolas Island) (Hatfield et al. 2019, p. 3). Current abundance is far below the estimated carrying capacity of California, 17,226 otters (95% CrI=9,739–30,087) (Tinker et al. 2021b, p. 1), a figure that does not include historical range in Oregon and Baja California, Mexico. Sea otters in the central portion of the mainland range (i.e., the RC subpopulation) are at or near local equilibrium abundance, which increases their susceptibility to natural and anthropogenic stressors and limits the potential for additional population growth in that area. The AN, MB, CC, and PC subpopulations are all well below equilibrium abundance (Table 2), primarily because of moderate or high shark-bite mortality. The exception in the northern region is the subpopulation in Elkhorn Slough (ES), an area that is shallow enough to exclude white sharks. Significant growth of the population will require range expansion into currently unoccupied habitat, but high shark-bite mortality in the areas that lie between densely occupied portions of the range and unoccupied habitat is limiting range expansion, which has not occurred in more than 20 years. As a result, only 7 of 29+ subpopulations are extant, 22+ are extirpated, and one (SB) had 2 individuals (as of the most recent census).

In the current conditions projection, the resiliency of subpopulations after 50 years was variable (Table 4). Several subpopulations (N1, DE, SF, S2, C2, and C3) had credible intervals that included zero; it was unknown whether these subpopulations became extant or remained extirpated by the end of the projection period. High shark bite mortality at the northern and southern peripheries impeded connectivity between more densely occupied subpopulations in central California (MB, ES, and RC) and adjacent unoccupied historical habitat, slowing range expansion. Nevertheless, HB in the north and SB, S1, and C1 in the south became extant during the projection period, though the wide credible interval indicated considerable uncertainty regarding the resiliency of these subpopulations. The SN subpopulation had a mean abundance at the end of the projection period that was greater than mean estimated carrying capacity and thus had strong potential to serve as a source of animals dispersing into other southern subpopulations.

Redundancy—The southern sea otter's range is severely curtailed, and catastrophic events remain a major consideration for the species. Although a major oil spill has not occurred within the southern sea otter's range, major oil spills have occurred immediately to the north and south of the occupied range, and oil spill risk from tankers and other large vessels that transit the California coast remains a primary threat. Such an oil spill, on its own, may not cause extinction, but it would likely severely compromise the ability of the species to withstand additional catastrophic events, such as an additional oil spill, a rapidly spreading novel disease (in sea otters or their prey), or negative system-wide impacts resulting from the crossing of a climate-induced tipping point. Multiple severe widespread events occurring within a short timeframe are highly plausible given the potential for climate-induced changes to affect the nearshore marine ecosystem in numerous ways. The spatial distribution of extant sea otter subpopulations is

mostly restricted to the central California coastline, increasing the species' vulnerability to catastrophic events.

In the current conditions projection, although there were a number of resilient subpopulations after 50 years, they were not sufficiently distributed throughout the southern sea otter's ecological settings or geographical range to spread risk in the case of a catastrophic event. The spatial distribution of extant sea otter subpopulations under this scenario was still mostly restricted to the central California coastline, increasing vulnerability to catastrophic events. Depending on the type, scale, number, and location of these events, the existence of multiple nascent or established subpopulations in the Southern California Bight (SB, S1, C1, and SN) under the current conditions projection could provide some redundancy.

Representation—The southern sea otter has low genetic and environmental diversity. The potentially deleterious alleles detected in southern sea otters and low levels of genetic variation are of concern. Inbred animals are generally less resilient to stress than outbred ones, and populations with low genetic diversity are less able to adapt to environmental change. The simulations conducted by Beichman et al. (2022) demonstrated that, without intervention, recessive genetic load in southern sea otters did not return to pre-fur-trade levels for 400 generations or $\approx 2,800$ years, though it was unlikely to cause extinction on its own. The southern sea otter's diminished evolutionary potential ("capacity to evolve genetically based changes in traits that increase population-level fitness in response to novel or changing environmental conditions"; Forester et al. 2022, p. 1), poses an additional extinction risk. When confronted with a changing environment and suite of stressors, species with low evolutionary potential are more likely to produce maladaptive phenotypes and thus to have reduced fitness, possibly leading to extirpation or extinction (Forester et al., pp. 2–5). Environmental diversity is similarly limited. Existing subpopulations are mostly within the Montereyan ecoregion, with only one subpopulation (SN) extant and one with 2 individuals (SB) in the Southern Californian ecoregion. The Mendocinian and Magdalenian ecoregions are unoccupied. Ecological diversity is also low, with most sea otters occurring on the outer coast in rocky or sandy areas. Only one estuary (ES) is fully reoccupied, and only one island (SN) and no island complexes (which can be defined as a series of islands connected by shallow-water habitat usable by sea otters), have been reoccupied.

In the current conditions projection, because interbreeding with northern sea otters from Washington remained very unlikely, given the limited northward range expansion of southern sea otters, the low genetic diversity in the southern sea otter was not expected to increase during the 50-year projection period. Environmental diversity was also limited. Whereas the historical range spanned four ecoregions, extant subpopulations after 50 years remained restricted to two ecoregions. Within the Mendocinian Ecoregion, all four subpopulations remained extirpated. Within the Montereyan Ecoregion, 7 of 13 subpopulations were extant, 3 were of unknown status, and 3 remained extirpated. Within the Southern Californian ecoregion, 4 of 11+ subpopulations were extant, 3 were of unknown status, and 5+ remained extirpated. No subpopulation was extant in the Magdalenian Ecoregion. The presence of the subpopulation in the Elkhorn Slough estuary (ES), and likely the Morro Bay estuary (which was lumped into the CC subpopulation), contributed to diversity in the ecological settings occupied by southern sea otter subpopulations, which were otherwise situated along the outer coast in rocky or sandy

areas. The status of subpopulations in other estuaries, like Drake's Estero (DE) and the extremely large complex of estuaries that is the San Francisco Bay (SF), was unknown. However, other estuaries in northern California and Oregon remained unoccupied, as did all bays and estuaries in Baja California. Complex island habitat began to be recolonized (C1), but the bay-headland-island complexes of Baja California, which formerly sustained expansive populations of southern sea otters, remained unoccupied.

5.5.2 Uncertainties

Numerous sources of uncertainty affect the preceding analysis. Influences on subpopulations continue to change, whereas the necropsy data with which the hazard analysis in the IPM is parameterized (Miller et al. 2020) date from 1998–2012. Notably, shark-bite mortality may now be a more significant factor in MB than during the necropsy study period (Hatfield et al. 2019, p.3). Additionally, the sampling of stranded carcasses is spatially biased because fresh sea otter carcasses are much less likely to be retrieved along the rugged, remote coastline at the center of the range than in other areas with much higher human populations and greater accessibility.

The issue of carrying capacity is especially complex because equilibrium abundance varies across space and time as a function of prey productivity, local habitat quality, and changing environmental conditions (Tinker et al. 2021b). Estimates of carrying capacity for the identified subpopulations (both extant and extirpated) (Tinker et al. 2021b) have large credible intervals, and the carrying capacities of substantial portions of the historical range (those in Baja California, Mexico) have not been estimated. Further, the methods by which carrying capacities are determined in Tinker et al. (2021b) are substantially different from those in the IPM (Tinker et al. 2021a), although the former estimates feed into the IPM for determining density-dependent hazards. Because the IPM is a stage-specific hazards model and not a simple logistic model as in Tinker et al. (2021b), the realized equilibrium is an emergent property rather than a forced parameter. Additionally, the section-specific estimates of carrying capacity in Tinker et al. (2021b) do not incorporate the increase in shark bite mortality after 1999 because that is treated as a separate, density-independent process ($\delta_{s,t}$ in equation 6 of Tinker et al. 2021b, p. 8). Thus, the numbers given in the “Estimated Carrying Capacity” column in Table 4 (and Tables 6, 7, and 8), which are from Tinker et al. (2021b), represent what carrying capacity is projected to be if shark bite mortality had remained at the relatively low values observed before 1999. In contrast, the realized equilibrium abundance values from IPM projections for each subpopulation (the “Abundance in 2071” column in Tables 4, 6, 7, and 8) are determined by the net effects of all hazards; thus, for areas that have experienced large increases in shark bite mortality in recent decades (e.g., AN, CC, and PC) equilibrium abundance as determined by the IPM will be considerably lower than the theoretical potential carrying capacity given in Tinker et al. (2021b). Additionally, the net effects of immigration/emigration affect the realized equilibrium densities, so for subpopulations that receive immigrants from adjacent larger populations, realized dynamic equilibriums can be slightly higher than the theoretical potential carrying capacity, especially where local shark bite mortality is projected to be low (e.g., ES and SN). Finally, the estimates of carrying capacity for currently unoccupied subpopulations present only a partial picture, as they do not account for hazards caused by intensive human activities in some areas that may be detrimental to sea otters.

The IPM simulations do incorporate both demographic and environmental process error (stochasticity) and parameter uncertainty in carrying capacity estimates and all other model parameters, which together contribute to the wide credible intervals around projected abundance estimates.

6.0 POTENTIAL FUTURE CONDITIONS

In order to anticipate potential future conditions, we forecasted the species' response to plausible future scenarios of environmental conditions. We based the three plausible future scenarios on IPCC RCP 4.5 (with no oil spill), RCP 8.5, and RCP 8.5 with a large oil spill. The scenarios were designed to explore the response of the species to environmental stressors and represent the range of plausible scenarios given the information currently available. To construct these scenarios, we relied on a number of assumptions, which we make explicit (Table 5), to anticipate how hazards could change under the scenarios. As under the current conditions projection, we projected scenarios 50 years and ran 1,000 iterations for each scenario.

6.1 Mechanisms Driving Changes in Climate-Related Hazards under the RCP 4.5 and RCP 8.5 Scenarios

Ongoing climate change includes complex, interrelated changes in temperatures, weather patterns, and ocean conditions. Changing physical conditions, such as increasing temperatures, hypoxia, or ocean acidification will have direct effects on some marine species. Other species will be affected by changes in the abundance, distribution, or other characteristics of their competitors or prey species. Changes in the timing of seasonal events may lead to mismatches in the timing of consumers' life history requirements with their habitat conditions (including prey availability as well as physical conditions; Mackas et al. 2007, p. 249). The combination of these effects is likely to cause changes in community dynamics, such as competitive interactions and predator-prey relationships, which are often complex and difficult to predict (Busch et al. 2013, pp. 827–831).

Climate change is not expected to affect sea otters significantly through direct pathways. However, it has the potential to exacerbate, through numerous indirect pathways, hazards that are currently affecting southern sea otters, as well as to introduce novel stressors. By the end of the 21st century, average air temperatures in California are expected to increase 2–4 °C (3.6–7.2 °F; RCP 4.5 scenario) or 4–7 °C (7.2–14.4 °F; RCP 8.5 scenario) (Pierce et al. 2018, p. iv). Accompanying this increase in air temperature will be an increase in mean sea surface temperatures (SST) and the potential for increased ocean temperature extremes, including marine heatwaves (Alexander et al. 2018).

Warmer ocean temperatures can exacerbate the susceptibility of marine organisms to disease (Harvell et al. 2002), as has been seen with withering syndrome in abalone (Crosson and Friedman 2018) and wasting disease in sea stars, which caused a trophic cascade that rippled through marine food webs (Burt et al. 2018, Harvell et al. 2019). The effects on sea otters of disease-related declines of invertebrates may be positive, negative, or mixed. Short-term effects may also differ from long-term effects. Sea otters in areas of central California that were previously at or near carrying capacity have benefited from a prey subsidy over the past several

years because the onset of wasting disease in 2013 eliminated predatory controls by sea stars on sea urchins and mussels, increasing their availability to sea otters (Hatfield et al. 2019, p. 3). Evidence from Alaska suggests that warmer waters can increase the caloric content of Aleutian green sea urchins (Weitzman 2020, pp. 55–56), which may also have benefited sea otters in central California. However, as of 2019, the effects of increased prey availability appeared to be diminishing (Hatfield et al. 2019, p. 3). A scenario in which disease devastated populations of prey species important to sea otters is also entirely plausible. In such a case, effects on sea otters could be strongly negative. Changes in the relative abundance of benthic invertebrates due to disease could have other indirect effects on sea otters. For instance, losses of kelp canopy cover associated with the spread of sea urchin barrens (following the loss of sea stars) may ultimately harm sea otters through negative effects on other kelp-dependent species that sea otters prey on and the loss of kelp canopy cover for resting and protection from white sharks (Nicholson et al. 2018).

Marine heat has the potential to affect canopy-forming kelps not just through trophic cascades but also directly, although the effect of thermal stress may be difficult to separate from the influence of other co-occurring stressors. During the marine heatwave of 2013–2015, which began to affect the California coast in 2014, sea surface temperatures 2.5 °C (4.5 °F) warmer than normal persisted for 226 days (Rogers-Bennett and Catton 2019, p. 1). During this time, thermal stress, low nutrient availability, and the release of purple sea urchin populations from predation by sea stars combined to reduce the coverage of bull kelp (*Nereocystis luetkeana*) along the coastline north of San Francisco to the Oregon border from a historical maximum of 50 km² to <2 km² (19 mi² to <0.8 mi²; Rogers-Bennett and Catton 2019, p. 2). Kelp beds in northern California had not recovered substantially as of 2020 (Rogers-Bennett and Catton 2019, p. 2; Garcia-Reyes et al. 2022, p. 7). Sea otters are currently absent from this portion of the coastline, so sea otter predation on sea urchins was not able to buffer the effects of the loss of sea star predation on sea urchins. During the same marine heatwave, giant kelp (*Macrocystis pyrifera*) in southern California and Baja California, Mexico, areas from which sea otters are also almost entirely absent, exhibited variable responses, with resistance (the ability to withstand increased temperatures without major losses of canopy cover) most closely associated with absolute SST and recovery (the return of kelp after initial losses) apparently driven by local environmental and biotic factors (Cavanaugh et al. 2019, p. 1).

Increasing SST will likely escalate the biomass, frequency, and duration of harmful algal and cyanobacterial blooms (Lopez et al. 2008; Wells et al. 2015; Cayole et al. 2016; McCabe et al. 2016). These blooms produce biotoxins, such as domoic acid and microcystin, that can debilitate or kill sea otters (Miller et al. 2010b; 2017, p. 31). The 2013–2015 marine heatwave caused an unprecedented harmful algal bloom (Cavole et al. 2016). The bloom stretched from the Aleutian Islands to Southern California and persisted for 6 months instead of a few weeks, as is typical. It produced extremely high concentrations of domoic acid; concentrations in Monterey Bay reached 10–30 times what is normally considered to be a high level (1,000 Ng L⁻¹) (NOAA Climate 2015, cited in Cavole et al. 2016).

Extreme daily precipitation events are expected to increase 5–15% (RCP 4.5 scenario) to 15–20% (RCP 8.5 scenario) (Pierce et al. 2018, p. iv), potentially overwhelming natural and human methods of filtration of storm runoff. Increased storm runoff may facilitate the transmission of

land-borne pathogens, such as *T. gondii* and *S. neurona*, to the nearshore marine environment (Van Wormer et al. 2016). Extensive loss of Pacific coastal wetlands due to submergence as a result of sea level rise (Thorne et al. 2018) will reduce their ability to filter pathogens before these pathogens enter the ocean (Shapiro et al. 2010). Climate-mediated range shifts in hosts and pathogens may bring novel pathogens into the marine environment (Harvell et al. 1999), and temperature increases may also affect the evolution of pathogens (Yan and Wu 2011).

Ocean acidification is expected to affect a broad range of calcifying marine organisms through decreases in survival, calcification, growth, development, and abundance (Kroeker et al. 2014). Mollusks and the larval stages of echinoderms are among the taxonomic groups most negatively affected (Kroeker et al. 2014, p. 1890). Chemical changes associated with ocean acidification interfere with shell development or maintenance in sea snails and marine bivalves (Busch et al. 2014, pp. 5, 8; Waldbusser et al. 2015, pp. 273–278). These effects can be exacerbated by hypoxic conditions (i.e., inadequate oxygen supply at the tissue level; Gobler et al. 2014, p. 5) and either offset or intensified by high temperatures (Kroeker et al. 2014, p. 1885). Realistic predictions of ecosystem response to future conditions should ideally include changes in temperature and dissolved oxygen that are likely to co-occur with changes in pH (Busch and McElhany 2016, p. 23). Marshall et al. (2017) modeled ecosystem dynamics in the California Current 50 years into the future resulting from an anticipated 0.2 unit decrease in pH (equivalent to a 55% increase in acidity) during the summer upwelling season (when pH is typically lowest), which is expected under the IPCC RCP 8.5 scenario. The strongest direct negative effects were on epibenthic invertebrates such as crabs, shrimps, benthic grazers, benthic detritivores, and bivalves, with indirect negative effects on other species, such as Dungeness crabs, that consume species affected by ocean acidification (Marshall et al. 2017, p. 1525). These groups comprise most of the prey species of sea otters in California. However, nearshore sea urchins, which are also important prey for sea otters, exhibited a strong positive response in the ecosystem model projections because they are fed upon by fish groups that experienced indirect negative effects (Marshall et al. 2017, p. 1534).

In summary, climate change is expected to affect sea otters negatively through the following indirect pathways: (1) by increasing exposure to harmful algal and cyanobacterial blooms, (2) by increasing susceptibility to white shark bites through losses in kelp canopy cover and increases in thermal conditions favorable to subadult white sharks, (3) by increasing exposure to land-borne and/or novel pathogens, and (4) by decreasing prey availability generally (though with possible increases in important prey species such as nearshore sea urchins). Given the complexity of interactions and the numerous sources of uncertainty, we modeled the plausible future scenarios under RCP 4.5 and RCP 8.5, respectively, as a marginal (10%) and substantial (30%) increase in the hazard rates of shark bite, protozoal infection, and infection (other) (Table 5). To simulate decreases in prey availability as a result of ocean acidification and warming, we adjusted average density at carrying capacity downward (10% decrease under RCP 4.5, 30% decrease under RCP 8.5). Other hazard rates were affected by this downward adjustment in carrying capacity: cardiac disease and natural causes (other), which are moderately density dependent, and ELS and acanthocephalan peritonitis, which are strongly density dependent (see Figure 6 in Tinker et. al. 2021a, p. 19). We modeled HAB intoxication (and thereby further influenced rates of cardiac disease) by adjusting upward the frequency and severity of harmful algal blooms (10% increase under RCP 4.5, 30% increase under RCP 8.5). We did not alter the hazard rates from the current

conditions projection for natural causes (other) or human causes (boat strikes, shootings, entanglements) under these scenarios.

6.2 Incorporating Effects of a Large Oil Spill into the RCP 8.5 Scenario

The IPM as described in Tinker et al (2021a) was subsequently modified to allow for the simulation of stochastic oil spill events in conjunction with other environmental changes. A detailed description of the methods used to model oil spills in the IPM is included here as an Appendix. In brief, numerous simulations of oil spill events within the sea otter range were conducted using an oil spill simulation tool developed by the National Oceanic and Atmospheric Administration, the results of which were spatially intersected with current and/or future distributions of sea otters and used to estimate mortality from acute and chronic exposures. The results of these simulations were incorporated into the IPM web tool, allowing users to select from a representative set of high-risk spill locations (near San Francisco or near Point Conception), spill sizes (medium or large), and seasonal wind and current conditions (winter or summer). Medium and large spill sizes were based on the U.S. Coast Guard's spill size categorization of Worst Case Discharge, Maximum Most Probable Discharge, and Average Most Probable Discharges for these regions (Appendix, p. 4).

