SOUTHERN SEA OTTER (*Enhydra lutris nereis*)

U.S. Fish and Wildlife Service, Ventura, California

STOCK DEFINITION AND GEOGRAPHIC RANGE

Southern sea otters occupy nearshore waters along the mainland coastline of California from San Mateo County to Santa Barbara County (Figure 1). A subpopulation of southern sea otters also exists at San Nicolas Island, Ventura County, as a result of translocation efforts initiated in 1987.

Figure 1. Current range of the southern sea otter (2019 census). Source: Hatfield et al. (2019).

Historically, southern sea otters ranged from present-day Punta Abreojos, Baja California, Mexico, to at least as far north as southern Oregon (Valentine et al. 2008). Sea otter mitogenomes from northern Oregon more closely resemble those of northern sea otters, suggesting the historical existence of a transitional zone or latitudinal cline, with gene flow occurring both to the north and south (Larson et al. 2012, Wellman et al. 2020). The killing of sea otters for their pelts during the fur trade of the 18th and 19th centuries extirpated the subspecies throughout most of its range. A small number of southern sea otters survived near Bixby Creek in Monterey County, California (Bryant 1915). Since receiving protection under the International Fur Seal Treaty in 1911, southern sea otters have gradually expanded northward and southward along the central California coast, reclaiming approximately 13 percent of their historical range (U.S. Fish and Wildlife Service [USFWS] 2015). Range expansion is of primary importance for reestablishing the subspecies and restoring the nearshore marine ecosystems of which southern sea otters were once a part (USFWS 2003, USFWS 2015). The estimated carrying capacity of California is 17,226 otters (95% credible interval = 9,739–30,087) (Tinker et al. 2021). The carrying capacity of the remainder of the southern sea otter's historical range has not been determined. Sea otter abundance varies considerably across the range, with the highest densities generally occurring in the rocky, kelp-dominated central portion (Seaside to Cayucos), where sea otters have been present the longest. A notable exception to this pattern is the Elkhorn Slough estuary, which supports the highest densities of sea otters within the southern sea otter range (Tinker et al. 2018, 2021). Densities in the northern and southern portions of the range (specifically Davenport to north Monterey Bay in the north and Pismo Beach to Lompoc in the south) are much lower, consistent with high levels of shark bite mortality (Tinker et al. 2016, Hatfield et al. 2019, Moxley et al. 2019). These areas tend to lack sufficient kelp cover (either due to substrate type or to environmental conditions that have caused kelp declines) and have increasingly become high-risk areas for shark-bite mortality (Nicholson et al. 2018).

All sea otters of the subspecies *Enhydra lutris nereis* are considered to belong to a single stock because of their recent descent from a single remnant population. Southern sea otters are geographically isolated from the other two recognized subspecies of sea otters, *E. l. lutris* and *E. l. kenyoni*, and have been shown to be distinct from these subspecies in genetic (Sanchez 1992, Cronin et al. 1996, Larson et al. 2002) and morphometric studies, although some phenotypic traits vary along a latitudinal cline (Wilson et al. 1991, Wellman 2018).

POPULATION SIZE

Data on population size have been gathered for more than 50 years. 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 conducted each spring. At San Nicolas Island, counts are conducted from shore quarterly, with the spring count taken as the official count for the year. Because the spring count produces uncorrected totals, the resulting metric is an index of population size rather than a true estimate of abundance. 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. In 2019, the range-wide index of abundance was 2,962 (Hatfield et al. 2019).

Minimum Population Estimate

The minimum population estimate for the southern sea otter stock is taken as the lesser of the latest combined raw counts from the mainland range and San Nicolas Island or the latest 3-year running average of the combined counts. In 2019, the combined raw count was 3,117, which is higher than the combined 3-year running average of 2,962. Therefore, the minimum population estimate is 2,962 animals (2,863 along the mainland and 99 at San Nicolas Island).

Current Population Trend

As recommended in the Final Revised Recovery Plan for the Southern Sea Otter (U.S. Fish and Wildlife Service 2003), 3-year running averages are used to characterize trends to dampen the effects of anomalous counts in any given year. Based on 3-year running averages of the annual spring counts, the rangewide (combined mainland and island) population growth trend over the 5-year period from 2015 to 2019 (inclusive) is flat at 0.12 percent per year (Hatfield et al. 2019; Figure 2).

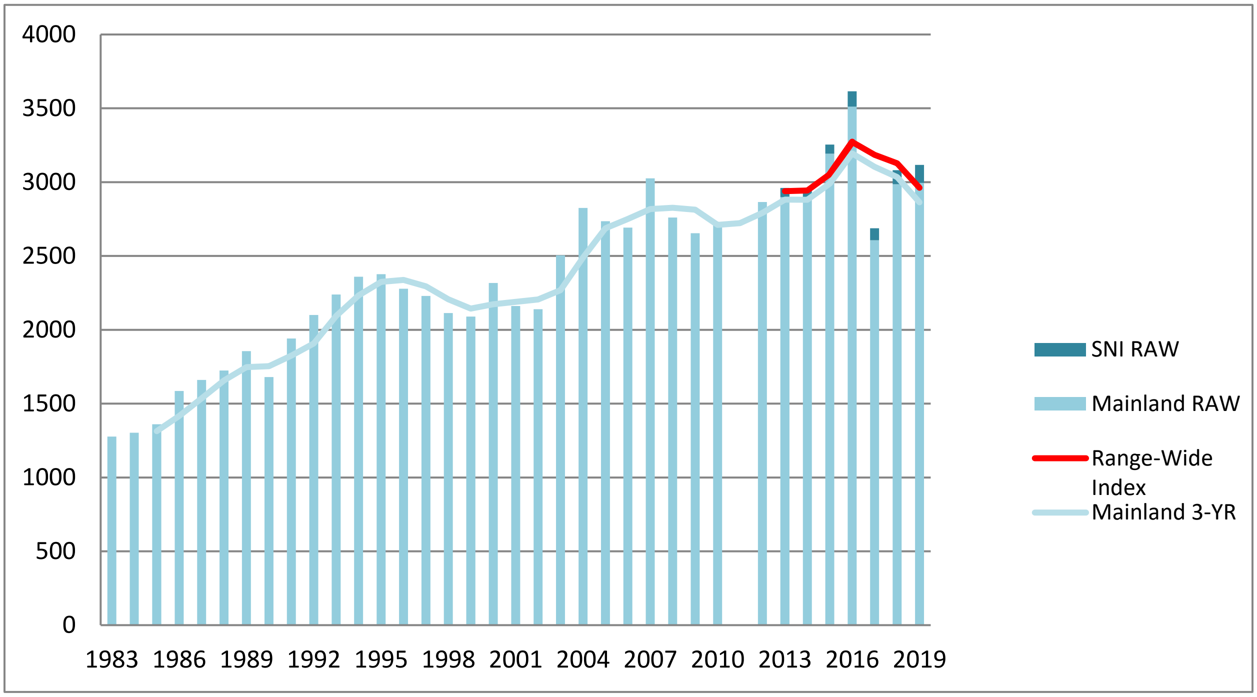

Figure 2. Southern sea otter counts 1983–2019. Bars show raw counts for each year, whereas lines represent 3-year running averages. The annual census was not completed in 2011 (due to weather) or 2020 (due to COVID-19 restrictions).

Regionally and inter-annually, counts can exhibit high variability. In the center portion of the mainland range (Seaside to Cayucos), population growth over the 2015–2019 period has averaged 2.4 percent per year. This positive trend is believed to be due to an increase in prey availability (sea urchins and mussels) over the past several years resulting from the absence of the predatory sunflower star, *Pycnopodia helianthoides*, due to sea star wasting disease. The effects of that prey subsidy now appear to be diminishing (Hatfield et al. 2019).

In the portion of the mainland range to the north of the central region (Pigeon Point to Seaside), the trend remains negative, with a decline over the 2015–2019 period averaging –8.7 percent per year (Hatfield et al. 2019). In the southern portion of the mainland range (Cayucos to Gaviota), the 5-year trend is also negative, averaging –1.6 percent per year (Hatfield et al. 2019). The regional trends in the northern and southern portions of the mainland range are consistent with continuing high levels of shark-bite mortality in these areas, which appears to be preventing range expansion (Hatfield et al. 2019).

The small subpopulation at San Nicolas Island continues the strong growth trend it has exhibited since approximately 2010. Annual growth has averaged 9.6 percent over the 2015–2019 period.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

We use the 5-year population trend to characterize current net productivity rates. During the 2015–2019 period, the trend of the mainland population was flat, averaging 0.12 percent per year, whereas growth of the San Nicolas Island population averaged 9.6 percent per year (Hatfield et al. 2019). Because most of the population occurs along the mainland coastline, the rangewide population growth trend is heavily influenced by the mainland population trend.

The maximum intrinsic growth rate of sea otters is between 0.20 and 0.25 (Estes 1990, Tinker 2015, Tinker et al. 2019*a*). Recovering or translocated populations of northern sea otters (*E. l. kenyoni*) at Attu Island, southeast Alaska, British Columbia, and Washington state all exhibited growth rates of up to 17 or 20 percent annually during the early stages of recovery (Estes 1990, Jameson and Jeffries 1999, Jameson and Jeffries 2005). Portions of these populations are now growing much more slowly in areas that are approaching carrying capacity, but subpopulations in more recently colonized areas continue to grow rapidly, and the area of occupied range and total population size have increased substantially (Jeffries et al. 2019, Tinker et al. 2019*a*, Nichol et al. 2020).

Reliable records of early population growth in California are not available. A localized subpopulation at the southern terminus of the mainland range (Cojo Anchorage) was observed to increase at an annual instantaneous growth rate of 0.19 (95% CL = 0.063–0.314) soon after recolonization during the period 2004–2013, though pups were seen only beginning in 2010 and sea otter numbers were seasonally variable, indicating that observed growth represented immigration in addition to births (Lafferty and Tinker 2014). For the California mainland range as a whole, the highest observed growth rate for any 5-year period since the early 1980s, when comparable trend data first become available, is 0.076 per year (95% CL = 0.066–0.086) during 1985–1989.[1] The highest 5-year annual growth rate recorded for the subpopulation at San Nicolas Island, is 0.192 (95% CL = 0.149–0.236) for 2009–2013.[2]

Excluding declines that occurred during periods of unusually elevated mortality (such as those caused by gill-net entanglements during the late 1970s and early 1980s [see Human-Caused Mortality and Serious Injury section] 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 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; 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).