We used the IPM web tool to assess the potential outcome of a large oil spill in conjunction with the climate-related changes expected under RCP 8.5. Specifically, we evaluated the scenario of a large oil spill (10 million gallons) occurring near San Francisco during the summer of 2037 (1/3 of the way through the 50-year projection period). The area outside San Francisco Bay is regularly transited by Very Large Crude Carriers, as defined by the U.S. Coast Guard, which have a full load discharge of approximately 80 million gallons (Appendix, p. 4). We selected this scenario for two reasons. First, it is a reasonably likely worst-case scenario. Although an oil spill could have much more catastrophic effects than those presented here (e.g., if the spill were larger, if it occurred earlier in the 50-year period, if wind and current conditions drove oil directly into densely occupied areas of the sea otter range, if more than one large oil spill occurred within the range during the 50-year period, etc.), in our judgment the selected scenario represents a reasonable balancing of what is plausible and what is likely. Second, the Appendix describes this same scenario in detail, though without the climate-related changes described above. Readers wishing to understand the influence of the oil-spill effects separately from the climate-driven effects may thus compare our results with those presented in the Appendix.

TABLE 5. ASSUMPTIONS REGARDING MECHANISMS DRIVING CHANGES IN HAZARDS UNDER PLAUSIBLE FUTURE SCENARIOS

STRESSOR	RCP 4.5	RCP 8.5	RCP 8.5 + LARGE OIL SPILL
Shark bite	Increases marginally (10%) range-wide as a result of increasing frequency of ENSO events (and associated storms) and ocean warming. These factors will decrease kelp canopy coverage marginally, increasing sea otters' vulnerability to white shark bites in areas with sufficient depth to accommodate white sharks; warm waters allow marginally increased intrusion of subadult white sharks into areas north of Point Conception. We accounted for increases in this hazard <i>directly</i> by adjusting shark bite mortality upward by 10%.	Increases substantially (30%) range-wide as a result of increasing frequency of ENSO events (and associated storms) and ocean warming. These factors will decrease kelp canopy coverage substantially, increasing sea otters' vulnerability to white shark bites in areas with sufficient depth to accommodate white sharks; warm waters allow substantially increased intrusion of subadult white sharks into areas north of Point Conception. We accounted for increases in this hazard <i>directly</i> by adjusting shark bite mortality upward by 30%.	Increases substantially (30%) range-wide as a result of increasing frequency of ENSO events (and associated storms) and ocean warming. These factors will decrease kelp canopy coverage substantially, increasing sea otters' vulnerability to white shark bites in areas with sufficient depth to accommodate white sharks; warm waters allow substantially increased intrusion of subadult white sharks into areas north of Point Conception. We accounted for increases in this hazard <i>directly</i> by adjusting shark bite mortality upward by 30%.
ELS	Increases marginally with decrease in prey availability due to ocean acidification and warming. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 10%.*	Increases substantially with decrease in prey availability due to ocean acidification and warming. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 30%.*	Increases substantially with decrease in prey availability due to ocean acidification and warming. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 30%.*
HAB Intoxication	Increases marginally with warming ocean waters. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 10%, respectively.**	Increases substantially with warming ocean waters. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 30%, respectively.**	Increases substantially with warming ocean waters. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 30%, respectively.**
Cardiac disease	Increases in association with marginally increased chronic HAB exposure. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 10%, respectively.**	Increases in association with substantially increased chronic HAB exposure. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 30%, respectively.**	Increases in association with substantially increased chronic HAB exposure. We accounted for increases in this hazard <i>indirectly</i> by adjusting the frequency and severity of domoic acid events upward by 30%, respectively.**

Protozoal infection	Increases marginally (10%) with increased severity of rainfall events and reduced buffers between human development and the ocean due to sea level rise. We accounted for increases in this hazard <i>directly</i> by adjusting protozoal infection upward by 10%.	Increases substantially (30%) with increased severity of rainfall events and reduced buffers between human development and the ocean due to sea level rise. We accounted for increases in this hazard <i>directly</i> by adjusting protozoal infection upward by 30%.	Increases substantially (30%) with increased severity of rainfall events and reduced buffers between human development and the ocean due to sea level rise. We accounted for increases in this hazard <i>directly</i> by adjusting protozoal infection upward by 30%.
Acanthocephalan peritonitis	Increases marginally with decrease in preferred prey availability due to ocean acidification. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 10%.*	Increases substantially with decrease in preferred prey availability due to ocean acidification. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 30%.*	Increases substantially with decrease in preferred prey availability due to ocean acidification. We accounted for increases in this hazard <i>indirectly</i> by adjusting average density at K downward by 30%.*
Infection (other)	Increases marginally (10%) with climate-driven changes in terrestrial species' ranges and possible introduction of novel pathogens into the nearshore environment. We accounted for increases in this hazard <i>directly</i> by adjusting infection (other) upward by 10%.	Increases substantially (30%) with climate-driven changes in terrestrial species' ranges and possible introduction of novel pathogens into the nearshore environment. We accounted for increases in this hazard <i>directly</i> by adjusting infection (other) upward by 30%.	Increases substantially (30%) with climate-driven changes in terrestrial species' ranges and possible introduction of novel pathogens into the nearshore environment. We accounted for increases in this hazard <i>directly</i> by adjusting infection (other) upward by 30%.
Natural causes (other)	Same as current conditions projection	Same as current conditions projection	Same as current conditions projection
Human causes (boat strikes, shootings, entanglements)	Same as current conditions projection	Same as current conditions projection	Same as current conditions projection
Human causes (large oil spill)	Reduced risk of oil spill due to reduced fossil fuel use; no oil spill risk is modeled in this scenario.	Continued reliance on fossil fuels perpetuates the risk of a large oil spill, but a large oil spill does not occur within the projection period.	Continued reliance on fossil fuels perpetuates the risk of a large oil spill. The modeled spill is large (10 million gallons) and occurs near San Francisco during the summer of 2037.
*Proportional changes to carrying capacity (average local density at K) affect all causes of death in the IPM that are moderately (cardiac disease, natural causes) or strongly (ELS, acanthocephalan peritonitis) density-dependent. Carrying capacity was adjusted downward by the stated amount to affect the associated hazard rates. **Proportional changes to the frequency and intensity of domoic acid events affect the causes of death in the IPM that vary with domoic acid exposure (HAB intoxication, cardiac disease). The frequency and intensity of domoic acid events were adjusted upward by the stated amount to affect the associated hazard rates.			

6.3 Plausible Future Scenarios: Results

6.3.1 RCP 4.5 Scenario

For the RCP 4.5 Scenario, we adjusted inputs to the IPM to model the plausible cumulative effects of low-intermediate climate change, as described in section 6.1 and summarized in Table 5. Specifically, we increased the hazard rates of shark bite, protozoal infection, and infection (other) directly by 10%. To simulate decreases in prey availability as a result of ocean acidification and warming, we adjusted average density at carrying capacity downward by 10%. Other hazard rates were affected by this downward adjustment in carrying capacity: cardiac disease and natural causes (other), which are moderately density dependent, and ELS and acanthocephalan peritonitis, which are strongly density dependent (see Figure 6 in Tinker et. al. 2021a, p. 19). We modeled HAB intoxication (and thereby further influenced rates of cardiac disease) by increasing the frequency and severity of harmful algal blooms by 10%. We did not alter the hazard rates from the current conditions projection for natural causes (other) or human causes (boat strikes, shootings, entanglements).

Under this scenario, the mean expected abundance of sea otters in 50 years was 3,454 (1,849–5,408) (Table 6 and Figures 12–13), 16% greater than the estimated starting population size of 2,975 independent animals in 2022. However, there was considerable uncertainty; although the mean population projection suggested modest population growth, plausible outcomes included more substantial growth as well as substantial declines (Figure 12). The following subpopulations were extant after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN (Table 6 and Figure 13; see Figure 8 for subpopulation locations), indicating no range expansion to the north but some range expansion to the south relative to current conditions in 2022.

TABLE 6. PROJECTED POPULATION SIZE AND DISTRIBUTION OF SEA OTTERS IN 50 YEARS, RCP 4.5 SCENARIO. CREDIBLE INTERVALS (CRI) ARE GIVEN IN PARENTHESES. SUBPOPULATIONS WITH A CRI THAT DOES NOT INCLUDE ZERO ARE CONSIDERED EXTANT. SUBPOPULATIONS WITH A MEAN ABUNDANCE GREATER THAN ZERO BUT WITH A CRI THAT INCLUDES ZERO ARE CONSIDERED UNKNOWN.

SUBPOPULATION	ECOREGION	STATUS	ABUNDANCE IN 2071 (80% CRI)	ESTIMATED CARRYING CAPACITY (95% CRI)
Oregon Central	Mendocinian	Extirpated	—	1287 (494–2600)
Oregon South	Mendocinian	Extirpated	—	1785 (686–3606)
Del Norte (N6)	Mendocinian	Extirpated	—	428 (136–1043)
Humboldt (N5)	Mendocinian	Extirpated	—	469 (186–987)
Mendocino North (N4)	Montereyan	Extirpated	—	118 (37–285)
Mendocino South (N3)	Montereyan	Extirpated	—	155 (50–370)
Sonoma (N2)	Montereyan	Extirpated	—	186 (60–447)
Marin	Montereyan	Unknown	35	742

(N1)			(0–125)	(257–1,709)
Drake's Estero (DE)	Montereyan	Unknown	12 (0–41)	62 (40–114)
San Francisco Bay (SF)	Montereyan	Unknown	130 (0–410)	3,503 (689–9,913)
Half Moon Bay (HB)	Montereyan	Unknown	156 (0–389)	744 (222–1,869)
Año Nuevo (AN)	Montereyan	Extant	255 (150–353)	440 (288–651)
Monterey Bay (MB)	Montereyan	Extant	215 (163–263)	248 (156–375)
Elkhorn Slough (ES)	Montereyan	Extant	154 (133–175)	115 (35–246)
Range Center (RC)	Montereyan	Extant	1,016 (803–1,216)	1,189 (864–1,584)
Central Coast (CC)	Montereyan	Extant	414 (81–735)	1,064 (755–1,450)
Point Conception (PC)	Montereyan	Extant	391 (86–651)	810 (572–1,147)
Santa Barbara (SB)	Southern Californian	Extant	95 (13–194)	165 (53–398)
Ventura (S1)	Southern Californian	Unknown	43 (0–120)	418 (138–959)
Los Angeles (S2)	Southern Californian	Unknown	1 (0–1)	200 (67–461)
Orange (S3)	Southern Californian	Extirpated	—	749 (234–1,842)
San Diego North (S4)	Southern Californian	Extirpated	—	320 (107–736)
San Diego South (S5)	Southern Californian	Extirpated	—	701 (223–1,684)
Northern Channel Islands (C1)	Southern Californian	Extant	158 (16–460)	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	Unknown	1 (0–3)	82 (24–201)
San Clemente Island (C3)	Southern Californian	Unknown	1 (0–4)	365 (119–857)
San Nicolas Island (SN)	Southern Californian	Extant	378 (327–429)	380 (184–694)
Baja California North	Southern Californian	Extirpated	—	Undetermined
Baja California South	Magdalenian	Extirpated	—	Undetermined
RANGE-WIDE TOTAL			3,454 (1,849–5,408)	21,764 (11,481–39,363) + Baja California

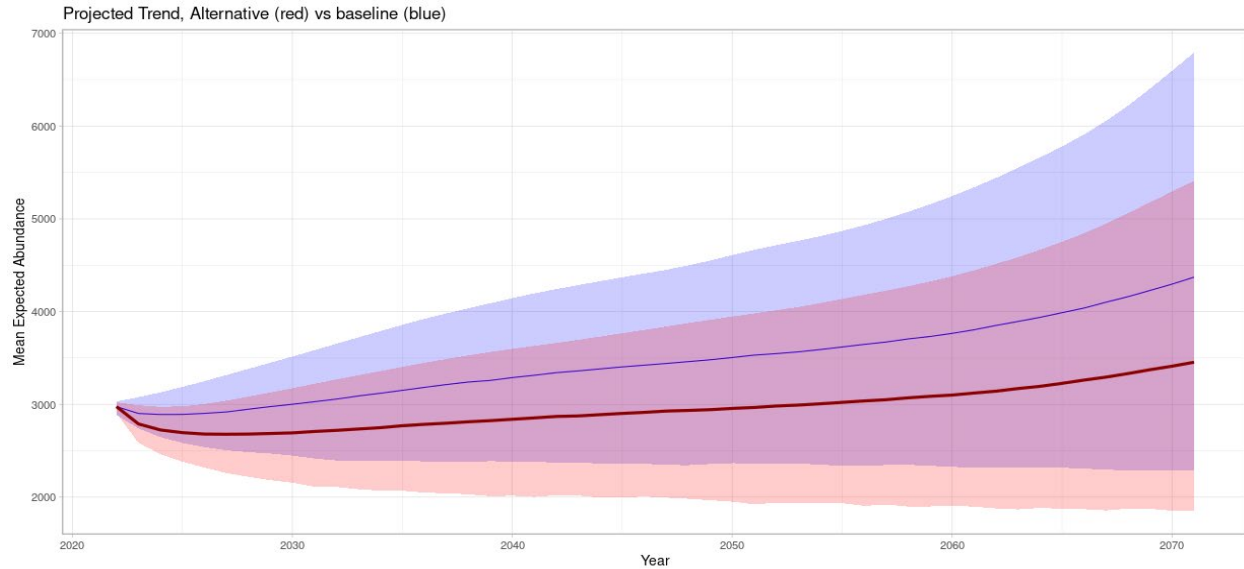


Figure 12. Projected population trend under the RCP 4.5 Scenario. The lines indicate mean expected abundance under current conditions, if carried forward (blue), and the RCP 4.5 scenario (red). Shading indicates the 80% credible intervals surrounding the respective means.

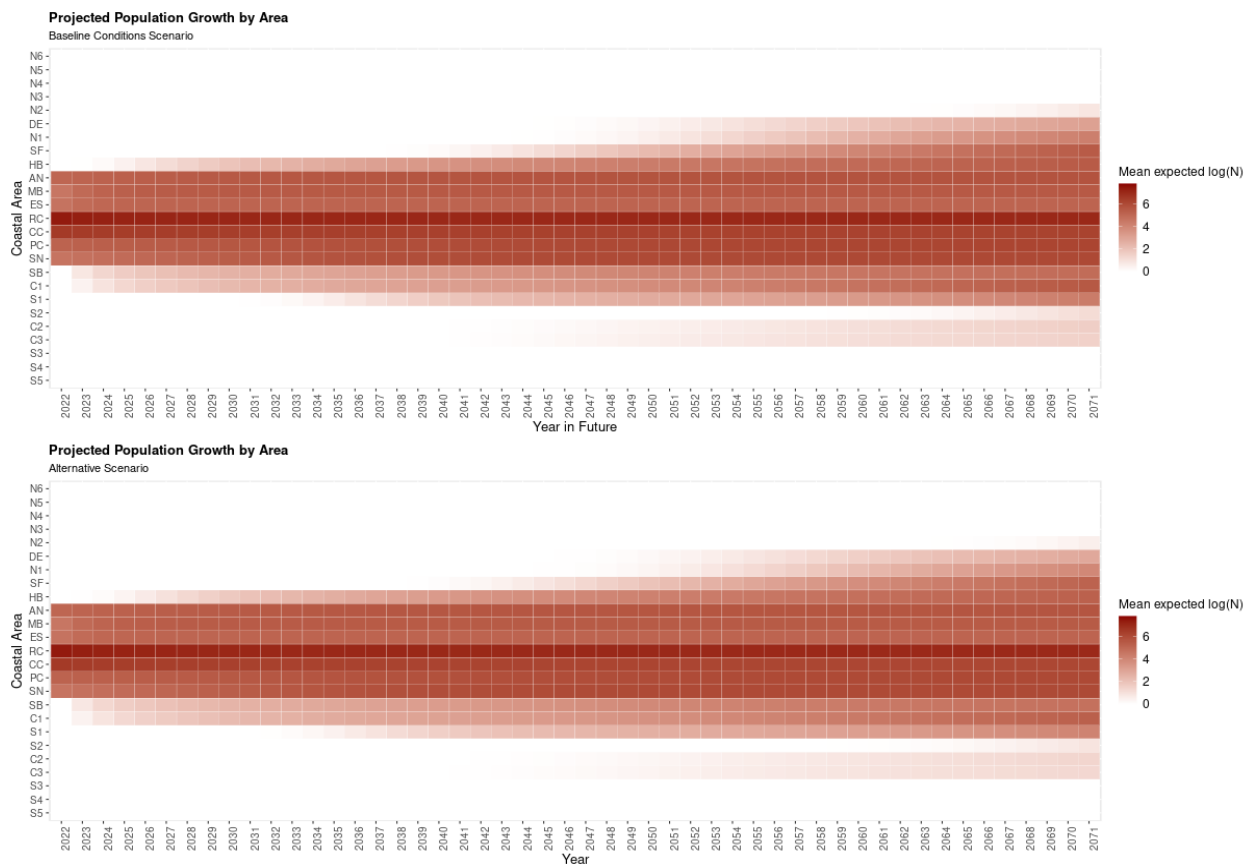


Figure 13. Projected population numbers by area under current conditions, if carried forward (top) and the RCP 4.5 scenario (bottom). Coastal areas within California, oriented north-south, are on the vertical axes. Shading indicates occupation of a coastal segment (darker red indicates higher numbers of sea otters). Sea otter abundance is on a log scale to increase the visibility of low sea otter numbers in newly occupied coastal segments.

Resiliency—Under the RCP 4.5 scenario, the resiliency of subpopulations was variable (Table 6). Most subpopulations to the north and south of the central California range remained extirpated. Additionally, 8 of the 17 subpopulations with a mean projected population size of 1 or more animals had credible intervals that included zero; it was unknown whether these subpopulations (N1, DE, SF, HB, S1, S2, C2, and C3) became extant or remained extirpated by the end of the projection period. Although SB and C1 became extant during the projection period, their relatively small mean population sizes and lower credible bounds in the teens suggested the potential for very low resiliency. On average, however, resiliency was somewhat increased relative to current conditions in 2022, with the addition of approximately 500 individuals rangewide. The SN subpopulation had a mean abundance at the end of the projection period that was approximately equal to its mean estimated carrying capacity, suggesting it had strong potential to serve as a source of animals dispersing into other southern subpopulations.

Redundancy—Redundancy under the RCP 4.5 scenario was slightly improved relative to current conditions in 2022, with the addition of the SB and CI subpopulations. However, even after 50 years, the extant subpopulations were not sufficiently distributed throughout the southern sea otter's ecological settings to spread risk in the case of catastrophic events. The spatial distribution of extant sea otter subpopulations under this scenario was still mostly restricted to the central California coastline, increasing the species' vulnerability to catastrophic events, such as one or more large oil spills, a rapidly spreading novel disease (in sea otters or their prey), or negative system-wide impacts resulting from the crossing of a climate-induced tipping point. Multiple severe widespread events occurring within a relatively short timeframe (i.e., with overlapping effects) are highly plausible given the potential for climate-induced changes to affect the nearshore marine ecosystem in numerous but unpredictable ways.