[1] Personal communication, Julie Yee, 2021. U.S. Geological Survey–Western Ecological Research Center, 2885 Mission Street, Santa Cruz, California 95060.
[2] Personal communication, Julie Yee, 2021. U.S. Geological Survey–Western Ecological Research Center, 2885 Mission Street, Santa Cruz, California 95060.

POTENTIAL BIOLOGICAL REMOVAL

Potential Biological Removal (PBR) is the product of three elements: the minimum population estimate (N_{min}); half the maximum net productivity rate (0.5 R_{max}); and a recovery factor (F_r). This can be written as: PBR = (N_{min})(½ of R_{max})(F_r).

For the southern sea otter stock, N_{min} is 2,962 (2,863 along the mainland and 99 at San Nicolas Island). Because the maximum population growth rate appears to be tightly constrained by habitat configuration, we use an R_{max} of 0.076 for the mainland portion of the population and an R_{max} of 0.192 for the island portion of the population. We use a recovery factor of 0.1 for the southern sea otter stock because N_{min} is below 5,000 and the species is vulnerable to a natural or human-caused catastrophe, such as an oil spill, due to its restricted geographic distribution in nearshore waters (Taylor et al. 2003). Therefore, the PBR for the southern sea otter stock is 12 [(2,863 x 0.5 x 0.076 x 0.1) + (99 x 0.5 x 0.192 x 0.1)].

This PBR number should be interpreted with caution. The formula used to calculate PBR is based on the assumption that a depleted stock will naturally grow toward OSP and that some surplus growth may be removed while still allowing recovery (NMFS 2016). However, the southern sea otter stock does not meet this assumption because the stock as a whole is well below its estimated OSP (see Status of Stock section) and is not growing, yet human-caused mortality is a not a major factor in the population's trend. Instead, natural factors, predominantly shark bite mortality, are driving population trends. It is also important to note that take of southern sea otters incidental to commercial fishing operations cannot be authorized under the MMPA. Thus, the provisions governing the authorization of incidental take in commercial fisheries at MMPA Sections 101(a)(5)(E) and 118, which include requirements to develop take reduction plans with the goal of reducing incidental mortality or serious injury of marine mammals to levels less than the PBR, do not apply to southern sea otters.

HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Fishery Information

Sea otters are susceptible to entanglement and drowning in gill nets. The set gill net fishery in California is estimated to have killed from 48 to 166 (average of 103) southern sea otters per year from 1973 to 1983 (Herrick and Hanan 1988) and 80 sea otters annually from June 1982 to June 1984 (Wendell et al. 1986). A 1991 closure restricted gill and trammel nets to waters deeper than 30 fathoms (55 meters) throughout most of the southern sea otter's range (California Senate Bill No. 2563). In 1990, NMFS started an observer program using at-sea observers, which provided data on incidental mortality rates relative to the distribution of fishing effort. The observer program was active through 1994, discontinued from 1995 to 1998, and reinstated in the Monterey Bay area in 1999 and 2000 because of concern over increased harbor porpoise mortality. Based on a detailed analysis of fishing effort, sea otter distributions by depth, and regional entanglement patterns during observed years, NMFS estimated southern sea otter mortality in the halibut set gill net fishery to have been 64 in 1990, zero from 1991 to 1994, 3 to 13 in 1995, 2 to 29 in 1996, 6 to 47 in 1997, 6 to 36 in 1998, 5 in 1999, and zero in 2000 (Cameron and Forney 2000; Carretta 2001; Forney et al. 2001). The increase in estimated mortality from 1995 to 1998 was attributed to a shift in set gill net fishing effort into areas where sea otters are found in waters deeper than 30 fathoms (55 meters).

Fishing with gill nets has since been further restricted throughout the range of the southern sea otter. An order prohibiting the use of gill and trammel nets year-round in ocean waters of 60 fathoms or less from Point Reyes, Marin County, to Point Arguello, Santa Barbara

County was made permanent in September 2002. In the waters south of Point Arguello, the Marine Resources Protection Act of 1990 (California Constitution Article 10B) defined a Marine Resources Protection zone in which the use of gill and trammel nets is banned. This zone includes waters less than 70 fathoms (128 meters) or within one nautical mile (1.9 kilometers), whichever is less, around the Channel Islands, and waters generally within three nautical miles (5.6 kilometers) offshore of the mainland coast from Point Arguello to the Mexican border. Although sea otters occasionally dive to depths of 328 feet (100 meters), the vast majority (>99 percent) of dives are to depths of 131 feet (40 meters) or less (Tinker et al. 2006a). Because of these restrictions and the current extent of the southern sea otter's range, southern sea otter mortalities resulting from entanglement in gill nets are likely to be at or near zero. Nevertheless, sea otters may occasionally transit areas that are not subject to closures, and levels of observer coverage of gill and trammel net fisheries are insufficient to confirm an annual incidental mortality and serious injury rate of zero in these fisheries (see Table 1) (Barlow 1989, Babcock et al. 2003). An estimated 37 vessels participate in the CA halibut/white seabass and other species set gillnet (>3.5" mesh) fishery (86 FR 3028; January 14, 2021).

Three southern sea otter interactions with the California purse seine fishery for northern anchovy and Pacific sardine have been documented. In 2005, a contract observer in the NOAA Fisheries California Coastal Pelagic Species observer program documented the incidental, non-lethal capture of two sea otters that were temporarily encircled in a purse seine net targeting northern anchovy but escaped unharmed by jumping over the corkline. In 2006, a contract observer in the same program documented the incidental, non-lethal capture of a sea otter in a purse seine net targeting Pacific sardine. Again, the sea otter escaped the net at end of the haul without assistance.[3] There are no data available to assess whether sea otter interactions with purse-seine gear are currently resulting in mortality or serious injury. An estimated 65 vessels participate in the CA anchovy, mackerel, and sardine purse seine fishery (86 FR 3028; January 14, 2021). An estimated 80 vessels participate in the CA squid purse seine fishery (86 FR 3028; January 14, 2021).

The potential exists for sea otters to drown in traps set for crabs, lobsters, and finfish, but this source of mortality is likely under-reported due to the difficulty of identifying drowning as a cause of death in sea otters, and only limited documentation of mortalities is available. Hatfield and Estes (2000) summarize records of 18 sea otter mortalities in trap gear, 14 of which occurred in Alaska. With the exception of one sea otter, which was found in a crab trap, all of the reported Alaska mortalities involved Pacific cod traps and were either recorded by NMFS observers or reported to NMFS observers by fishers. As of 2000, four sea otters were known to have died in trap gear in California: one in a lobster trap near Santa Cruz Island in 1987; a mother and pup in a trap with a 10-inch diameter opening (presumed to be an experimental trap) in Monterey Bay in 1987; and one in a rock crab trap 0.5 miles off Pt. Santa Cruz, California (Hatfield and Estes 2000). In 1995, the U.S. Geological Survey began opportunistic efforts to observe the finfish trap fishery in California. These efforts were supplemented with observations by the California Department of Fish and Game (CDFG) in 1997 and two hired observers in 1999. No sea otters were found in the 1,624 traps observed (Hatfield and Estes 2000). However, a very high level of observer coverage would be required to see any indication of trap mortality, even if mortality levels were high enough to substantially reduce the rate of population growth (Hatfield et al.

[3] Personal communication, Lyle Enriquez, 2006. Southwest Regional Office, NOAA, U.S. National Marine Fisheries Service, 501 West Ocean Boulevard, Long Beach, CA 90802.

2011). In 2016, a dead sea otter was found in a lobster trap pulled by California Department of Fish and Wildlife wardens in the Port San Luis Area near Avila Beach. The discovery occurred on April 8, several weeks after commercial lobster season had closed (March 16) and traps should have been removed from the water.[4]

Controlled experiments conducted by the U.S. Geological Survey and the Monterey Bay Aquarium demonstrated that sea otters would enter a baited commercial finfish trap with inner trap funnel openings of 5.5 inches in diameter (Hatfield and Estes 2000). Hatfield et al. (2011) confirmed that some sea otters exposed to finfish, lobster, and mock Dungeness crab traps in a captive setting would succeed in entering them. Based on experiments with carcasses and live sea otters, they concluded that finfish traps with 5-inch-diameter circular openings would largely exclude diving sea otters; that circular openings of 5.5 to 6 inches in diameter and rectangular openings 4 inches high (typical of Dungeness crab pots) would allow the passage of sea otters up to about 2 years of age; and that the larger fyke openings of spiny lobster pots and finfish traps with openings larger than 5 inches would admit larger sea otters. Reducing the fyke-opening height of Dungeness crab traps by one inch (to 3 inches) would exclude nearly all diving sea otters while not significantly affecting the number or size of harvested crabs (Hatfield et al. 2011). Since January 2002, CDFG has required 5-inch sea-otter-exclusion rings to be placed in live-fish traps used along the central coast from Pt. Montara in San Mateo County to Pt. Arguello in Santa Barbara County. No rings are required for live-fish traps used in the waters south of Point Arguello, and no rings are currently required for lobster or crab traps regardless of their location in California waters. Estimates of the number of vessels participating in pot and trap fisheries off California are given in parentheses: CA Dungeness crab pot (501); CA rock crab pot (124); CA spiny lobster (186); and CA nearshore finfish live trap/hook-and-line (93) (86 FR 3028; January 14, 2021).

Available information on incidental mortality and serious injury of southern sea otters in commercial fisheries is very limited. Due to the lack of observer coverage, a reliable, science-based estimate of the annual rate of mortality and serious injury cannot be determined. Commercial fisheries believed to have the potential to kill or injure southern sea otters are listed in Table 1. Due to the nature of potential interactions (entrapment or entanglement followed by drowning), serious injury is unlikely to be detected prior to the death of the animal.