Representation—Representation under the RCP 4.5 scenario was similar to current conditions in 2022. Because the N1, DE, SF, and HB subpopulations had credible intervals that included zero, it was unknown whether they became extant. Therefore, the AN subpopulation remained the northernmost extant subpopulation. Given the lack of northward range expansion, interbreeding with northern sea otters from Washington remained very unlikely, and thus the low genetic diversity in the southern sea otter was not expected to increase during the projection period. Whereas the historical range spanned four ecoregions, extant subpopulations after 50 years remained restricted to two ecoregions, as under current conditions in 2022. Within the Mendocinian Ecoregion, all four subpopulations remained extirpated. Within the Montereyan Ecoregion, 6 of 13 subpopulations were extant, 4 were of unknown status, and 3 remained extirpated. Within the Southern Californian ecoregion, 3 of 11+ subpopulations were extant, 4 were of unknown status, and 5+ remained extirpated. No subpopulation was extant in the Magdalenian Ecoregion. The presence of the subpopulation in the Elkhorn Slough estuary (ES), and likely the Morro Bay estuary (which was lumped into the CC subpopulation), contributed to diversity in the ecological settings occupied by southern sea otter subpopulations, which were otherwise situated along the rocky outer coast or in sandy embayments. The status of subpopulations in other estuaries, like Drake's Estero (DE) and the extremely large complex of estuaries that is the San Francisco Bay (SF), was unknown. However, other estuaries in northern California and Oregon remained unoccupied, as are did all bays and estuaries in Baja California. Complex island habitat began to be recolonized (C1), but the bay-headland-island complexes of Baja California, which formerly sustained expansive populations of southern sea otters, remained

unoccupied.

6.3.2 RCP 8.5 Scenario

For the RCP 8.5 Scenario, we adjusted inputs to the IPM to model the plausible cumulative effects of extreme climate change, in which there are no significant global efforts to limit or reduce emissions, as described in section 6.1 and summarized in Table 5. Specifically, we increased the hazard rates of shark bite, protozoal infection, and infection (other) directly by 30%. To simulate decreases in prey availability as a result of ocean acidification and warming, we adjusted average density at carrying capacity downward by 30%. Other hazard rates were affected by this downward adjustment in carrying capacity: cardiac disease and natural causes (other), which are moderately density dependent, and ELS and acanthocephalan peritonitis, which are strongly density dependent (see Figure 6 in Tinker et al. 2021a, p. 19). We modeled HAB intoxication (and thereby further influenced rates of cardiac disease) by increasing the frequency and severity of harmful algal blooms by 30%. We did not alter the hazard rates from the current conditions projection for natural causes (other) or human causes (boat strikes, shootings, entanglements), and we did not add any oil spill.

Under this scenario, the mean expected abundance of sea otters in 50 years was 2,075 (1,205–3,091) (Table 7, Figures 14–15), 30% smaller than the estimated starting population size of 2,975 independent animals in 2022. However, there was considerable uncertainty, with plausible outcomes ranging from a slight increase to severe declines relative to the 2022 population estimate (Figure 14). The range extent increased only slightly from current conditions in 2022. The following subpopulations were extant after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN (Table 7 and Figure 15; see Figure 8 for subpopulation locations), indicating no range expansion to the north but some range expansion to the south (though with small mean population sizes) relative to current conditions in 2022.

TABLE 7. PROJECTED POPULATION SIZE AND DISTRIBUTION OF SEA OTTERS IN 50 YEARS, RCP 8.5 SCENARIO. CREDIBLE INTERVALS (CRI) ARE GIVEN IN PARENTHESES. SUBPOPULATIONS WITH A CRI THAT DOES NOT INCLUDE ZERO ARE CONSIDERED EXTANT. SUBPOPULATIONS WITH A MEAN ABUNDANCE GREATER THAN ZERO BUT WITH A CRI THAT INCLUDES ZERO ARE CONSIDERED UNKNOWN.

SUBPOPULATION	ECOREGION	STATUS	ABUNDANCE IN 2071 (80% CRI)	ESTIMATED CARRYING CAPACITY (95% CRI)
Oregon Central	Mendocinian	Extirpated	—	1287 (494–2,600)
Oregon South	Mendocinian	Extirpated	—	1785 (686–3,606)
Del Norte (N6)	Mendocinian	Extirpated	—	428 (136–1,043)
Humboldt (N5)	Mendocinian	Extirpated	—	469 (186–987)
Mendocino North (N4)	Montereyan	Extirpated	—	118 (37–285)
Mendocino South (N3)	Montereyan	Extirpated	—	155 (50–370)
Sonoma	Montereyan	Extirpated	—	186

(N2)				(60–447)
Marin (N1)	Montereyan	Unknown	4 (0–2)	742 (257–1,709)
Drake’s Estero (DE)	Montereyan	Unknown	2 (0–2)	62 (40–114)
San Francisco Bay (SF)	Montereyan	Unknown	18 (0–51)	3,503 (689–9,913)
Half Moon Bay (HB)	Montereyan	Unknown	36 (0–148)	744 (222–1,869)
Año Nuevo (AN)	Montereyan	Extant	169 (90–248)	440 (288–651)
Monterey Bay (MB)	Montereyan	Extant	152 (105–193)	248 (156–375)
Elkhorn Slough (ES)	Montereyan	Extant	114 (94–131)	115 (35–246)
Range Center (RC)	Montereyan	Extant	729 (537–904)	1,189 (864–1,584)
Central Coast (CC)	Montereyan	Extant	242 (29–485)	1,064 (755–1,450)
Point Conception (PC)	Montereyan	Extant	236 (34–440)	810 (572–1,147)
Santa Barbara (SB)	Southern Californian	Extant	37 (7–121)	165 (53–398)
Ventura (S1)	Southern Californian	Unknown	8 (0–21)	418 (138–959)
Los Angeles (S2)	Southern Californian	Unknown	—	200 (67–461)
Orange (S3)	Southern Californian	Extirpated	—	749 (234–1,842)
San Diego North (S4)	Southern Californian	Extirpated	—	320 (107–736)
San Diego South (S5)	Southern Californian	Extirpated	—	701 (223–1,684)
Northern Channel Islands (C1)	Southern Californian	Extant	44 (10–116)	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	Unknown	—	82 (24–201)
San Clemente Island (C3)	Southern Californian	Unknown	—	365 (119–857)
San Nicolas Island (SN)	Southern Californian	Extant	283 (244–323)	380 (184–694)
Baja California North	Southern Californian	Extirpated	—	Undetermined
Baja California South	Magdalenian	Extirpated	—	Undetermined
RANGE-WIDE TOTAL			2,075 (1,205–3,091)	21,764 (11,481–39,363) + Baja California

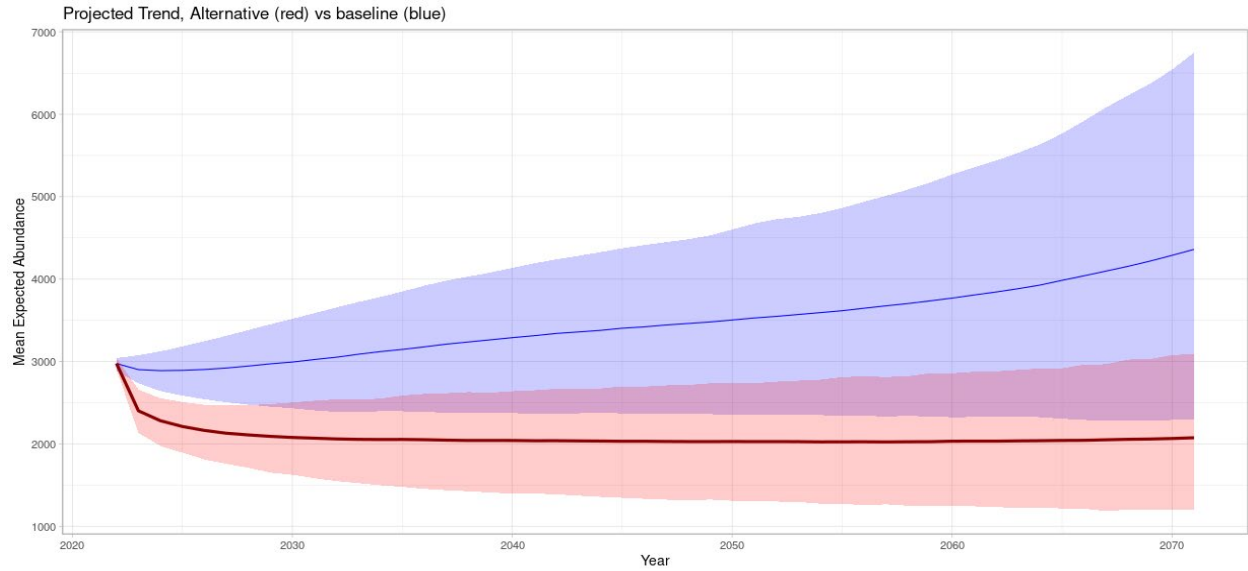


Figure 14. Projected population trend under the RCP 8.5 Scenario. The lines indicate mean expected abundance under current conditions, if carried forward (blue), and the RCP 8.5 scenario (red). Shading indicates the 80% credible intervals surrounding the respective means.

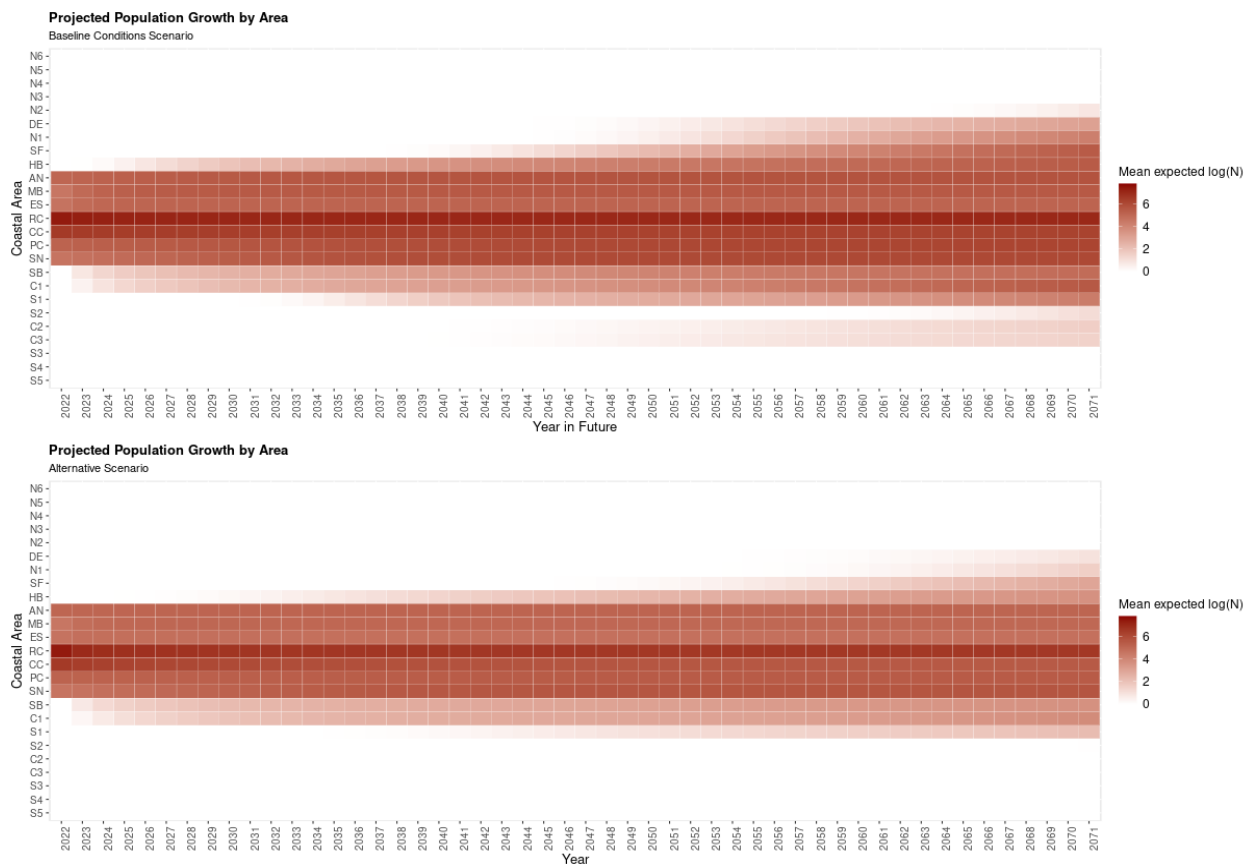


Figure 15. Projected population numbers by area under current conditions, if carried forward (top) and the RCP 8.5 scenario (bottom). Coastal areas within California, oriented north-south, are on the vertical axes. Shading indicates occupation of a coastal segment (darker red indicates higher numbers of sea otters). Sea otter abundance is on a log scale to increase the visibility of low sea otter numbers in newly occupied coastal segments.

Resiliency—Under the RCP 8.5 scenario, the resiliency of subpopulations (Table 7) was reduced (see Table 9 for comparisons between scenarios). Most subpopulations to the north and south of the central California range remained extirpated. Additionally, 5 of the 14 subpopulations with a mean projected population size of 1 or more animals had credible intervals that included zero; it was unknown whether these subpopulations (N1, DE, SF, HB, S1) became extant or remained extirpated by the end of the projection period. The newly extant subpopulations, SB and C1, had small mean population sizes and lower credible bounds of 7 and 10, respectively, suggesting the potential for extremely low resiliency of these subpopulations. All extant subpopulations were subject to substantial proportional increases in shark-bite mortality and a suite of other stressors (reductions in prey availability and increases in disease), as described above. These changes translated into smaller mean population sizes generally, though with considerable uncertainty around these means. This reduced resiliency relative to current conditions in 2022 was reflected in a mean abundance at the end of the projection period that was approximately 900 individuals smaller than that under current conditions in 2022.

Climate-related effects were greatest in the northern and southern regions of the central California range because of the strong influence of shark-bite mortality on demographic trends and the already-high rates of shark-bite mortality in those areas. The reduced resiliency of these subpopulations (AN, CC, and PC) strongly impeded connectivity between more densely occupied subpopulations in central California (MB, ES, and RC) and adjacent unoccupied historical habitat, further slowing range expansion. SN was less affected by proportional increases in shark bite mortality and disease because these hazards already had low rates of occurrence in this subpopulation. However, this subpopulation had a mean abundance at the end of the projection period that was below its mean estimated carrying capacity (see section 5.5.2) primarily because of climate-related changes (i.e., the forced reduction in average local density at carrying capacity due to changes in prey availability). Despite its reduced mean population size, it retained some potential to serve a source of animals dispersing into other southern subpopulations.

Redundancy—Redundancy under the RCP 8.5 scenario was slightly improved relative to current conditions in 2022, with the addition of the SB and CI subpopulations, though as noted above, these subpopulations had the potential for extremely low resiliency. However, even after 50 years, the extant subpopulations were not sufficiently distributed throughout the southern sea otter's ecological settings or geographical range to spread risk in the case of catastrophic events. The spatial distribution of extant sea otter subpopulations under this scenario was still mostly restricted to the central California coastline, increasing the species' vulnerability to catastrophic events, such as one or more large oil spills, a rapidly spreading novel disease (in sea otters or their prey), or negative system-wide impacts resulting from the crossing of a climate-induced tipping point. Multiple severe widespread events occurring within a relatively short timeframe (i.e., with overlapping effects) are highly plausible given the potential for climate-induced changes to affect the nearshore marine ecosystem in numerous but unpredictable ways.

Representation—Representation under the RCP 8.5 scenario was similar to current conditions in 2022. Because the N1, DE, SF, and HB subpopulations had credible intervals that included zero, it was unknown whether they became extant. Therefore, the AN subpopulation remained the northernmost extant subpopulation. Given the lack of northward range expansion, interbreeding

with northern sea otters from Washington remained very unlikely, and thus the low genetic diversity in the southern sea otter was not expected to increase during the projection period. Whereas the historical range spanned four ecoregions, extant subpopulations after 50 years remained restricted to two ecoregions, as under current conditions in 2022. Within the Mendocinian Ecoregion, all four subpopulations remained extirpated. Within the Montereyan Ecoregion, 6 of 13 subpopulations were extant, 4 were of unknown status, and 3 remained extirpated. Within the Southern Californian ecoregion, 3 of 11+ subpopulations were extant, 1 was of unknown status, and 7+ remained extirpated. No subpopulation was extant in the Magdalenian Ecoregion. The presence of the subpopulation in the Elkhorn Slough estuary (ES), and likely the Morro Bay estuary (which was lumped into the CC subpopulation), contributed to diversity in the ecological settings occupied by southern sea otter subpopulations, which were otherwise situated along the rocky outer coast or in sandy embayments, but the status of the DE and SF subpopulations was unknown. Other estuaries in northern California and Oregon remained unoccupied, as are did all bays and estuaries in Baja California. Complex island habitat began to be recolonized (C1), but the bay-headland-island complexes of Baja California, which formerly sustained expansive populations of southern sea otters, remained unoccupied.

6.3.3 RCP 8.5 + Large Oil Spill Scenario

For the RCP 8.5 Scenario, we adjusted inputs to the IPM to model the plausible cumulative effects of extreme climate change, in which there are no significant global efforts to limit or reduce emissions, as described in section 6.1 and summarized in Table 5. Specifically, we increased the hazard rates of shark bite, protozoal infection, and infection (other) directly by 30%. To simulate decreases in prey availability as a result of ocean acidification and warming, we adjusted average density at carrying capacity downward by 30%. Other hazard rates were affected by this downward adjustment in carrying capacity: cardiac disease and natural causes (other), which are moderately density dependent, and ELS and acanthocephalan peritonitis, which are strongly density dependent (see Figure 6 in Tinker et al. 2021a, p. 19). We modeled HAB intoxication (and thereby further influenced rates of cardiac disease) by increasing the frequency and severity of harmful algal blooms by 30%. We did not alter the hazard rates from the current conditions projection for natural causes (other) or human causes (boat strikes, shootings, entanglements). To examine how an oil spill could interact with climate-driven stressors, we added to this scenario a large oil spill (10 million gallons) near San Francisco during the summer of 2037 (1/3 of the way through the 50-year projection) (see section 6.2).

Under this scenario, the mean expected abundance of sea otters in 50 years was 1,992 (1,144–2,946) (Table 8, Figures 16–17), 33% smaller than the estimated starting population size of 2,975 independent animals in 2022. Although there was considerable uncertainty, plausible outcomes ranged from moderate to severe declines in abundance relative to the 2022 population estimate (Figure 16). The following subpopulations were extant after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN (Table 8 and Figure 17; see Figure 8 for subpopulation locations), indicating no range expansion to the north but some range expansion to the south (though with small mean population sizes) relative to current conditions in 2022.

TABLE 8. PROJECTED POPULATION SIZE AND DISTRIBUTION OF SEA OTTERS IN 50 YEARS, RCP 8.5 + LARGE OIL SPILL SCENARIO. CREDIBLE INTERVALS (CRI) ARE GIVEN IN PARENTHESES. SUBPOPULATIONS WITH A CRI THAT DOES NOT INCLUDE ZERO ARE CONSIDERED EXTANT. SUBPOPULATIONS WITH A MEAN ABUNDANCE GREATER THAN ZERO BUT WITH A CRI THAT INCLUDES ZERO ARE CONSIDERED UNKNOWN.