[4] Personal communication, Todd Tognazzini, 2016. Patrol Lieutenant, San Luis Obispo/Southern Monterey Counties, California Department of Fish and Wildlife, 3196 South Higuera, Suite A, San Luis Obispo, CA 93401.

Table 1. Summary of available information on incidental mortality and serious injury of southern sea otters in commercial fisheries that have the potential to interact with southern sea otters.

Fishery Name	Category	Year(s)	Number of Vessels[1]	Data Type	Percent Observer Coverage[2]	Observed Mortality/ Serious Injury	Estimated Mortality/ Serious Injury	Mean Annual Mortality/ Serious Injury
CA halibut/white seabass and other species set gillnet (>3.5")	2	2015 2016 2017 2018 2019	37	n/a n/a observer observer observer	n/a n/a ≈10% n/a n/a	n/a n/a 0 0 0	n/a	n/a
CA anchovy, mackerel, and sardine purse seine[3]	3	2015–2019	65	n/a	not observed	n/a	n/a	n/a
CA squid purse seine	3	2015–2019	80	n/a	not observed	n/a	n/a	n/a
CA Dungeness crab pot	2	2015–2019	501	n/a	not observed	n/a	n/a	n/a
CA rock crab pot[3]	3	2015–2019	124	n/a	not observed	n/a	n/a	n/a
CA spiny lobster	2	2015 2016 2017 2018 2019	186	n/a incidental n/a n/a n/a	not observed not observed not observed not observed not observed	n/a 1[4] n/a n/a n/a	≥1	≥0.2
CA nearshore finfish live trap/hook and line[3]	3	2015–2019	93	n/a	not observed	n/a	n/a	n/a
Unidentified hook/line/net[7]	n/a	2015 2016 2017 2018 2019	n/a	stranding	n/a	1[5] 2[6] 0 0 1	≥4	≥0.8
TOTAL							≥5	≥1.0

Note: n/a indicates that data are not available or are insufficient to estimate mortality/serious injury.

[1] Vessel numbers are from the final List of Fisheries for 2021 (86 FR 3028; January 14, 2021).

[2] Personal communication, Jim Carretta, 2013, 2016, 2017, 2019, 2021. Southwest Fisheries Science Center, NOAA, U.S. National Marine Fisheries Service, 8604 La Jolla Shores Drive, La Jolla, CA 92037.

[3] Category III fisheries are not required to accommodate observers aboard vessels due to the remote likelihood of mortality and serious injury of marine mammals.

[4] This sea otter mortality was incidentally discovered by CDFW wardens in 2016 while retrieving an illegally set lobster trap.

[5] This sea otter was seriously injured, rehabilitated, and released.

[6] One sea otter died; the other was seriously injured, rehabilitated, and released in 2016 but then shot in 2017.

[7] Because it is often not possible to make a definitive determination whether entanglements are due to commercial or recreational gear, we have included here all known strandings caused by entanglement in unidentified gear. As a result, mortality in commercial fishing gear may be overestimated for this category.

Other Mortality

An effort to document all southern sea otter strandings (live and dead sea otters that wash ashore) has been underway since 1968. Relative mortality is calculated by dividing the number of carcasses retrieved in a given year along the mainland coastline by the number of sea otters recorded in the annual census along the mainland for that same year. Strandings at San Nicolas Island are rarely recovered, so this area is not represented in the totals. Values for relative mortality represent an index rather than a true per capita mortality rate because fewer than half of the animals that die in the wild are recovered (Gerber et al. 2004) and because the spring count itself is an index rather than a true estimate of abundance. Relative mortality values can also be influenced by other factors, such as changes in the proportion of sea otters occupying areas that are amenable to carcass deposition and detection (sandy embayments vs. steep rocky coastline) or changes in the intensity of beach monitoring. Nevertheless, it remains the best available index for tracking mortality rates over time.

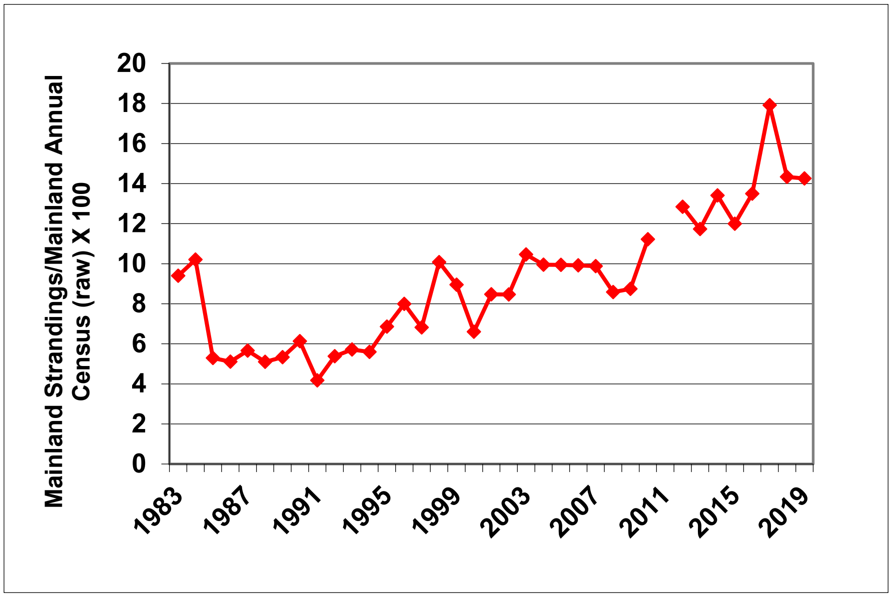

Figure 2. Strandings of southern sea otters relative to the annual census (mainland only) 1983–2019. The annual census was not completed in 2011 (due to weather) or 2020 (due to COVID-19 restrictions).

Relative mortality was roughly constant at about 5 percent during the period of population growth from 1985–1995 but somewhat higher during periods of apparent population decline (the early 1980s and 1996–1999) (Figure 3). Whereas the population decline during the early 1980s has been attributed to gill net mortality (Estes 1990), the cause of the decline during the late 1990s has not been determined (Estes et al. 2003). Unusually high numbers of stranded southern sea otters were recovered in 2003, resulting in a relative mortality rate of 10.5 percent and prompting declaration of an Unusual Mortality Event. Intoxication by domoic acid produced by blooms of the alga *Pseudonitzschia australis* is believed to have been an important contributor (Jessup et al. 2004), but no one cause has been identified as being responsible.

In recent years, relative mortality has exceeded 2003 levels, averaging 14.4 percent from 2015–2019 (Figure 3). Relative mortality spiked in 2017, but this apparent increase may be an artifact of the unusually low raw census count that year, which is believed to have been influenced by sparse surface kelp canopy, which resulted in poor counting conditions. The absolute number of sea otters that stranded in 2017 (467) is similar to the number that stranded along the mainland in 2016 (474), 2018 (428), and 2019 (427).

The increases in relative mortality that have occurred during the past two decades appear to be due largely to an accelerating increase in shark bite mortality, particularly in the northern and southern portions of the mainland range (north of Seaside and, most markedly, from Estero Bay to Point Conception) (Tinker et al. 2016, Hatfield et al. 2019). The stranding rate of shark-bitten sea otters has increased dramatically relative to the population index, much more than the stranding rates of sea otters due to all other causes combined. This fact suggests that alternate explanations for the increase in the relative frequency of shark-bitten carcasses, such as increased monitoring and carcass recovery efforts or decreased per capita mortality due to other factors, are unlikely (Tinker et al. 2016). Rangewide, the estimated probability that a stranded sea otter will be shark-bitten has increased threefold, from 19 percent in 1990 to 61 percent in 2013; in the southern portion of the range this probability has increased eightfold, from 8 percent in 1990 to 68 percent in 2013 (see Tinker et al. 2016 for associated 95-percent confidence bounds). These shark bites are non-consumptive and probably investigatory. Contributing factors include possible increases in white shark (*Carcharodon carcharias*) numbers; changes in white shark behavior and distribution due to increasing populations of northern elephant seals (*Mirounga angustirostris*) and California sea lions (*Zalophus californianus*) along the California coastline and warm water intrusions that have allowed juvenile white sharks to use more northerly habitat (Tinker et al. 2016, Moxley et al. 2019); and loss of kelp cover, which is thought to provide protection from shark attacks (Nicholson et al. 2018). High rates of shark bite mortality appear to be responsible for the lack of population growth at the range peripheries, which in turn likely explains the lack of range expansion at both the north and south ends of the mainland range (Hatfield et al. 2019).

Population dynamics in the central portion of the mainland range (Seaside to Cayucos) appear to be influenced primarily by density-dependent resource limitation (Tinker et al. 2019*b*). Physiological condition and nutritional status in turn influence the susceptibility of sea otters to environmental stressors (including pathogens, pollutants, and intoxicants produced during harmful algal blooms), which may result in death by a variety of proximate causes, including infectious disease, intra-specific aggression, intoxication, and other pathological conditions (Tinker et al. 2019*b*, Miller et al. 2020). Lower per-capita food availability also leads to greater reliance on sub-optimal prey, which increases exposure and susceptibility to novel disease-causing pathogens (Johnson et al. 2009, Tinker et al. 2019*b*).

Non-fishery-related anthropogenic mortality of sea otters is a result of indirect and direct causes. Boat strikes typically cause several deaths each year. Shootings are a relatively low but persistent source of anthropogenic mortality and in some cases appear to be related to fishery interactions. This cause of death is likely under-reported due to the lack of systematic radiographs of all carcasses. Other rare sources of anthropogenic mortality include debris entanglement, non-boat vehicle strikes, and complications associated with research activities. Stranding data indicate that from 2015–2019, 12 sea otters were struck by boats, 3 were shot[5], 1 was struck by a car while attempting to cross a roadway, 1 was struck by a train while attempting to cross railroad tracks, and 2 died of complications associated with research (U.S. Geological Survey and CDFW unpublished data). Total observed anthropogenic mortality for 2015–2019, excluding any fisheries-related mortality, is 19, yielding an estimated mortality of ≥19 and a mean annual mortality of ≥3.8. Disease (including biotoxin intoxication) is an important proximate cause of death in sea otters and has indirect links to human behavior. However, due to the complexity of the pathways by which sea otters are being affected by land-borne pathogens and pollution and the synergistic relationship between sea otter susceptibility to disease and

[5] An additional animal, not included in this total to avoid double-counting, was shot and killed in 2017 after having sustained serious injuries from entanglement in fishing gear in 2016 and being successfully rehabilitated and released (see Table 1).

density-dependent resource limitation, the anthropogenic contribution to disease-related mortality in sea otters is difficult to quantify. Therefore, animals that died of disease are not included in the anthropogenic mortalities reported here.