SUBPOPULATION	ECOREGION	STATUS	ABUNDANCE IN 2071 (80% CRI)	ESTIMATED CARRYING CAPACITY (95% CRI)
Oregon Central	Mendocinian	Extirpated	—	1287 (494–2600)
Oregon South	Mendocinian	Extirpated	—	1785 (686–3606)
Del Norte (N6)	Mendocinian	Extirpated	—	428 (136–1043)
Humboldt (N5)	Mendocinian	Extirpated	—	469 (186–987)
Mendocino North (N4)	Montereyan	Extirpated	—	118 (37–285)
Mendocino South (N3)	Montereyan	Extirpated	—	155 (50–370)
Sonoma (N2)	Montereyan	Extirpated	—	186 (60–447)

Marin (N1)	Montereyan	Extirpated	—	742 (257–1,709)
Drake's Estero (DE)	Montereyan	Extirpated	—	62 (40–114)
San Francisco Bay (SF)	Montereyan	Unknown	2 (0–1)	3,503 (689–9,913)
Half Moon Bay (HB)	Montereyan	Unknown	14 (0–70)	744 (222–1,869)
Año Nuevo (AN)	Montereyan	Extant	160 (82–236)	440 (288–651)
Monterey Bay (MB)	Montereyan	Extant	148 (101–188)	248 (156–375)
Elkhorn Slough (ES)	Montereyan	Extant	112 (93–130)	115 (35–246)
Range Center (RC)	Montereyan	Extant	708 (500–889)	1,189 (864–1,584)
Central Coast (CC)	Montereyan	Extant	241 (29–490)	1,064 (755–1,450)
Point Conception (PC)	Montereyan	Extant	236 (32–449)	810 (572–1,147)
Santa Barbara (SB)	Southern Californian	Extant	37 (7–119)	165 (53–398)
Ventura (S1)	Southern Californian	Unknown	8 (0–21)	418 (138–959)
Los Angeles (S2)	Southern Californian	Extirpated	—	200 (67–461)
Orange (S3)	Southern Californian	Extirpated	—	749 (234–1,842)
San Diego North (S4)	Southern Californian	Extirpated	—	320 (107–736)
San Diego South (S5)	Southern Californian	Extirpated	—	701 (223–1,684)
Northern Channel Islands (C1)	Southern Californian	Extant	44 (10–111)	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	Extirpated	—	82 (24–201)
San Clemente Island (C3)	Southern Californian	Extirpated	—	365 (119–857)
San Nicolas Island (SN)	Southern Californian	Extant	283 (244–323)	380 (184–694)
Baja California North	Southern Californian	Extirpated	—	Undetermined
Baja California South	Magdalenian	Extirpated	—	Undetermined
RANGE-WIDE TOTAL			1,993 (1,144–2,946)	21,764 (11,481–39,363) + Baja California

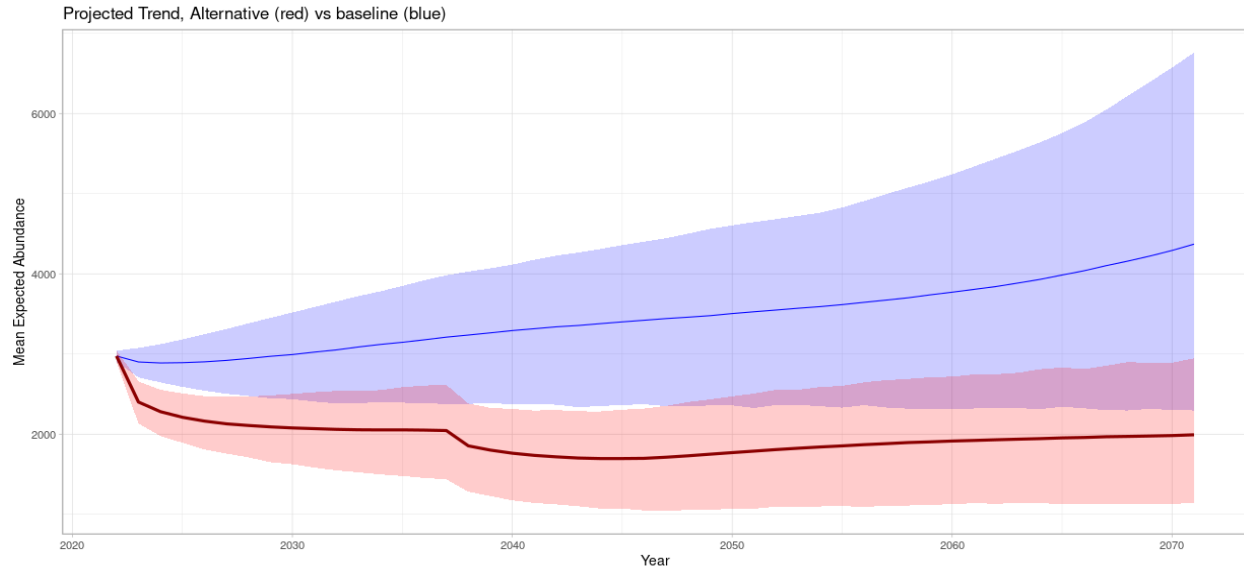


Figure 16. Projected population trend under the RCP 8.5 + Large Oil Spill Scenario. The lines indicate mean expected abundance under current conditions, if carried forward (blue), and the RCP 8.5 + Large Oil Spill Scenario (red). Shading indicates the 80% credible intervals surrounding the respective means.

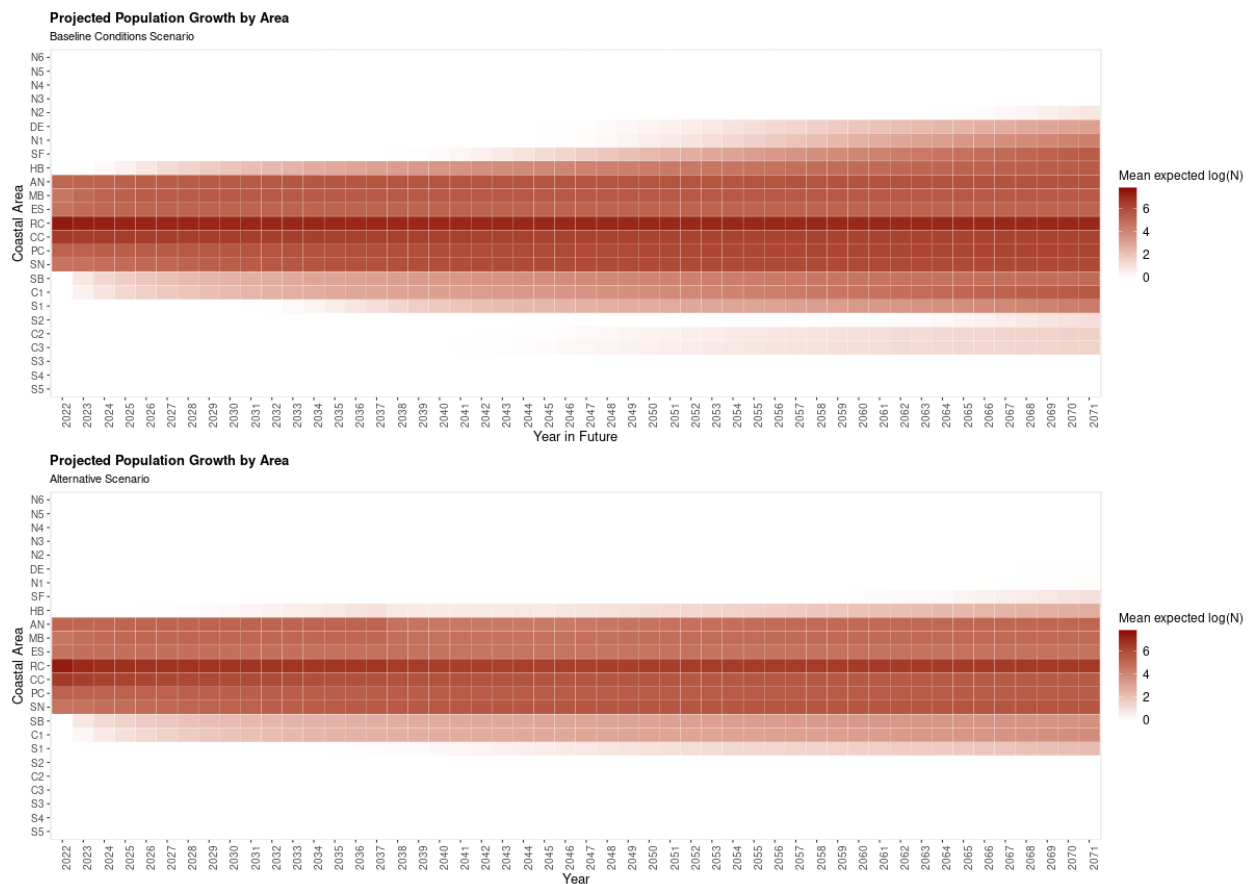


Figure 17. Projected population numbers by area, current conditions (top) and RCP 8.5 + Large Oil Spill (bottom) scenarios. Coastal areas within California, oriented north-south, are on the vertical axes. Shading indicates occupation of a coastal segment (darker red indicates higher numbers of sea otters). Sea otter abundance is on a log scale to increase the visibility of low sea otter numbers in newly occupied coastal segments.

Resiliency—Under the RCP 8.5 + Large Oil Spill scenario, the resiliency of subpopulations (Table 8) was reduced (see Table 9 for comparisons between scenarios). Most subpopulations to the north and south of the central California range remained extirpated. Additionally, 3 of the 12 subpopulations with a mean projected population size of 1 or more animals had credible intervals that included zero; it was unknown whether these subpopulations (SF, HB, S1) became extant or remained extirpated by the end of the projection period. The newly extant subpopulations, SB and C1, had small mean population sizes and lower credible bounds of 7 and 10, respectively, suggesting the potential for extremely low resiliency of these subpopulations. All extant subpopulations were subject to substantial proportional increases in shark-bite mortality and a suite of other stressors (reductions in prey availability and increases in disease), as described above. These changes translated into smaller mean population sizes generally, though with considerable uncertainty around these means. This reduced resiliency relative to current conditions in 2022 was reflected in a mean abundance at the end of the projection period that was approximately 1,000 individuals smaller than that under current conditions in 2022.

Climate-related effects, which include an expected increase in shark-bite mortality, were greatest in the northern and southern regions of the central California range because of the strong influence of shark-bite mortality on demographic trends and the already-high rates of shark-bite mortality in those areas. The reduced resiliency of these subpopulations (AN, CC, and PC) would have strongly impeded connectivity between more densely occupied subpopulations in central California (MB, ES, and RC) and adjacent unoccupied historical habitat, further slowing range expansion. SN was less affected by proportional increases in shark bite mortality and disease because these hazards already had low rates of occurrence in this subpopulation. However, this subpopulation had a mean abundance at the end of the projection period that was below its mean estimated carrying capacity (see section 5.5.2) primarily because of climate-related changes (i.e., the forced reduction in average local density at carrying capacity due to changes in prey availability). Despite its reduced mean population size, it retained some potential to serve a source of animals dispersing into other southern subpopulations.

Diminished resiliency was also apparent in the slow rebound of the population from the simulated large oil spill near San Francisco in 2037, which was compounded by the chronic effects of oil exposure (Figure 16; see also Appendix). However, acute and chronic effects of the spill were somewhat muted by the climate-related factors limiting population growth and range expansion at the northern range periphery even before the spill was simulated to occur. Limited northward range expansion prior to the spill reduced the overall population exposure. Nevertheless, the resulting smaller size of the population during the years following the oil spill made it especially vulnerable to stochastic disturbance. The population declined from 2,046 (1,437–2,615) in 2037 to 1,713 (1,041–2,354) in 2047. In this context, the lower bound of the credible interval is especially relevant, because if the population declined to very small numbers, even relatively small additional perturbations could potentially cause extirpation of subpopulations or even extinction of the species. However, simulations of the RCP 8.5 scenario without a large oil spill resulted in a similar range-wide total population size by the end of the projection period—2,075 (1,205–3,091) as opposed to 1,993 (1,144–2,946)—indicating that the primary drivers of population trends at the broadest timescale, so long as the population survived extinction during its period of greatest vulnerability, were the combined effects of other climate-related factors.

Redundancy—Redundancy under the RCP 8.5 + Large Oil Spill scenario was slightly improved relative to current conditions in 2022, with the addition of the SB and CI subpopulations, though as noted above, these subpopulations had the potential for extremely low resiliency. However, even after 50 years, the extant subpopulations were not sufficiently distributed throughout the southern sea otter’s ecological settings or geographical range to spread risk in the case of catastrophic events. The spatial distribution of extant sea otter subpopulations under this scenario was still mostly restricted to the central California coastline, increasing the species’ vulnerability to catastrophic events, such as an additional large oil spill, a rapidly spreading novel disease (in sea otters or their prey), or negative system-wide impacts resulting from the crossing of a climate-induced tipping point. Multiple severe widespread events occurring within a relatively short timeframe (i.e., with overlapping effects) are highly plausible given the potential for climate-induced changes to affect the nearshore marine ecosystem in numerous but unpredictable ways.

Representation—Representation under the RCP 8.5 + Large Oil Spill scenario was similar to current conditions in 2022. Because the SF and HB subpopulations had credible intervals that included zero, it was unknown whether they became extant. Therefore, the AN subpopulation remained the northernmost extant subpopulation. Given the lack of northward range expansion, interbreeding with northern sea otters from Washington remained very unlikely, and thus the low genetic diversity in the southern sea otter was not expected to increase during the projection period. Whereas the historical range spanned four ecoregions, extant subpopulations after 50 years remained restricted to two ecoregions, as under current conditions in 2022. Within the Mendocinian Ecoregion, all four subpopulations remained extirpated. Within the Montereyan Ecoregion, 6 of 13 subpopulations were extant, 2 were of unknown status, and 5 remained extirpated. Within the Southern Californian ecoregion, 3 of 11+ subpopulations were extant, 1 was of unknown status, and 7+ remained extirpated. No subpopulation was extant in the Magdalenian Ecoregion. The presence of the subpopulation in the Elkhorn Slough estuary (ES), and likely the Morro Bay estuary (which was lumped into the CC subpopulation), contributed to diversity in the ecological settings occupied by southern sea otter subpopulations, which were otherwise situated along the rocky outer coast or in sandy embayments, but the subpopulation in Drake’s Estero (DE) remained extirpated, and the status of the SF subpopulation was unknown (and the added representation was negligible regardless, with an upper credible interval of only one sea otter). Other estuaries in northern California and Oregon remained unoccupied, as did all bays and estuaries in Baja California. Complex island habitat began to be recolonized (C1), but the bay-headland-island complexes of Baja California, which formerly sustained expansive populations of southern sea otters, remained unoccupied.

6.4 Future Conditions—Summary

6.4.1 Factors Influencing Viability (Resiliency, Redundancy, and Representation)

The three plausible future scenarios considered here, RCP 4.5, RCP 8.5, and RCP 8.5 + Large Oil Spill, resulted in divergent effects on resiliency. The RCP 4.5 scenario resulted in a 16% increase in the mean number of southern sea otters relative to current conditions in 2022, whereas the RCP 8.5 and RCP 8.5 + Large Oil Spill scenarios resulted in declines of 30% and 33%, respectively (all with considerable uncertainty, as reflected in the large credible intervals

surrounding these means). However, in all three plausible future scenarios most subpopulations to the north and south of the central California range remained extirpated at the end of the 50-year projection, and connectivity with resource-abundant unoccupied habitat was curtailed by high shark-bite mortality. Redundancy after 50 years was slightly improved under all three plausible future scenarios relative to current conditions in 2022 due to the addition of the SB and CI subpopulations in southern California, though in all cases these subpopulations had small or very small population sizes and thus the potential for extremely low resiliency. In no scenario were extant sea otter subpopulations sufficiently distributed throughout the southern sea otter's ecological settings or geographical range to spread risk in the case of catastrophic events, especially in light of the risk of multiple severe widespread events occurring within a relatively short timeframe as a result of climate-induced effects. Representation, both in terms of genetic diversity and distribution across a diversity of ecoregions, remained compromised under all three plausible future scenarios and was similar to current conditions in 2022.

6.4.2 Uncertainties

Numerous uncertainties should be kept in mind when considering these results (see also section 5.5.2). The IPM used to conduct the simulations for the current conditions projection and alternative future scenarios uses a biologically sound, spatially explicit, and quantitative approach to integrate the combined effects of different hazards on demographic processes, and it explicitly accounts for numerous sources of uncertainty, as reflected in wide credible intervals in its results (Tinker et al. 2021a). However, the uncertainty captured by the IPM does not reflect future changes in parameter values that may occur (such as the locations or rates of shark bite mortality or other hazards) if these changes are not explicitly included as scenario inputs, and it does not account for anthropogenic hazards in areas that are currently unoccupied but may become occupied in projections under certain scenarios, such as San Francisco Bay (Tinker et al. 2021a, p. 27). Additionally, projections of the responses of living marine organisms to global change, such as the one on which we base our assessment of the potential effects of ocean acidification on sea otters' prey resources (Marshall et al. 2017), contain uncertainty arising from internal variability, model uncertainty, and scenario uncertainty (Cheung et al. 2016, pp. 1284–1285). Marshall et al. (2017, pp. 1536–1538) account for some of this uncertainty by choosing representative years in terms of oceanography and biogeochemistry, by averaging over the 10 final years of the model run, or by aggregating results across space; however, as the authors acknowledge, such averaging can obscure dramatic changes in particular locations (or times). Potentially extreme outcomes could have significant effects on sea otters at small spatial or temporal scales, but we are unable to account for them in the scenarios we present here. Further, although we attempt to translate these ecosystem model results and downscaled climate projections (Pierce et al. 2018, p. iv) into appropriate quantitative inputs for the IPM simulations, the translation itself is based on a qualitative assessment of how numerous factors might interact to change the risks experienced by sea otters.

7.0 OVERALL SYNTHESIS

Threats to the southern sea otter identified at the time of listing were curtailment of its range as a consequence of the maritime fur trade, the threat of a major oil spill, the inadequacy of existing regulatory mechanisms to protect habitat, possible threats from pollution and competition with humans, and a possible loss of genetic diversity (due to near-extinction during the fur trade) (42 FR 2965; January 14, 1977). Since then, the range-wide population index has increased to 2,962 as of 2019 (the most recent year a full census was completed); the mainland range has increased by approximately 210 km (130 mi) to encompass roughly 500 km (310 mi) of linear coastline; and a translocated subpopulation has taken hold at San Nicolas Island. Despite these improvements, the southern sea otter's range remains extremely curtailed relative to its ecological settings and geographic range. Primarily because of high shark-bite mortality, net range expansion has not occurred in more than 20 years. Seven of 29+ subpopulations are currently extant. The results of population projections based on current conditions and three plausible future scenarios (Table 9) indicated that meaningful improvements in resiliency, redundancy, and representation are unlikely to occur on their own.

TABLE 9. COMPARISON OF PROJECTED POPULATION SIZE AND DISTRIBUTION IN 50 YEARS UNDER CURRENT CONDITIONS AND PLAUSIBLE FUTURE SCENARIOS. CREDIBLE INTERVALS ARE GIVEN IN PARENTHESES BELOW THE MEAN ESTIMATED ABUNDANCE. SUBPOPULATIONS WITH A CREDIBLE INTERVAL THAT DOES NOT INCLUDE ZERO ARE CONSIDERED TO BE EXTANT. EXTIRPATED SUBPOPULATIONS, AND THOSE OF UNKNOWN STATUS (THOSE WITH A CREDIBLE INTERVAL THAT INCLUDES ZERO), ARE SHADED GRAY.