The mean annual mortality/serious injury reported here and in Table 1 are minimum estimates.[6] Documentation of these sources of mortality comes primarily from necropsies of beach-cast carcasses, which constitute a subset (roughly half) of all dead southern sea otters and likely do not represent an unbiased sample with respect to cause of death because carcass deposition and retrieval are dependent on carcass size, location, wind, currents and other factors, including the cause of death itself (Estes et al. 2003, Gerber et al. 2004, Tinker et al. 2006a). Within this subset, the cause of death of many recovered carcasses is unknown, either because the carcass is too decomposed for examination or because cause of death cannot be determined (Gerber et al. 2004).[7] The "relative mortality" rate is therefore an underestimate of the true mortality rate. Because it is unknown to what extent the levels of human-caused mortality documented in beach-cast carcasses are representative of the relative contributions of known causes or of human-caused mortality as a whole, we are unable to give upper bounds for these estimates.

STATUS OF STOCK

The southern sea otter is designated a fully protected mammal under California State law (California Fish and Game Code §4700) and was listed as a threatened species in 1977 (42 FR 2965) pursuant to the federal Endangered Species Act, as amended (16 U.S.C. 1531 et seq.) (ESA). As a consequence of its threatened status, the southern sea otter is considered to be a "strategic stock" and "depleted" under the MMPA.

According to the Southern Sea Otter Recovery Plan (U.S. Fish and Wildlife Service 2003), the 3-year average count (range-wide population index) would have to exceed 3,090 for three consecutive years for southern sea otters to be considered for delisting under the Endangered Species Act. A 5-year review, which analyzed the status of the southern sea otter in relation to the recovery criterion and the five statutory delisting criteria, concluded in 2015 that it still meets the definition of threatened and recommended no change in listing status (USFWS 2015). However, the range-wide population index reached the three-year threshold in 2018. 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 at 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. The U.S. Fish and Wildlife Service initiated a review of the status of the southern sea otter in 2019 (84 FR 36116; July 26, 2019).

Under Public Law 99-625, the San Nicolas Island colony was formerly considered to be an experimental population (52 FR 29754; August 11, 1987), but the experimental population designation was removed upon termination of the translocation program and its respective

[6] This statement applies to all causes of death mentioned here except research-related mortalities. Research-related mortalities are unlikely to be undetected because of the intensive monitoring that tagged sea otters receive.
[7] In 2012, for example, the cause of death of approximately 35 percent of recovered carcasses was unknown. Personal communication, Brian Hatfield, 2013. Wildlife Biologist, USGS-Western Ecological Research Center, Hwy. 1, P.O. Box 70, San Simeon, CA 93452.

translocation and management zones (77 FR 75266; December 19, 2012). With the termination of the translocation program, the special status afforded to southern sea otters within the management and translocation zones pursuant to Public Law 99-625 also ended. However, the National Defense Authorization Act for Fiscal Year 2016 included provisions directing the Secretary of the Navy to establish Southern Sea Otter Military Readiness Areas (Areas) at San Nicolas Island and San Clemente Island (where sea otters do not currently occur). Military readiness activities[8] conducted within these Areas are subject to certain exemptions under the ESA and MMPA.[9]

The status of the southern sea otter in relation to its OSP level has not been formally determined, but population counts are well below the candidate value proposed by Tinker et al. (2021) for California: 10,236. This number represents 59.4 percent of the projected carrying capacity estimate of 17,226 sea otters (Tinker et al. 2021). This candidate value is for California only, as it does not account for habitat outside California but within the historical range of the subspecies.

Based on the currently available data, the minimum level of human caused mortality and serious injury is ≥ 4.8 sea otters per year (≥ 1.0 from fishery sources in Table 1 + ≥ 3.8 from other human caused serious injury and mortality). The known mortality is thus less than PBR. However, due to the lack of observer data for several commercial fisheries that may interact with sea otters and biases in the stranding data, it is not possible to make a science-based estimate of the annual mortality and serious injury associated with fisheries and other sources of human-caused mortality and serious injury. Consequently, it is not possible to make a science-based determination of whether the total mortality and serious injury of sea otters due to human-caused mortalities and serious injuries is insignificant and approaching a zero mortality and serious injury rate.

Habitat Issues

Sea otters are particularly vulnerable to oil contamination (Kooyman and Costa 1979; Siniff et al. 1982), and oil spill risk from large vessels that transit the California coast remains a primary threat to the southern sea otter (USFWS 2015). The stock's vulnerability to oil spills has been exacerbated by the historically slow pace of natural range expansion (resulting from the spatial configuration of available habitat along the mainland California coast and the limited mobility of reproductive females) and by the curtailment of range expansion caused by high levels of shark-bite mortality at the range ends (Tinker et al. 2016, Hatfield et al. 2019).

Food limitation and nutritional deficiencies in densely populated areas of the range appear to be primary drivers of sea otter mortality, either directly or as a consequence of dietary specialization (Bentall 2005, Tinker et al. 2006b, Tinker et al. 2008a, Johnson et al. 2009, Tinker et al. 2019*b*). Poor body condition increases susceptibility to environmental stressors, such as

[8] According to the NDAA, "The term 'military readiness activity' has the meaning given that term in section 315(f) of the Bob Stump National Defense Authorization Act for Fiscal Year 2003 (16 U.S.C. 703 note) and includes all training and operations of the armed forces that relate to combat and the adequate and realistic testing of military equipment, vehicles, weapons, and sensors for proper operation and suitability for combat use."

[9] With respect to the ESA, Sections 4 and 9 do not apply to the incidental taking of any southern sea otter in the Areas in the course of conducting a military readiness activity, and any sea otter within the Areas is to be treated for the purposes of section 7 as a member of a species that is proposed to be listed as endangered or threatened under the ESA. With respect to the MMPA, Sections 101 and 102 do not apply with respect to the incidental taking of any sea otter in the Areas in the course of conducting a military readiness activity.

pathogens, pollutants, and intoxicants produced during harmful algal blooms (Tinker et al. 2019*b*). Although shark bites are the most common primary cause of death in California, infectious disease (especially acanthocephalan peritonitis and protozoal encephalitis) is the most prevalent cause of death when primary and contributing cause of death are combined (Miller et al. 2020). Other common non-traumatic causes of death in California include harmful algal and cyanobacterial blooms and bacterial infections (Miller et al. 2020).

Acanthocephalans are thorny-headed worms that infect the intestinal tract. 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 (Miller et al. 2020).

Protozoal parasites can cause severe encephalitis in sea otters. *Toxoplasma gondii* is shed in the feces of both wild and domestic cats (Dubey et al. 1970; Miller et al. 2002, 2004, 2008, 2020). *Sarcocystis neurona* is shed in the feces of opossums (*Didelphis virginiana* and *D. albiventris*) (Kreuder et al. 2003, Miller et al. 2010, 2020).

Harmful algal or cyanobacterial blooms, which are exacerbated in some cases by anthropogenic inputs of nitrogen or phosphorus into the nearshore marine environment and by ocean warming (Mos 2001, Kudela et al. 2008, Vezie et al. 2002, Gobler et al. 2017), can cause acute, subacute, or chronic effects in exposed sea otters (Kreuder et al. 2003, Miller et al. 2010). Biotoxins released during harmful blooms include domoic acid, which is produced by marine diatoms of the genus *Pseudonitzschia*, and microcystin, which is produced by freshwater cyanobacteria of the genus *Microcystis*. Domoic acid intoxication of sea otters was first reported in 2003 (Kreuder et al. 2003) and has subsequently been associated with increased risk of cardiac disease (Kreuder et al. 2005, Moriarty et al. 2021).

Studies of contaminants have documented accumulations of dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyl-dichloroethylene (DDE) (Bacon 1994; Bacon et al. 1999), and polychlorinated biphenyls (PCBs) in stranded sea otters (Nakata et al. 1998), as well as the presence of butyltin residues, which are known to be immunosuppressant (Kannan et al. 1998). Kannan et al. (2006, 2007) found a significant association between infectious diseases and elevated concentrations of perfluorinated contaminants and polychlorinated biphenyls (PCBs) in the livers of sea otters, suggesting that chemical contaminants may influence patterns of sea otter mortality.

The effects of climate change may affect southern sea otters by modifying hydrological processes that influence the transport of pathogens and contaminants from land to the nearshore marine environment (Walther et al. 2002). It also has the potential to alter the frequency of algal blooms in both freshwater and the marine environment (Gobler et al. 2017). Increasing ocean temperatures may increase the incidence and spread of disease among marine organisms (Burge et al. 2014, Harvell et al. 2019), with potentially negative or positive effects on sea otters depending on the particular ecological relationships affected. Warming events are expected to lead to increased presence of juvenile white sharks in areas that overlap with sea otters and a decrease in kelp canopy cover and the protections it affords from shark attacks (Nicholson et al. 2018, Moxley et al. 2019). In addition to increasing ocean temperatures, changes in the carbonate chemistry of the oceans due to increasing atmospheric CO_2 levels (ocean acidification) may pose a serious threat to marine organisms, particularly calcifying organisms (Kroeker et al. 2010, Kurihara and Shirayama 2004, Kurihara et al. 2008, Stumpp et al. 2011, Gazeau et al. 2013, Marshall et al. 2017), many of which are important prey for sea otters. Because of the apparent synergistic relationship between food limitation and disease, potential climate-driven declines in food availability may in turn result in increased susceptibility to disease.

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