SUB-POPULATION	ECOREGION	CURRENT CON-DITIONS*	ABUNDANCE IN 2071 (80% Cri)				ESTIMATED CARRYING CAPACITY (95% Cri)
			CURRENT CONDITIONS PROJECTED**	RCP 4.5	RCP 8.5	RCP 8.5 + LARGE OIL SPILL	
Oregon Central	Mendocinian	—	—	—	—	—	1,287 (494–2,600)
Oregon South	Mendocinian	—	—	—	—	—	1,785 (686–3,606)
Del Norte (N6)	Mendocinian	—	—	—	—	—	428 (136–1,043)
Humboldt (N5)	Mendocinian	—	—	—	—	—	469 (186–987)
Men-docino North (N4)	Montereyan	—	—	—	—	—	118 (37–285)
Men-docino South (N3)	Montereyan	—	—	—	—	—	155 (50–370)
Sonoma (N2)	Montereyan	—	—	—	—	—	186 (60–447)
Marin (N1)	Montereyan	—	68 (0–212)	35 (0–125)	4 (0–2)	—	742 (257–1,709)

Drake's Estero (DE)	Montereyan	—	21 (0–61)	12 (0–41)	2 (0–2)	—	62 (40–114)
San Francisco Bay (SF)	Montereyan	—	232 (0–626)	130 (0–410)	18 (0–51)	1 (0–1)	3,503 (689–9,913)
Half Moon Bay (HB)	Montereyan	—	232 (3–506)	156 (0–389)	36 (0–148)	14 (0–70)	744 (222–1,869)
Año Nuevo (AN)	Montereyan	162	306 (191–409)	255 (150–353)	169 (90–248)	160 (82–236)	440 (288–651)
Monterey Bay (MB)	Montereyan	81	250 (194–300)	215 (163–263)	152 (105–193)	148 (101–188)	248 (156–375)
Elkhorn Slough (ES)	Montereyan	104	176 (153–199)	154 (133–175)	114 (94–131)	112 (93–130)	115 (35–246)
Range Center (RC)	Montereyan	1,642	1,170 (933–1,383)	1,016 (803–1,216)	729 (537–904)	708 (500–889)	1,189 (864–1,584)
Central Coast (CC)	Montereyan	694	523 (137–889)	414 (81–735)	242 (29–485)	241 (29–490)	1,064 (755–1,450)
Point Conception (PC)	Montereyan	180	486 (146–767)	391 (86–651)	236 (34–440)	236 (32–449)	810 (572–1,147)
Santa Barbara (SB)	Southern Californian	2***	131 (17–229)	95 (13–194)	37 (7–121)	37 (7–119)	165 (53–398)
Ventura (S1)	Southern Californian	—	76 (12–206)	43 (0–120)	8 (0–21)	8 (0–21)	418 (138–959)
Los Angeles (S2)	Southern Californian	—	2 (0–9)	1 (0–1)	—	—	200 (67–461)
Orange (S3)	Southern Californian	—	—	—	—	—	749 (234–1,842)
San Diego North (S4)	Southern Californian	—	—	—	—	—	320 (107–736)
San Diego South (S5)	Southern Californian	—	—	—	—	—	701 (223–1,684)
Northern Channel Islands (C1)	Southern Californian	—	253 (20–629)	158 (16–460)	44 (10–116)	44 (10–111)	958 (320–2,194)
Santa Catalina Island (C2)	Southern Californian	—	4 (0–11)	1 (0–3)	—	—	82 (24–201)
San Clemente Island (C3)	Southern Californian	—	3 (0–9)	1 (0–4)	—	—	365 (119–857)

San Nicolas Island (SN)	Southern Californian	99	427 (372–483)	378 (327–429)	283 (244–323)	283 (244–323)	380 (184–694)
Baja California North	Southern Californian	—	—	—	—	—	Un-determined
Baja California South	Magdalenian	—	—	—	—	—	Un-determined
RANGE-WIDE TOTAL		2,692	4,362 (2,288–6,699)	3,454 (1,849–5,408)	2,075 (1,205–3,091)	1,992 (1,144–2,946)	21,764 (11,481–39,363) + Baja California
*Current conditions are based on the most recent complete survey data (from 2019). ****The current conditions projection is included for comparison but does not represent a plausible future scenario because environmental conditions are expected to change. ***Not included in total.							

7.1 Resiliency

The current abundance of 2,962 (Hatfield et al. 2019, p. 3) is far below the estimated carrying capacity of California, 17,226 sea otters (95% CrI=9,739–30,087) (Tinker et al. 2021b, p. 1), a figure that does not include historical range in Oregon and Baja California, Mexico. While this comparison provides general context, the southern sea otter's status with respect to carrying capacity is more meaningful at the subpopulation level because of the relatively small home ranges of sea otters, especially adult females. A subpopulation's status with respect to carrying capacity influences the effects of different stressors on subpopulations and determines which subpopulations have the potential for further growth. Sea otters in the central portion of the mainland range (i.e., the RC subpopulation) are at or near local carrying capacity, meaning there is limited potential for additional population growth in that subpopulation. Although the RC subpopulation is resilient, per-capita prey limitation negatively affects body condition and increases the susceptibility of individuals to natural and anthropogenic stressors. The AN, MB, CC, and PC subpopulations are all well below equilibrium abundance (Table 2), primarily because of moderate or high shark-bite mortality. Although these subpopulations could theoretically grow substantially based on resource availability, high shark-bite mortality has reduced the numbers of sea otters in these subpopulations and their resiliency. The exception in the northern region is the subpopulation in Elkhorn Slough (ES), an area that is shallow enough to exclude white sharks. Significant growth of the population as a whole will require range expansion into currently unoccupied habitat, but high shark-bite mortality in the subpopulations that lie between the most densely occupied portions of the range (ES and RC) and unoccupied habitat is limiting range expansion, which has not resulted in a net increase in more than 20 years. As a result, only 7 of 29+ subpopulations are extant, 22+ are extirpated, and one (SB) has 2 individuals (as of the most recent census).

Under the current conditions projection (which was included for comparison but does not represent a plausible future scenario because environmental conditions are expected to change), the mean population size increased over the next 50 years, from an estimated 2,975 sea otters in

2022 to 4,362 (80% CrI 2,288–6,699) in 2071. The wide credible interval around this estimate indicated considerable uncertainty in the underlying processes, with plausible outcomes including substantial growth as well as substantial declines. Some range expansion occurred to the north and south of the range that was occupied as of 2022, resulting in the following extant subpopulations: HB, AN, MB, ES, RC, CC, PC, SB, S1, C1, and SN (HB, SB, S1, and C1 became extant during the projection period, though the wide credible intervals and lower credible bounds of 20 or fewer sea otters indicated considerable uncertainty regarding the resiliency of these subpopulations; see Figure 8 for subpopulation locations).

Under the RCP 4.5 scenario, the mean abundance of sea otters in 2071 was 3,454 (1,849–5,408), a 16% increase from the estimated starting population size of 2,975 independent animals in 2022. As with the current conditions projection, plausible outcomes included more substantial growth as well as substantial declines. The range extent increased only slightly from current conditions in 2022, with no range expansion to the north but some range expansion to the south. The projection resulted in the following extant subpopulations after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN. SB and C1 became extant during the projection period, though with relatively small mean population sizes and lower credible bounds in the teens, suggesting very low resiliency in these subpopulations.

Under the RCP 8.5 scenario, the mean abundance of sea otters in 50 years was 2,075 (1,205–3,091), a 30% decrease from the estimated starting population size of 2,975 independent animals in 2022. However, there was considerable uncertainty, with plausible outcomes ranging from a slight increase to severe declines relative to the 2022 population estimate. The projection resulted in the same extant subpopulations as under the RCP 4.5 scenario, though with fewer sea otters in each subpopulation. The following subpopulations were extant after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN. SB and C1 again became extant during the projection period, though with relatively small mean population sizes and lower credible bounds of 10 or fewer, suggesting the potential for very low resiliency of these subpopulations).

Under the RCP 8.5 + Large Oil Spill scenario, the mean abundance of sea otters in 50 years was 1,992 (1,144–2,946), 33% smaller than the estimated starting population size of 2,975 independent animals in 2022. Although there was still considerable uncertainty, plausible outcomes ranged from moderate to severe declines relative to the 2022 population estimate. The projection resulted in the same extant subpopulations as under the RCP 4.5 and RCP 8.5 scenarios, though with slightly fewer sea otters in the northern and central subpopulations than under the RCP 8.5 scenario. The following subpopulations were extant after 50 years: AN, MB, ES, RC, CC, PC, SB, C1, and SN. SB and C1 again became extant during the projection period, though with relatively small mean population sizes and lower credible bounds of 10 or fewer, suggesting the potential for very low resiliency of these subpopulations).

Under current conditions and the plausible future scenarios, the ability of southern sea otter subpopulations to withstand and bounce back from stochastic events was expected to remain compromised by high levels of shark bite mortality, which maintained low sea otter abundance relative to estimated carrying capacity in subpopulations in the northern (AN) and southern (CC and PC) portions of the current southern sea otter range and slowed or prevented range expansion to the north and south of these areas. The SN subpopulation grew in all three cases,

demonstrating its strong potential to serve as a source of animals dispersing into other southern subpopulations. However, resiliency was reduced range-wide by changes related to climate under the three plausible future scenarios, which intensified some known hazards, including mortality from shark bites, and resulted in smaller mean subpopulation sizes in occupied areas of the range, detrimentally affecting the ability of southern sea otter subpopulations to withstand and bounce back from stochastic events.

7.2 Redundancy

The southern sea otter's range is currently severely curtailed, and catastrophic events remain a major consideration for the species. Although a major oil spill has not occurred within the southern sea otter's range, oil spill risk from tankers and other large vessels that transit the California coast remains a primary threat. Such an oil spill, on its own, may not cause extinction, but it would likely severely compromise the ability of the species to withstand additional events, such as a rapidly spreading novel disease (in sea otters or their prey) or negative system-wide impacts resulting from the crossing of a climate-induced tipping point. Multiple severe widespread events occurring within a relatively short timeframe (i.e., with overlapping effects) are highly plausible given the potential for climate-induced changes to affect the nearshore marine ecosystem in numerous but unpredictable ways. The spatial distribution of extant sea otter subpopulations is mostly restricted to the central California coastline, increasing the species' vulnerability to catastrophic events.

Although there were a number of resilient subpopulations after 50 years, they were not sufficiently populated or distributed throughout the southern sea otter's ecological settings or geographical range to reduce the risk of long-term impacts in the case of one or more catastrophic events. Depending on the type, scale, number, and location of these events, the existence of subpopulations in the Southern California Bight, particularly SN (SB and C1, which became extant under the RCP 4.5, RCP 8.5, and RCP 8.5 + Large Oil Spill scenarios, had potentially very low resiliency), could provide some redundancy. However, because the southern sea otters' spatial distribution under all scenarios was still mostly restricted to the central California coastline, it remained vulnerable to severe, widespread events.

7.3 Representation

The southern sea otter currently has low genetic and environmental diversity. The potentially deleterious alleles detected in southern sea otters and low levels of genetic variation are of concern. Inbred animals are generally less resilient to stress than outbred ones, and populations with low genetic diversity are less able to adapt to environmental change. The simulations conducted by Beichman et al. (2022) demonstrated that, without intervention, recessive genetic load in southern sea otters did not return to pre-fur-trade levels for 400 generations or $\approx 2,800$ years, though it was unlikely to cause extinction on its own. The southern sea otter's diminished evolutionary potential ("capacity to evolve genetically based changes in traits that increase population-level fitness in response to novel or changing environmental conditions"; Forester et al. 2022, p. 1), poses an additional extinction risk. When confronted with a changing environment and suite of stressors, species with low evolutionary potential are more likely to produce maladaptive phenotypes and thus to have reduced fitness, possibly leading to extirpation

or extinction (Forester et al., pp. 2–5). Environmental diversity is similarly limited. Existing subpopulations are mostly within the Montereyan ecoregion, with only one subpopulation (SN) extant and one with 2 individuals (SB) in the Southern Californian ecoregion. The Mendocinian and Magdalenian ecoregions are unoccupied.

Genetic and environmental diversity in the southern sea otter was not expected to increase during the projection period under future scenarios. Under all scenarios, interbreeding with northern sea otters from Washington remained very unlikely, given the limited northward range expansion of southern sea otters. Extant subpopulations after 50 years remained restricted to two of the four ecoregions in which southern sea otter subpopulations historically occurred. Under the three future scenarios, no subpopulations were extant within the Mendocinian Ecoregion, 6 of 13 subpopulations were extant within the Montereyan Ecoregion, 3 of 11+ subpopulations were extant within the Southern Californian ecoregion, and no subpopulation was extant in the Magdalenian Ecoregion. Because genetic diversity and environmental diversity did not increase, the southern sea otter’s evolutionary potential under future scenarios remained compromised, posing an extinction risk in light of changing environmental conditions.

8.0 RECOVERY CRITERIA EVALUATION

The recovery strategy in the 2003 recovery plan was to create conditions that would enable the southern sea otter population to increase to a size that allowed the species to persist following most natural or human-caused perturbations. The following criteria were developed to provide guidance on when reclassification would be appropriate (Service 2003, pp. 22–27; Table 9).

TABLE 9. RECOVERY CRITERIA

STATUS	CRITERION
Endangered	The southern sea otter population should be considered for reclassification as endangered under the Act if the population declines to a level fewer than or equal to an effective population size of 500 animals (Mace and Lande 1991). Until better information is available, we recommend using a multiplier of 3.7 to convert effective population size to actual population size (Ralls et al. 1983), or 1,850 animals. Therefore, the southern sea otter population should be considered endangered if, based on standard survey counts (i.e., spring surveys), the average population level over a 3-year period is fewer than or equal to 1,850 animals.
Threatened	The southern sea otter population should be considered threatened under the Act if the average population level over a 3-year period is greater than 1,850 animals, but fewer than 3,090 animals.
Delisted	The southern sea otter population should be considered for delisting under the Act when the average population level over a 3-year period exceeds 3,090 animals.

Franklin (1980) suggested that an effective population size of 500 is generally the minimum population size across species that allows a population to be resilient to changes in the environment on genetic grounds (i.e., to maintain evolutionary potential in perpetuity). At or above this number, the loss of genetic variation due to small population size was expected to be balanced or exceeded by the gains of mutation. However, based on new theoretical and empirical evidence, this recommendation has since been changed to 1,000 (Frankham et al. 2014). The number of actual individuals in a population required to achieve the desired effective population size will vary by the genetic diversity in the population (Frankel and Soulé 1981).

At the time of the 2003 recovery plan, the recovery criteria were predicated on the assumption that, for southern sea otters, effective population size would scale with an actual population size at a ratio of 1:3.7, as proposed by Ralls et al. (1983). Therefore, minimum viable population size was calculated to be approximately 1,850 animals. That is, an actual population size of 1,850 would have an effective population size of 500. Based on this calculation, the recovery plan recommended reclassification of the southern sea otter to endangered if the population declined to fewer than or equal to an effective population size of 500, or an actual population size of 1,850 over a 3-year period based on standard survey counts (i.e., spring surveys). The criterion for delisting the southern sea otter was based on the number of animals needed to ensure that an effective population size of 500 would survive following a major oil spill event, such as the *Exxon Valdez* oil spill, and to ensure that a declining trend of 5% per year could be detected. Based on the assumption that between 880 and 1600 southern sea otters could contact oil following a 40-million-liter (10,500,000-gallon or 250,000-barrel) oil spill event (the size of the *Exxon Valdez* spill) in central California, the delisting criterion required 1,240 animals in addition to the 1,850 animals assumed to provide an effective population size of 500, or an actual population size of 3,090 over 3-year period based on standard survey counts.

New information has revealed that the effective population size in southern sea otters is not increasing with population growth (Gagne et al. 2018). The researchers combined thirteen years of demographic and genetic data from 1,006 sea otters to assess multiple estimators of genetic diversity and effective population size. They found low levels of genetic diversity in southern sea otters when compared to other sea otter populations. Expected heterozygosity was 0.50, as compared to the range of 0.48–0.86 reported across sea otter populations by Larson et al. (2012), and did not change significantly over the 13-year study period (Gagne et al. 2018, p. 8). Rangewide estimates of effective population size were found to be as low as 341 using genetic data and as high as 1,230 using demographic data. Within just Monterey County, the genetic and demographic estimates were much more consistent, ranging from an effective population size of 200 animals using genetic data to 278 using demographic data. Gagne et al. (2018, p. 8) suggest that cryptic population structure across the sea otter range could be the primary factor responsible.

Because a single, precise estimate of the actual number of southern sea otters that corresponds to an effective population size of 500 could not be determined, Gagne et al. (2018, p. 9) concluded that use of an effective population size to actual population size scaling factor is not an appropriate metric for recovery criteria for this species. They suggested alternate approaches, such as population viability analyses, that would allow for the determination of extinction risk based on genetic and demographic factors (Gagne et al. 2018, p. 9).

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APPENDIX

Incorporating Oil Spill Simulations into the Southern Sea Otter

Integrated Population Model

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for

U.S. Fish and Wildlife Service

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Introduction

Southern sea otters face a variety of threats that have limited their potential to fully recover from depletion, re-occupy their historical range, and reach an optimum sustainable population size (OSP). Some of these threats are ongoing, chronic stressors that routinely affect survival and reproductive success, including shark-bite mortality, many types of infectious disease and nutritional limitation (Miller et al. 2020). Other threats, in contrast, can be described as rarely occurring events with unpredictable timing, but whose demographic impacts can be severe. For sea otters, a prime example of the latter class of threats are major oil spill events (Ballachey et al. 2014). Sea otters are highly susceptible to mortality from oil spills, as they occupy nearshore coastal waters where oil spills can occur and the insulative capacity of their pelage (on which they are entirely dependent for thermoregulation) is compromised by exposure to oil (Williams et al. 1988). Major oil spill events are rare and unpredictable phenomenon, yet when they have occurred in coastal waters inhabited by sea otters previously, they have resulted in extremely high levels of mortality and reductions in abundance, and so potentially represent a major threat to recovery (Monson et al. 2000).

At the request of the U.S. Fish and Wildlife Service (USFWS), the U.S. Geological Survey (USGS) developed an integrated population model (IPM) for southern sea otters (Tinker et al. in press). The IPM is an analytical tool that can assist in management decisions by allowing for quantitative evaluation of the impact of specific threats on future population growth, investigations of how future environmental changes are likely to affect recovery, and assessments of the efficacy of alternative management scenarios. However, while the IPM incorporates extensive data on sea otter vital rates and a wide variety of different threats, it does not currently include the ability to evaluate rare and unpredictable phenomenon such as oil spills. This omission means that the projections of the IPM may be biased to a certain degree as they do not account for the potential effects of a major oil spill, which, while having a low probability on any given year, nonetheless has the potential to cause substantial depletion of the

population should it occur. Fortunately, incorporation of oil spill threats into the IPM is now feasible given the availability of 1) extensive data on the survival impacts of oil exposure for sea otters, both immediate/acute and long term/chronic (Davis et al. 1988, Williams et al. 1988, Monson et al. 2000, Ballachey et al. 2002), and 2) spatially explicit simulation models of the wind- and current-base dispersal of oil after spills that might occur in different locations (Beegle-Krause 2001, Marta-Almeida et al. 2013). Here, we combine these two information sources and expand the existing IPM model structure to incorporate the dynamics of simulated oil spills. To accomplish this we use the National Oceanic and Atmospheric Administration's (NOAA) oil spill simulation environment (Beegle-Krause 2001) to: 1) conduct a large number of simulations of large oil spill events within the sea otter range; 2) spatially intersect these simulated spills with the current and/or future distributions of sea otters; and 3) estimate levels of exposure to and mortality from oil spills, including both acute and chronic (long-term) exposure. The results of these simulations are then incorporated into the existing IPM web tool, to allow for more unbiased projections of future population dynamics that allow for the possibility of rare/stochastic oil spill events and their consequences.

Methods

Overview

To create realistic oil spill scenarios for southern sea otters, we made use of the General NOAA Operational Modeling Environment (GNOME), a set of modeling tools for predicting the fate and transport of pollutants (such as oil) spilled in water (Beegle-Krause 2001). We ran oil spill simulations for a representative set of spill locations in the northern and southern portions of the sea otters' range, and for multiple volumes of oil and seasonal environmental conditions (using historical data on winds and currents). We next created an intersection model (following Bodkin and Udevitz 1994) to estimate the expected exposure of sea otters to surface oil at concentrations sufficient to cause mortality. Finally, we

use the distribution of simulation results to parameterize an oil spill module incorporated into the existing integrated population model (IPM) for southern sea otters (Tinker et al. in press).

Oil Spill Simulations

To facilitate batch-processing and automation of the oil spill simulations, we used the “PgGnome” programming environment (<http://noaa-orr-erd.github.io/PyGnome/index.html>), an implementation of GNOME in the Python programming language that allows customized oil spill scenarios to be specified and run as scripts (Hou et al. 2015). We wished to evaluate oil spill impacts to sea otter populations based on various scenarios defined by a combination of spill location, size (i.e., number of gallons spilled) and seasonal environmental conditions (i.e., winds and currents, which determine how surface oil moves and disperses over time). The two locations of interest were the northern end of the sea otters’ range near the entrance to San Francisco, and the southern end of the sea otters’ range near Pt Conception (Figure 1): both these locations are recognised as high-risk areas based on the regular transit of VLCC (Very Large Crude Carriers) with a full load discharge of 2 million barrels, or about 80 million gallons. We wished to evaluate impacts of a medium-size spill and a large (catastrophic) spill, which we defined based on the US Coast Guard categorization of spill size for each region, including Worst Case Discharge (WCD), Maximum Most Probable Discharge (MMPD), and Average Most Probable Discharges (AMPD). Based on the WCD, MMPD and AMPD values for Long Beach and San Francisco we settled on a medium spill size of 1 million gallons and a large spill size of 10 million gallons. For each general location and spill size, we ran oil spill simulations for several sets of arbitrary spatial coordinates and date ranges selected from summer months (May – August) and winter months (October – February). These date ranges were selected because winds and currents in coastal California tend to vary between these two seasons. Table 1 shows a summary of the key parameters for all simulations run.

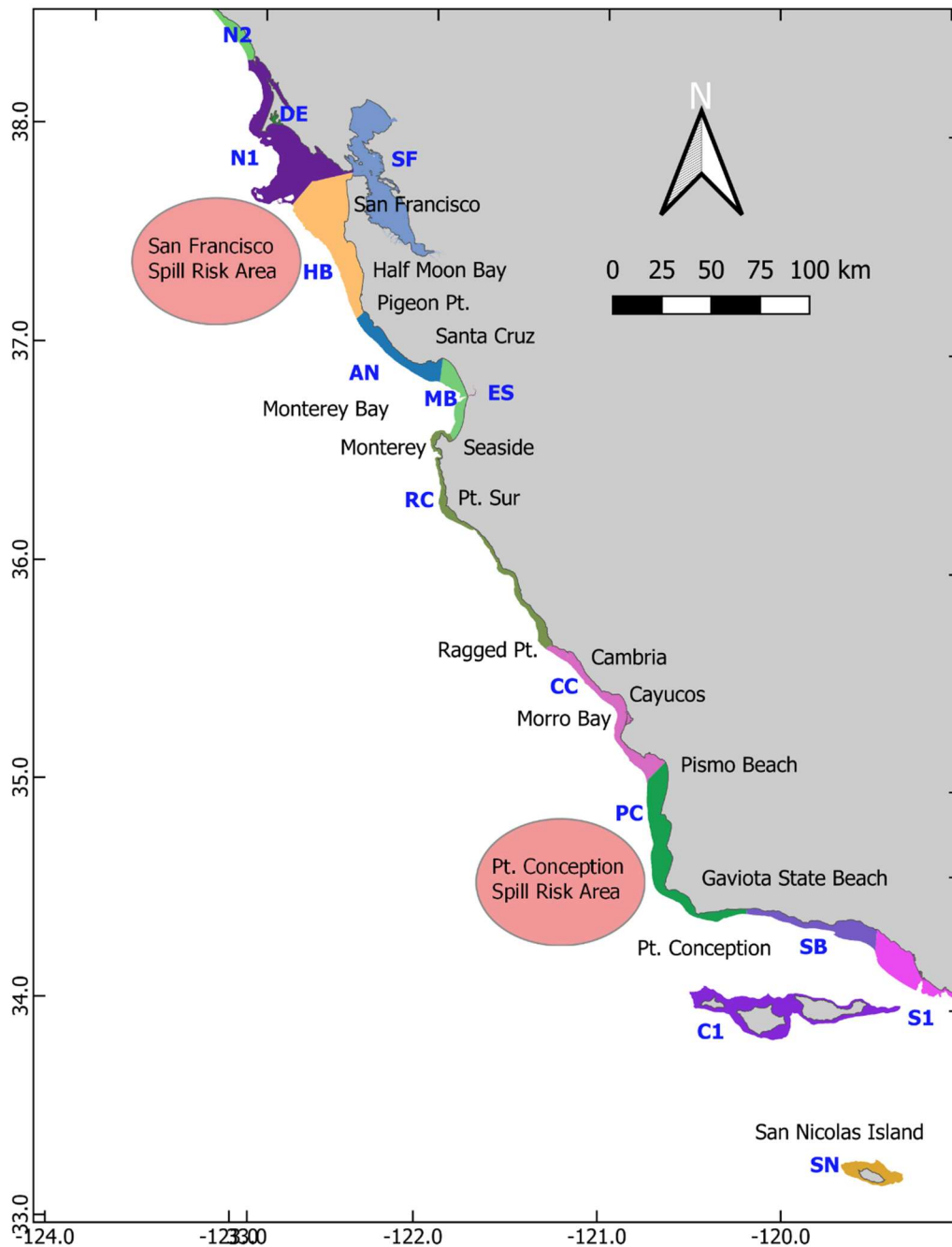


Figure 1. Map of the study area, the central coast of California. Potential habitat for southern sea otters is indicated by the band of colored polygons ("coastal sections"), identified by 2-letter codes (blue font). The distribution of sea otters in 2019 extended from Pigeon Pt. in the north (section AN) to Gaviota State Beach in the south (section PC), plus section SN (San Nicolas Island). Two high-risk areas for oil spills considered in this analysis are indicated by red ovals: the area offshore of San-Francisco at the north end of the range, and the area off Pt. Conception at the south end of the range.

Table 1. Summary of parameters used for simulating oil spills using “PyGnome” software. Large spills assumed to be 10 million gallons and medium spills assumed to be 1 million gallons of Alaskan Crude (with a point release at the specified location). Wind and current time series of data were downloaded for 1 month beginning at the date specified, with 15-minute time steps.

Sim #	Location	Season	Spill size	Date (wind/currents)	Latitude	Longitude
1	San Francisco	Winter	Large	November 1, 2015	37.450	-122.900
2	San Francisco	Winter	Large	November 1, 2015	37.550	-122.750
3	San Francisco	Winter	Large	December 1, 2015	37.450	-122.900
4	San Francisco	Winter	Large	December 1, 2015	37.550	-122.750
5	San Francisco	Winter	Large	October 1, 2016	37.450	-122.900
6	San Francisco	Winter	Large	October 1, 2016	37.550	-122.750
7	San Francisco	Winter	Medium	November 1, 2015	37.450	-122.900
8	San Francisco	Winter	Medium	November 1, 2015	37.550	-122.750
9	San Francisco	Winter	Medium	December 1, 2015	37.450	-122.900
10	San Francisco	Winter	Medium	December 1, 2015	37.550	-122.750
11	San Francisco	Winter	Medium	October 1, 2016	37.450	-122.900
12	San Francisco	Winter	Medium	October 1, 2016	37.550	-122.750
13	San Francisco	Summer	Large	July 1, 2014	37.450	-122.900
14	San Francisco	Summer	Large	July 1, 2014	37.550	-122.750
15	San Francisco	Summer	Large	June 1, 2016	37.450	-122.900
16	San Francisco	Summer	Large	June 1, 2016	37.550	-122.750
17	San Francisco	Summer	Large	August 1, 2016	37.450	-122.900
18	San Francisco	Summer	Large	August 1, 2016	37.550	-122.750
19	San Francisco	Summer	Medium	July 1, 2014	37.450	-122.900
20	San Francisco	Summer	Medium	July 1, 2014	37.550	-122.750
21	San Francisco	Summer	Medium	June 1, 2016	37.450	-122.900
22	San Francisco	Summer	Medium	June 1, 2016	37.550	-122.750
23	San Francisco	Summer	Medium	August 1, 2016	37.450	-122.900
24	San Francisco	Summer	Medium	August 1, 2016	37.550	-122.750
25	Pt. Conception	Winter	Large	November 1, 2014	34.800	-121.000
26	Pt. Conception	Winter	Large	November 1, 2014	35.000	-121.300
27	Pt. Conception	Winter	Large	December 1, 2014	34.800	-121.000
28	Pt. Conception	Winter	Large	December 1, 2014	35.000	-121.300
29	Pt. Conception	Winter	Large	February 1, 2015	34.800	-121.000
30	Pt. Conception	Winter	Large	February 1, 2015	35.000	-121.300
31	Pt. Conception	Winter	Medium	November 1, 2014	34.800	-121.000
32	Pt. Conception	Winter	Medium	November 1, 2014	35.000	-121.300
33	Pt. Conception	Winter	Medium	December 1, 2014	34.800	-121.000
34	Pt. Conception	Winter	Medium	December 1, 2014	35.000	-121.300
35	Pt. Conception	Winter	Medium	February 1, 2015	34.800	-121.000
36	Pt. Conception	Winter	Medium	February 1, 2015	35.000	-121.300
37	Pt. Conception	Summer	Large	July 1, 2014	34.800	-121.000

Sim #	Location	Season	Spill size	Date (wind/currents)	Latitude	Longitude
38	Pt. Conception	Summer	Large	July 1, 2014	35.000	-121.300
39	Pt. Conception	Summer	Large	July 1, 2015	34.800	-121.000
40	Pt. Conception	Summer	Large	July 1, 2015	35.000	-121.300
41	Pt. Conception	Summer	Large	May 1, 2016	34.800	-121.000
42	Pt. Conception	Summer	Large	May 1, 2016	35.000	-121.300
43	Pt. Conception	Summer	Medium	July 1, 2014	34.800	-121.000
44	Pt. Conception	Summer	Medium	July 1, 2014	35.000	-121.300
45	Pt. Conception	Summer	Medium	July 1, 2015	34.800	-121.000
46	Pt. Conception	Summer	Medium	July 1, 2015	35.000	-121.300
47	Pt. Conception	Summer	Medium	May 1, 2016	34.800	-121.000
48	Pt. Conception	Summer	Medium	May 1, 2016	35.000	-121.300

(Table 1, continued)

A key feature of the GNOME system is that oil spill weathering and dispersal/movement is determined by realistic physical drivers, including winds and currents. Historical records of winds and currents can be downloaded for the focal area, sampled from a range of time periods, thereby ensuring that the simulation results are representative and realistic. We accessed historical ocean current and wind data in a “GNOME-compatible” format using the GNOME Online Oceanographic Data Server, or “GOODS” (<https://gnome.orr.noaa.gov/goods>). For ocean currents we used the global Hybrid Coordinate Ocean Model (HYCOM) + NCODA Global Hindcast Analysis (<https://www.hycom.org>), with 1/12 degree resolution, downloaded from the HYCOM THREDDS server (<http://tds.hycom.org/thredds/catalog.html>). We created 30-day time series of gridded current data for the study area, with a 15-minute interval temporal resolution, initiated at arbitrary date ranges during summer months and winter months between 2014 and 2020 (Table 1). We then created time series of wind data at 15-minute intervals for those same date ranges using NOAA weather buoy data (Buoy 46012 for San Francisco area and Buoy 46011 for Pt. Conception area) downloaded from the National Data Buoy Center (NDBC) accessed through the Gnome GOODS site (<https://gnome.orr.noaa.gov/goods/winds/NDBC/NDBC>).

In addition to specifying spill location, size, winds and currents, the other parameters required to run oil spill simulations in PyGnome include: a coastal base map (we used the coastline from north of San Francisco to Ventura County), a random diffusion parameter ($RDP = 100000$: diffusion acts in conjunction with currents and winds to move/disperse oil), the type of oil (which determines properties such as rate of bio-degradation and evaporation; we assumed Alaskan Crude which has a low rate of weathering), spill duration (we assumed an instantaneous point release), and the number of discrete elements to track. The discrete elements, or “spots”, represent a specified volume (or mass) of oil, the amount of which is determined by dividing total spill volume at time of release by the number of spots (1000 by default). The spatial coordinates of each spot are tracked over time by GNOME, and the joint distribution of all spot positions defines the spatial extent of the surface oil slick at specified times after the spill (Figure 2). The mass of oil in each spot is also tracked; this is necessary because oil mass decreases as a function of weathering processes. In addition to the volume (or mass) and geographic position, the other key parameter tracked for each spot is its surface area (i.e., the size of the “patch of ocean” that the spot’s oil mass is spread out over). The effective radius of the surface area patch approaches 0 at the point of release but then increases over time as a function of dispersal and movement by wind and waves. Dividing oil volume by surface area gives the effective concentration (or surface thickness) of each spot at each point in time, information which is used to determine the capacity of a spot to cause significant oiling of any wildlife that occurs within its effective area, or sweep (French-McCay 2004).

For each of the scenarios listed in Table 1 we ran a 30-day simulation, as this period proved long enough to evaluate full potential impacts within the sea otter range. For each 30-day simulation we calculated the position of each oil spot at 15-minute intervals and saved the mean spot positions each hour, creating a time series of oil spill trajectories (Figure 2). The results were saved as NetCDF files, which were then used for the next step of estimating the intersection between surface oil slicks and sea otters.

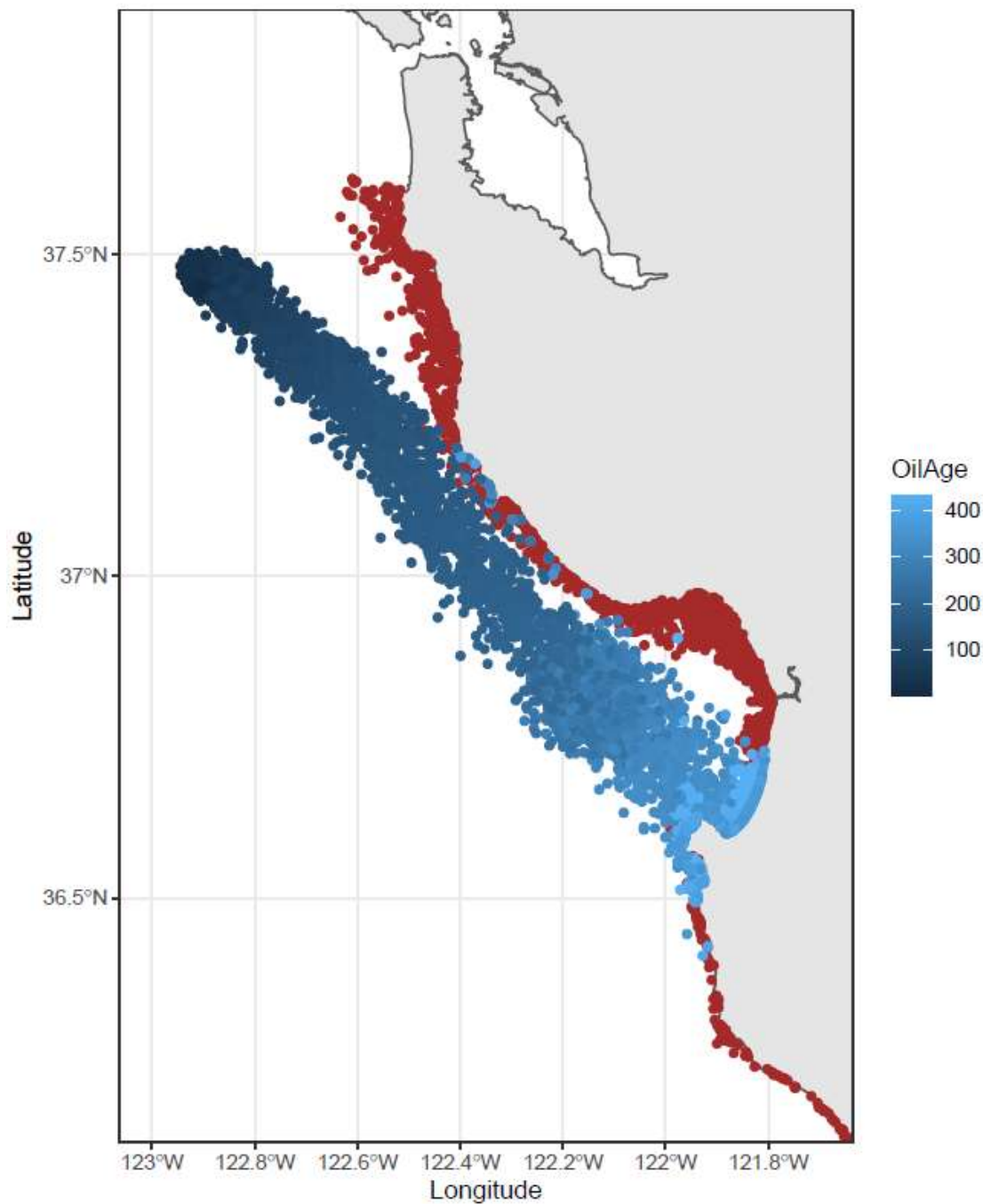


Figure 2. Sample oil spill trajectory (blue dots) predicted by a single simulation run in PyGnome. A hypothetical sea otter population is represented by red dots along the coastline, with the number of intersections of oil and sea otter points (see detail in Figure 3) indicating the extent of oil exposure to this spill scenario. Each blue dot represents an oil “splot”, a unit corresponding to 1000 gallons of crude oil at time of release in this example (assuming 1000 splots and 1 million gallon spill). The “age” of oil splots (hours since spill) is represented by color intensity (lighter blue = more time elapsed): for a given splot the radius of ocean affected increases and oil concentration decreases as a function of oil age.

Oil Spill – Sea Otter Intersections

To estimate the population-level impacts of an oil spill scenario, we developed a spatiotemporal intersection model similar to previous analyses of oil spill impacts to wildlife (e.g. Bodkin and Udevitz 1994, French-McCay 2004, Amstrup et al. 2006). The intersection model is simple in principle, as illustrated by the cartoon in Figure 3. The trajectory and spatial sweep of the oil spill is captured by the time series of location records of oil spots generated by PyGnome simulations (see above). Spots are also defined by their spatial extent (patch area) and oil concentration (units of g/m^2 of ocean surface) at each time step. The exposure of a hypothetical sea otter to oil during a single time step is calculated as the summed concentrations of spots whose patch areas overlap with the geographic position of the sea otter. If an otter's exposure level during any time step exceeds a threshold value, that otter is significantly oiled. Based on previous research we set the threshold level as $10 \text{ g}/\text{m}^2$, a level of oil exposure that was found to be associated with 75-99% mortality for marine birds and fur-bearing

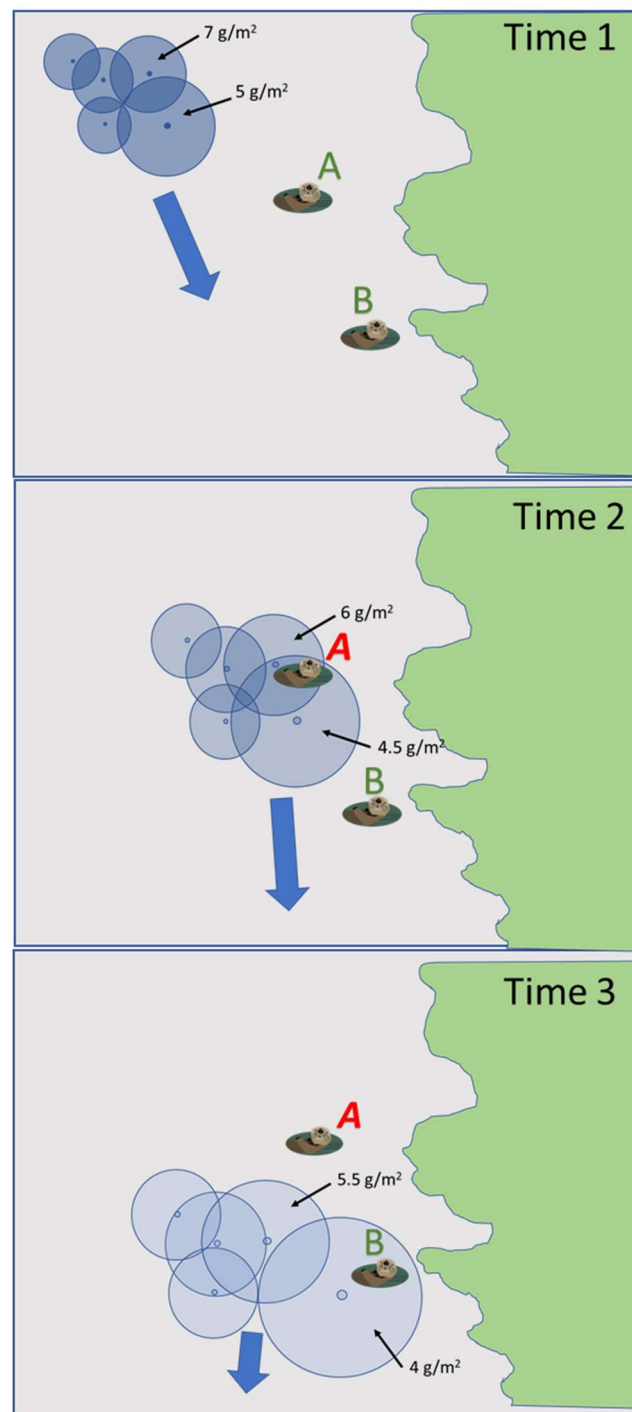


Figure 3. Illustration of oil spill/sea otter intersection analysis. At time 1, two otters (A and B) are in the path of an approaching oil slick, shown as 5 spots (with associated patch areas). At time 2, otter A is overlapped by 2 spots with combined density of $10.5 \text{ g}/\text{m}^2$, and so is considered oiled. At time 3, otter B is overlapped by 1 spot, with density $4 \text{ g}/\text{m}^2$, and so is not considered significantly oiled.

mammals (French-McCay 2004). We note that the intersection model calculates oil exposure, not mortality: in the IPM model (see next step) the user can specify the mortality rate for significantly oiled otters, though we note that mortality rates are generally assumed to be quite high (i.e., 90-100%) based on reports from previous oil spills (Williams et al. 1988, Ballachey et al. 2002, French-McCay 2004).

While simple in principle, implementing the intersection model was considerably more complex due to the computational challenges of variable and overlapping oil spot trajectories, uncertain distributions of sea otters, and large numbers of both. The fine-scale distribution of sea otters is variable from day to day, and at a courser scale the distribution changes over years as the population expands to the north and south. To accommodate these two sources of uncertainty in sea otter distributions, we made use of a recently-published habitat-based carrying capacity model for California sea otters (Tinker et al. 2021).

One of the results of the carrying capacity model is a spatial layer that describes localized relative density (i.e., the relative probability of finding a sea otter) at the scale of a 100m grid for all coastal California. For each of the coastal sections defined within the IPM (see Figure 1; coastal sections are used to describe demographic processes for sub-sets of the population) we used this spatial density layer within a Geographic Information System (GIS) to generate random spatial distributions of 1000 sea otters, such that the density of the distribution matched the model-predicted localized density at equilibrium. Two key advantages to this approach are 1) each randomly-generated distribution is reflective of observed distributions of sea otters over 35 years of surveys, as informed by the habitat layers such as depth, distance from shore, substrate type and kelp canopy cover (Tinker et al. 2021); 2) these distributions can be generated both for currently occupied areas *AND* for coastal sections that are not currently occupied but may be colonized within the foreseeable future (i.e. during IPM model projections). We created a program to intersect the output of the PyGnome oil spill simulations (the NetCDF files containing time series spot locations for each oil spill simulation) with the randomly generated distributions of sea otter positions in each coastal section (Figures 4-7).

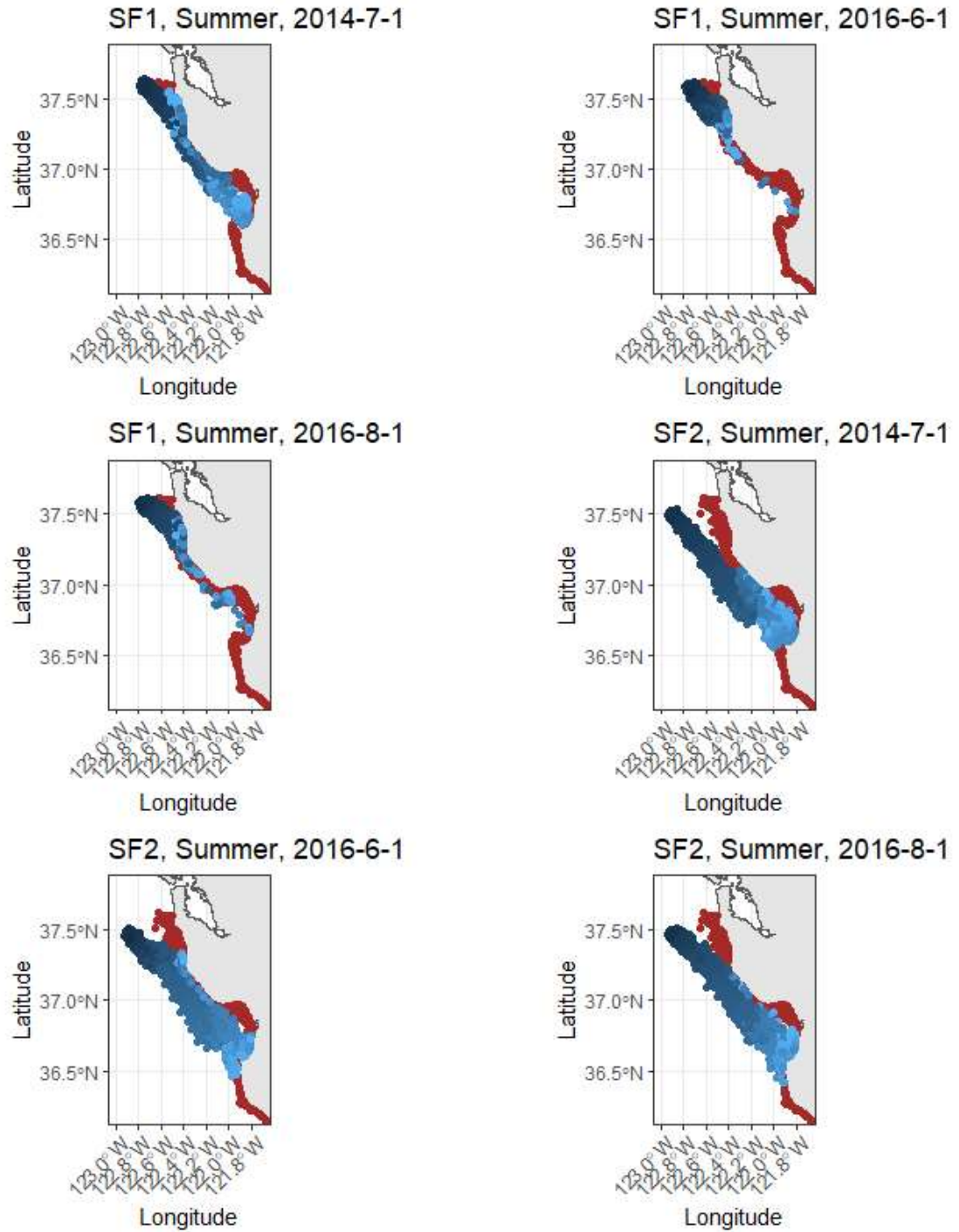


Figure 4. Maps of oil spill simulation results for San Francisco area during summer months. Each map shows a time-series of locations of 1000 oil splots (blue dots) predicted by a single simulation in PyGnome; the “age” of splots (hours since spill) is represented by color intensity (see Figure 2 for details). Randomly-generated sea otter locations are shown as red dots along the coastline: the number of intersections of oil splots and sea otter points (see detail in Figure 3) determines the extent of oil exposure to each spill scenario.

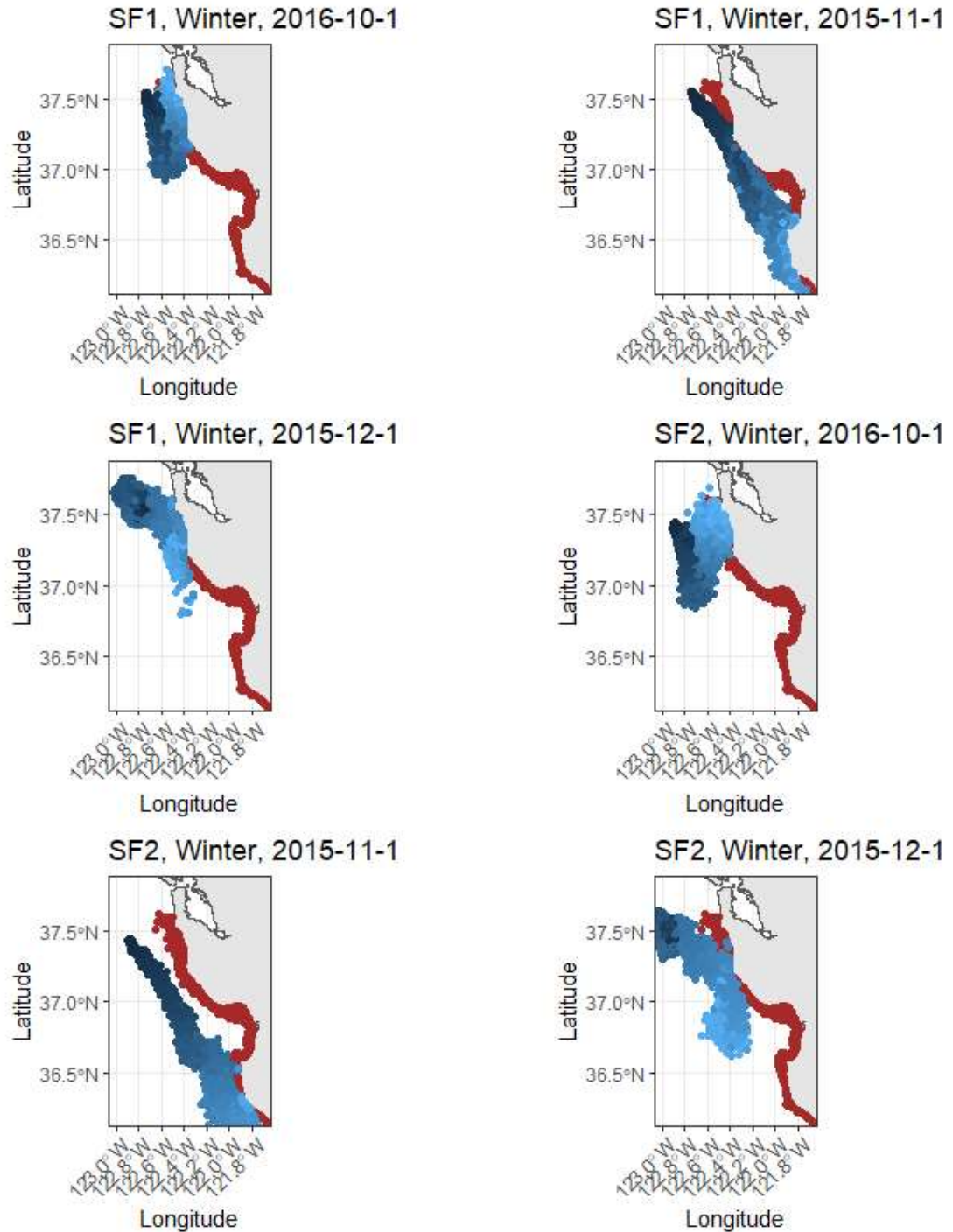


Figure 5. Maps of oil spill simulation results for San Francisco area during winter months. Each map shows a time-series of locations of 1000 oil splots (blue dots) predicted by a single simulation in PyGnome; the “age” of splots (hours since spill) is represented by color intensity (see Figure 2 for details). Randomly-generated sea otter locations are shown as red dots along the coastline: the number of intersections of oil splots and sea otter points (see detail in Figure 3) determines the extent of oil exposure to each spill scenario.

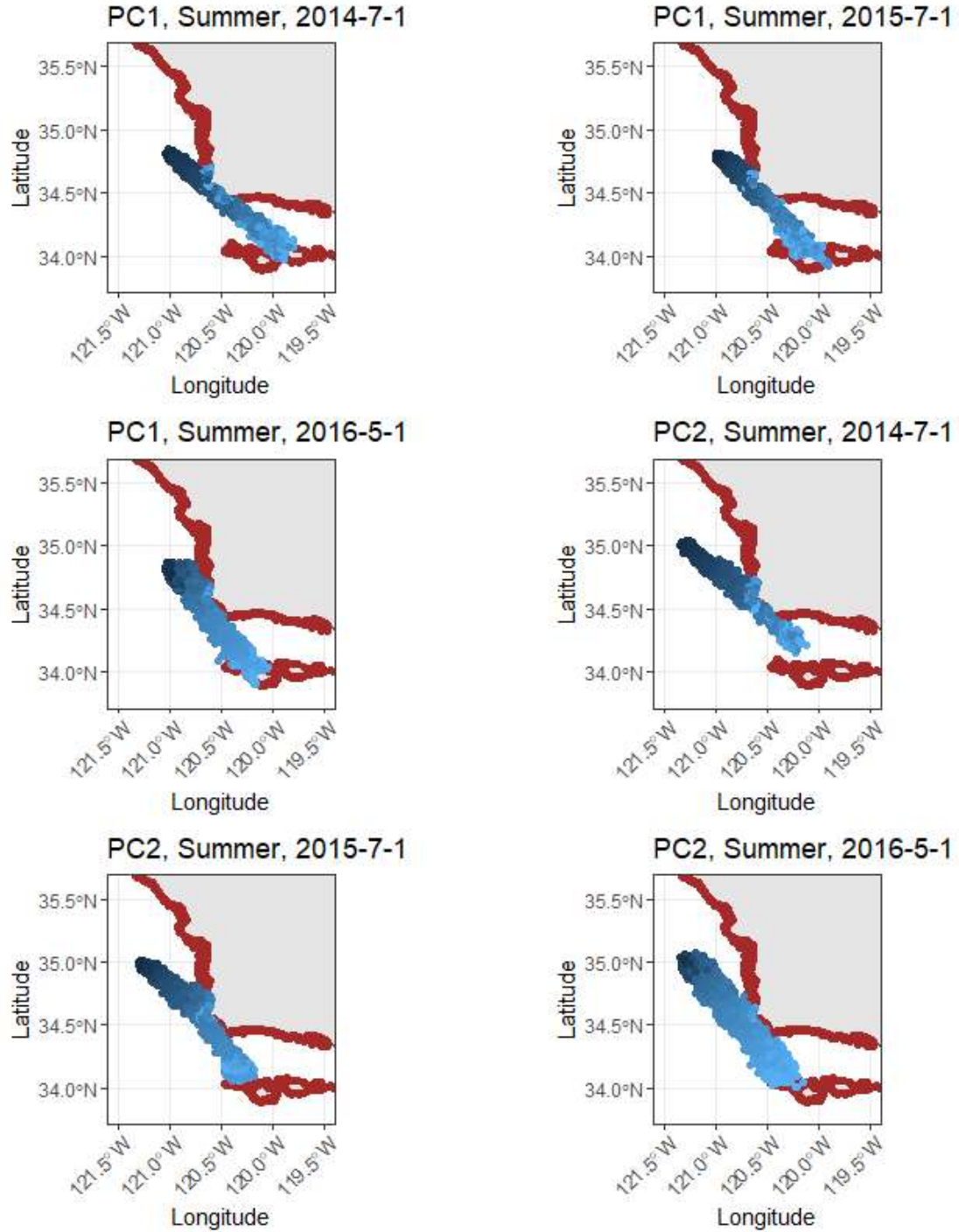


Figure 6. Maps of oil spill simulation results for Pt. Conception area during summer months. Each map shows a time-series of locations of 1000 oil splots (blue dots) predicted by a single simulation in PyGnome; the “age” of splots (hours since spill) is represented by color intensity (see Figure 2 for details). Randomly-generated sea otter locations are shown as red dots along the coastline: the number of intersections of oil splots and sea otter points (see detail in Figure 3) determines the extent of oil exposure to each spill scenario.

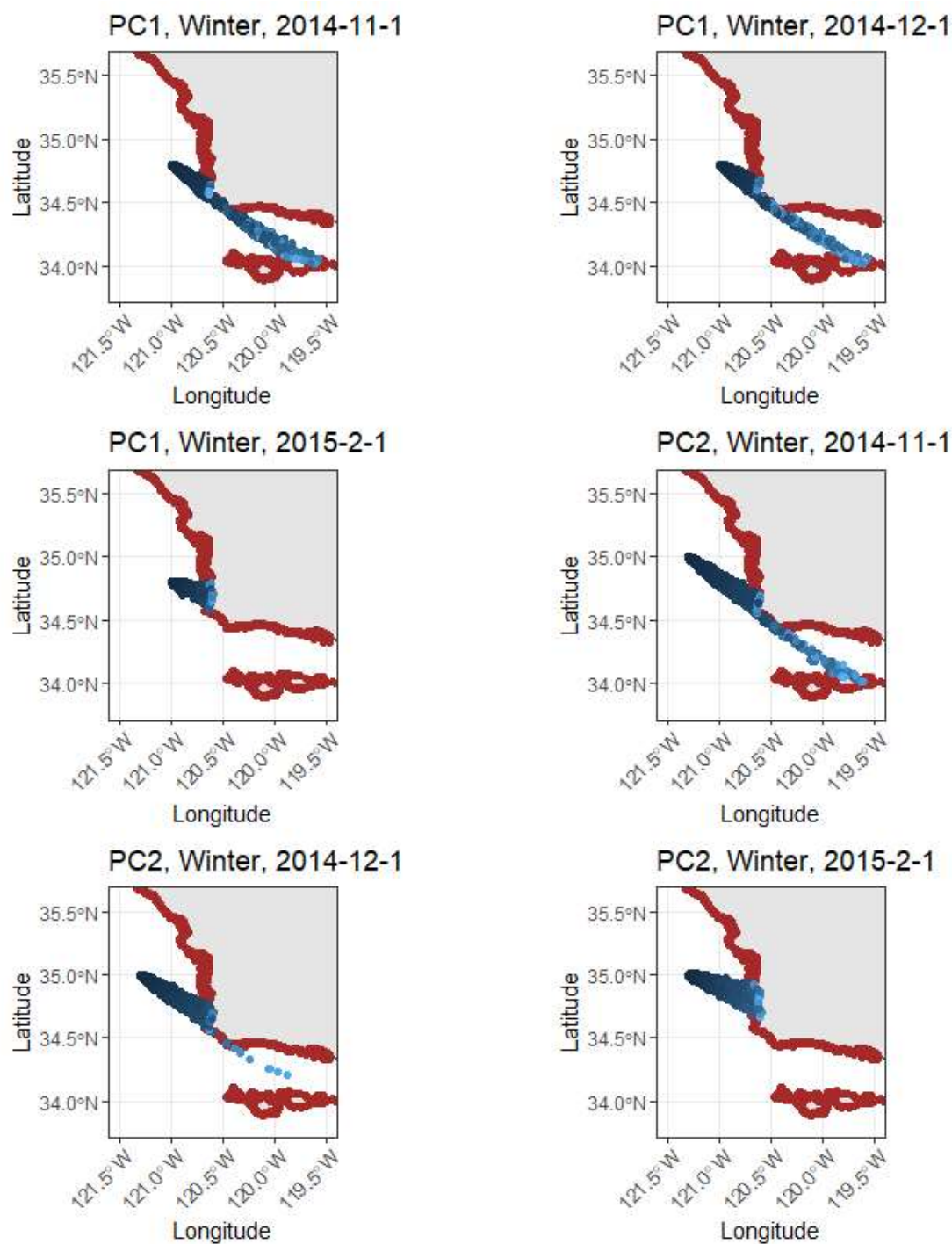


Figure 7. Maps of oil spill simulation results for Pt. Conception area during winter months. Each map shows a time-series of locations of 1000 oil splots (blue dots) predicted by a single simulation in PyGnome; the “age” of splots (hours since spill) is represented by color intensity (see Figure 2 for details). Randomly-generated sea otter locations are shown as red dots along the coastline: the number of intersections of oil splots and sea otter points (see detail in Figure 3) determines the extent of oil exposure to each spill scenario.

For each simulated oil spill (Figures 4-7), we sequentially stepped through the 1-hour time steps, and for the 1000 splots and 1000 simulated sea otter positions we generated a pairwise distance matrix (using the “pdist” function in R). For each otter we summed the oil concentrations for all the splots whose effective radius was greater than the distance between the splot and the otter (such that the patch of ocean affected by that splot would overlap with the otter: Figure 3). If the summed concentration of overlapping splots exceeded the threshold value, that otter was tagged as oiled. We then repeated the above calculations for each hourly time step, and at the end of the time series we tabulated the proportion of otters in each coastal section that were significantly oiled at one or more times during the 30-day time series.

For each of the 8 unique oil spill scenarios (as defined by location, season, and spill size) we tabulated the mean and standard deviation of the proportion of otters expected to be oiled within each coastal section, based on the distribution of results for multiple oil spill simulations and intersection analyses (Table 2). We also fit beta distributions to the results, the parameters of which could be used to generate stochastic “proportion-oiled” values for population simulations using the IPM (see next step). Specifically, by drawing random “proportion-oiled” values from the appropriate beta distributions for a given oil spill scenario and coastal section, and multiplying that proportion by the current number of otters in that section, one can generate stochastic distributions for the expected number of oiled otters. Applying this approach to the most recent survey estimates (Hatfield et al. 2019) for the three northernmost occupied coastal sections (AN, MB and RC; Figure 1), we can project that a large oil spill (10 million gallons) occurring near San Francisco during summer months would be expected to result in significant oiling of 242 otters, with 95% quantiles of 56 – 522 (Figure 8). We note that these values only account for initial oil exposure, and do not account for subsequent mortality that may occur due to chronic effects of continued oil exposure after the spill: we discuss this consideration in the next step.

Table 2. Summary of the means and standard errors (in parentheses) of the proportion of otters exposed to significant concentrations of oil after a spill, for each of 8 scenarios (rows) and for each of 8 coastal sections (columns). Refer to Figure 1 for locations of coastal sections.

Sc.#	Region	Season	Size	Proportion Oiled by Coastal Section							
				HB	AN	MB	RC	CC	PC	SB	C1
1	North	Summer	Large	0.55 (0.43)	0.38 (0.36)	0.24 (0.25)	0.12 (0.14)	-	-	-	-
2	North	Summer	Med	0.30 (0.32)	0.07 (0.12)	0.07 (0.10)	0.06 (0.06)	-	-	-	-
3	North	Winter	Large	0.59 (0.40)	0.16 (0.17)	0.02 (0.04)	0.17 (0.26)	-	-	-	-
4	North	Winter	Med	0.30 (0.35)	0.06 (0.12)	0.01 (0.01)	0.01 (0.02)	-	-	-	-
5	South	Summer	Large	-	-	-	-	0.01 (0.01)	0.26 (0.10)	0.01 (0.01)	0.21 (0.20)
6	South	Summer	Med	-	-	-	-	0.01 (0.01)	0.06 (0.05)	0.01 (0.01)	0.00 (0.00)
7	South	Winter	Large	-	-	-	-	0.01 (0.01)	0.23 (0.10)	0.01 (0.01)	0.01 (0.00)
8	South	Winter	Med	-	-	-	-	0.01 (0.01)	0.09 (0.05)	0.01 (0.01)	0.01 (0.01)

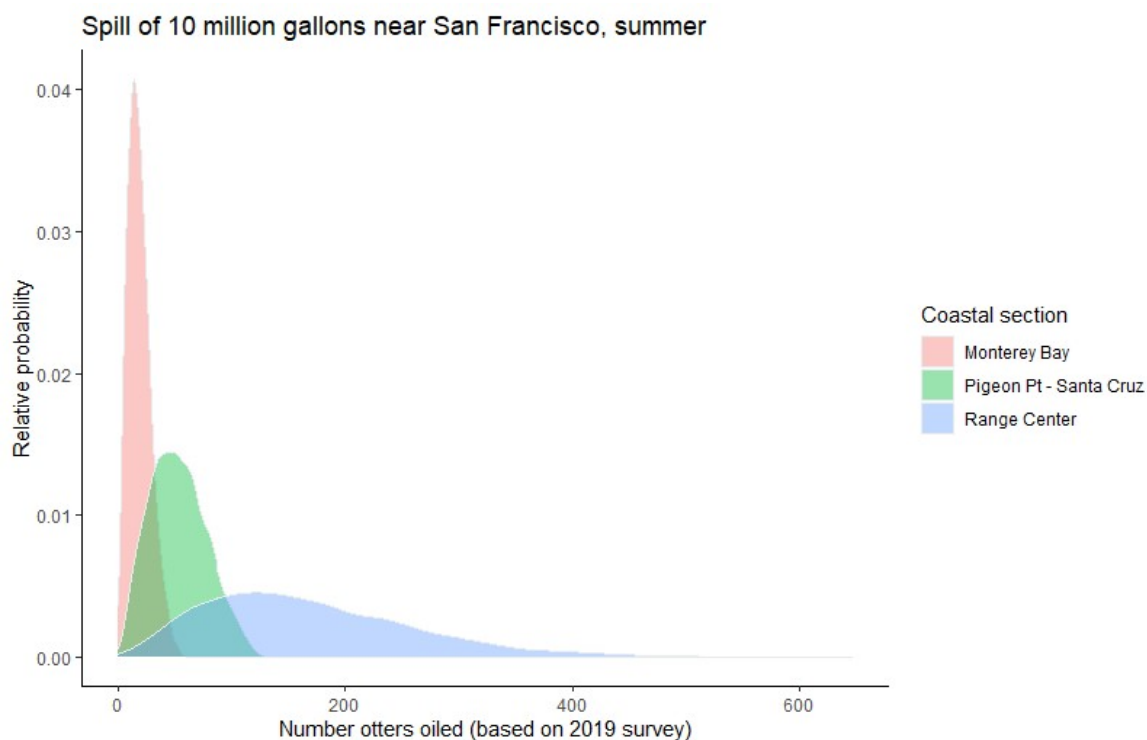


Figure 8. Frequency distributions for the expected number of otters that would be oiled (exposed to a slick of $>10 \text{ g/m}^2$) in each of 3 coastal sections after a large oil spill occurring near San Francisco during summer months. Refer to Figure 1 for locations of coastal sections.

Incorporating Oil Spill Simulation Results into the IPM

A detailed description of the methods used to develop the Integrated Population Model (IMP), including model structure and parameterization, are provided elsewhere (Tinker et al. in press). Here, we describe how the oil spill simulation results were incorporated into the IPM, allowing for estimation of oil spill impacts on future projections of population growth and range expansion. A new user-interface panel was added to the existing web-based IPM interface (Figure 9), allowing a user to adjust several parameters related to oil spill simulations, and thereby evaluate acute and chronic effects of major oil spills under different assumptions and conditions. For both “baseline” and “alternative” model projection scenarios, the user can adjust 1) whether major oil spills can (or will) occur; 2) the size of spill to simulate (Large or Medium) as well as season and region, where “North” corresponds to the San Francisco Bay risk area and “South” corresponds to the Pt. Conception risk area (the user can also select “Random”, to get a random selection of location, season, and spill size); 3) the acute death rate for otters that are significantly oiled (encounter an oil concentration $> 10 \text{ g/m}^2$); and 4) the approximate frequency with which major oil spills are expected to occur. The latter setting applies to simulations in which oil spills are probabilistic (i.e., a spill can occur on any year with a probability corresponding to the user-set frequency): oil spill occurrence can also be “forced”, in which case it is assumed to occur 1/3 of the way through the model projection. Based on these user-selected settings, IPM simulations are run and compared for baseline vs. alternative scenarios. We note that other factors (e.g. cause-specific hazard rates, re-introductions, changes to environmental settings) can also be adjusted, so that oil spill impacts can be considered in conjunction with other expected changes or management actions.

For IPM simulations in which an oil spill occurs, the spill location, size and season are randomly selected (unless these parameters have been set as “fixed” by the user). The proportion of otters in each coastal section that is exposed to oil is then assigned stochastically by drawing a random value from the beta distribution associated with the specified oil spill scenario (see previous section for details).

Run Simulations Now

Create a Scenario and Run IPM simulations (baseline vs. alternative scenario) **NOTE: Please review all Modification Tabs before running simulations**

Number of years to simulate

10 19 28 37 46 50 55 64 73 82 91 100

Number of times to replicate simulations

20 100 1,000

Adjust Mortality Adjust Distribution Adjust Environment **Oil Spills**

Evaluate acute and chronic effects of major oil spills under different scenarios, described by region (North = HB, South = PC), season and size of the oil spill. 'Large' spill equals 10 million gallons, 'Medium' spill equals 1 million gallons. NOTE: spill events can be set as deterministic (forced) or random/probabilistic, and scenarios can be specified and compared between baseline and alternative model runs. Leaving checkbox blank runs simulations with no MAJOR oil spills.

☐ Set Baseline Oil Spill Scenario?

☐ Set Alternative Oil Spill Scenario?

Baseline Oil Spill Scenario

None / Random

Alternative Oil Spill Scenario

None / Random

Baseline Spill Frequency: Avg. #spills/100yrs ('0'= none or forced)

0 1 2 3 4 5 6 7 8 9 10

Alternative Spill Frequency: Avg. #spills/100yrs ('0'= none or forced)

0 1 2 3 4 5 6 7 8 9 10

Death rate for otters 'significantly' oiled (encountering oil conc. > 10g/m2)

0.5 0.95 1

Map of Coastal Areas

Simulation Results, Trend Plots

Simulation Results, Heat Maps

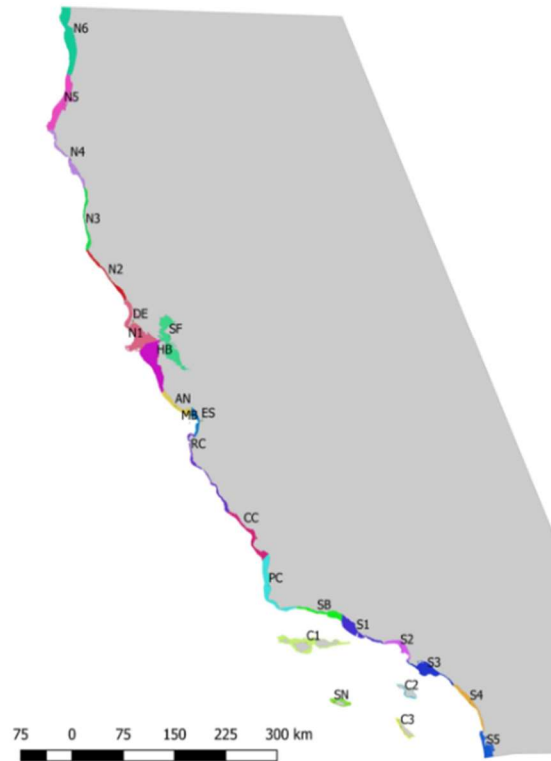


Figure 9. Screen shot of the web-based interface of the IPM, showing the new “Oil Spills” tab with user-adjustable parameters.

The randomly selected proportion of otters oiled in each coastal section is then multiplied by the projected number of otters in that section at the time of the spill, and this product is multiplied by the user-specified “oiled otter mortality rate”, to obtain the total number of acute mortalities from oil exposure. These oil-related deaths are proportionally allocated among age/sex classes according to abundance (i.e., it is assumed that all otters are equally at risk from oil exposure), and dead animals are subtracted from the population vector prior to other hazards being applied (i.e., it is assumed that acute oil mortalities are independent from and thus additive to other hazards).

In addition to the acute mortality caused by exposure to oil during the spill event, follow-up studies of previous oil spills suggest that there are also likely to be chronic effects that can last years after the initial spill (Monson et al. 2000, 2011). These longer-term effects may be caused in part by chronic health impacts to individual animals based on their initial exposure (Ballachey et al. 2002), and in part by continued exposure by foraging sea otters to residual oil in the environment (e.g. by digging up Infaunal prey from contaminated sediments). Predicting exactly what chronic effects will occur in California is impossible, so to be conservative we used the *Exxon Valdez* example to inform our estimates of longer-term impacts. Monson et al. (2000) found that in the core area of Prince William Sound affected by the *Exxon Valdez* spill, prime age survival was reduced by approximately 50% initially and then slowly increased back to baseline levels after approximately 10 years. Accordingly, we used the point estimate of the proportion of individual otters oiled in each coastal section (P_{oiled}) as an estimate of the proportion of the population in that section that would suffer long-term impacts. The first year after a spill we multiplied the baseline survival rates for all age classes by $0.5 * P_{oiled}$, thereby assuming that those animals in the affected area would experience a 50% reduction in survival. The following year the multiplier for survival was $0.55 * P_{oiled}$, the year after that it was $0.6 * P_{oiled}$, and so on until after 10 years survival rates had returned to their baseline levels. In this way, the chronic effects on survival were scaled to the magnitude of spill impacts in each coastal section.

The results of the IPM simulation model reflect both acute mortality from initial oil exposure and longer-term chronic effects, as described above. These results suggest that the cumulative impacts on population trends could be quite substantial, depending on where and when a spill occurred. For example, Figure 10a shows a comparison of projected population trends (range-wide) for southern sea otters under baseline conditions vs. an alternative scenario where a large oil spill occurs near San Francisco in the summer of 2037 (1/3 of the way through the 50-year projection period). In this scenario the initial spill is projected to kill approximately 275 animals on average (assuming 95% mortality for significantly oiled otters), however the chronic effects on survival lead to reduced growth so that after 6 years the projected population size for the oil-spill scenario is 572 animals lower (on average) than the equivalent baseline scenario where no oil spill occurred. In contrast, a medium size oil spill near Pt. Conception in the winter of 2037 would have much less impacts on range-wide population trends (Figure 10b). In this case the initial loss from acute mortality was estimated to be only 43 animals on average, and the reduction in abundance (relative to baseline) after 6 years was 114 animals on average.

The effects of a catastrophic oil spill on sea otter populations can be substantial, with full recovery to pre-spill population status taking years to decades (Garrott et al. 1993, Monson et al. 2000, 2011, Ballachey et al. 2014). Directly translating the results of the *Exxon Valdez* oil spill to the California sea otter population is problematic, as the geography and physical environment of these regions is vastly different. Our approach here, combining spatially explicit simulation models of the wind- and current-base dispersal of oil after a major spill in California (using the GNOME oil spill simulation environment) with a spatially explicit and demographically structured projection model of the southern sea otter population, provides an analytically sound and transparent method for evaluating potential oil spill impacts on sea otter recovery in California.

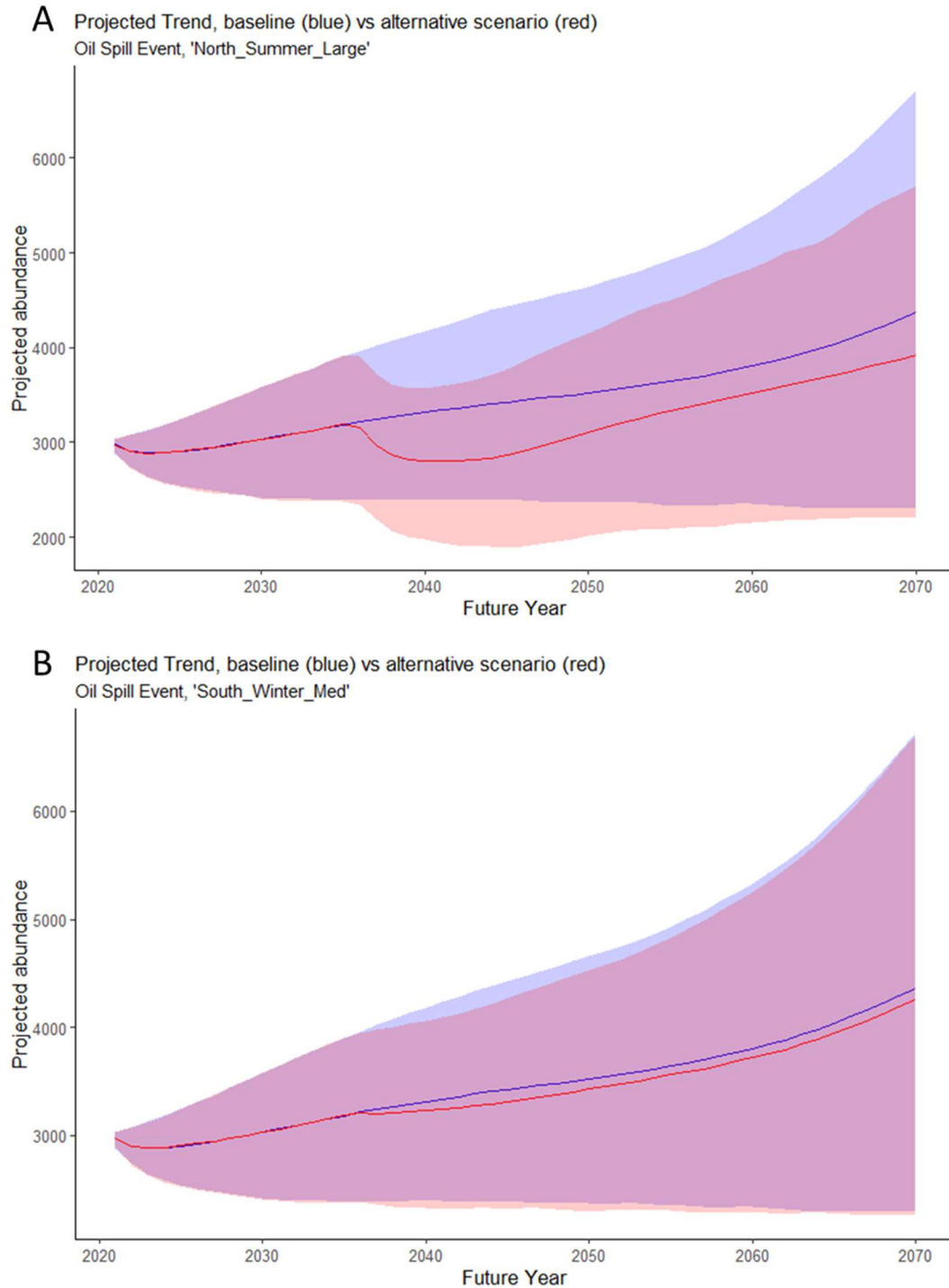


Figure 10. Projected abundance over time of the southern sea otter population under a baseline scenario (blue) vs. alternative scenarios (red) in which a catastrophic oil spill event occurs. A) an alternative scenario of a large oil spill event (10 million gallons) near San Francisco during the summer of 2037. B) an alternative scenario of a medium oil spill event (1 million gallons) near Pt. Conception during the winter of 2037. Lines correspond to mean expected abundance and shaded ribbons correspond to 80% confidence intervals for expected abundance.

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