

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.

Historically, southern sea otters ranged from present-day Punta Abreojos, Baja California, Mexico, to at least as far north as Newport, Oregon (Valentine et al. 2008). 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 historic range (U.S. Fish and Wildlife Service 2015). Range expansion is of primary importance for recovery of the subspecies and restoration of the nearshore marine ecosystems of which southern sea otters were once a part (USFWS 2003, USFWS 2015). The estimated carrying capacity of California is approximately 16,000 animals (Laidre et al. 2001). The carrying capacity of the remainder of the southern sea otter’s historic range has not been determined.

Sea otter abundance varies considerably across the range, with the highest densities occurring in the central portion (Seaside to Cayucos), where sea otters have been present the longest. Sea otter densities tend to be most stable from year to year in rocky, kelp-dominated areas, which are primarily occupied by females, dependent pups, and territorial males. In contrast, sandy and soft-bottom habitats (in particular those in Monterey Bay, Estero Bay, and Pismo Beach to Pt. Sal) tend to be occupied by non-territorial males and sub-adult animals of both sexes (and only rarely by adult females and pups) and are more variable in abundance from

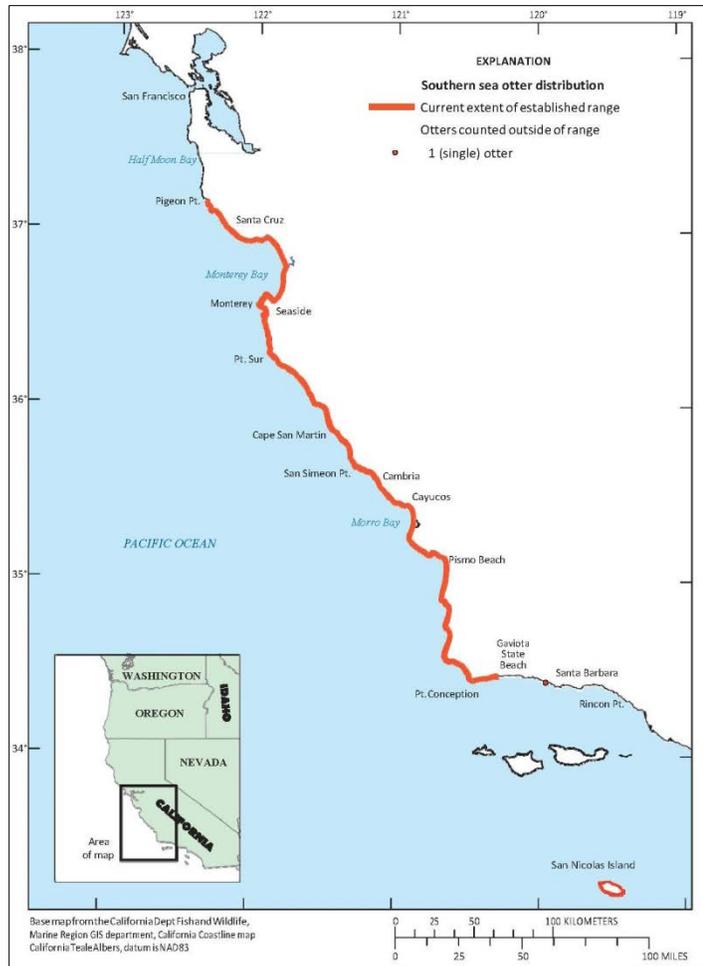


Figure 1. Current range of the southern sea otter (2016 census). Source: Tinker and Hatfield 2016.

year to year. This variation is driven in part by the long-distance movements and seasonal redistribution of males (Tinker et al. 2008b). Many males migrate to the range peripheries during winter and early spring, apparently to take advantage of more abundant prey resources, but then return to the range center in search of estrous females (Jameson 1989, Ralls et al. 1996, Tinker et al. 2008b). Mating and pupping occur year round, with a birth peak from October to January and a secondary peak in March and April (Chinn et al. 2016).

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 studies of cranial morphology (Wilson et al. 1991) and variation at the molecular level (Sanchez 1992; Cronin et al. 1996; Larson et al. 2002).

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 2016, the range-wide index of abundance was 3,272 (Tinker and Hatfield 2016).

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 2016, the combined raw count was 3,615, but the combined 3-year running average was only 3,272. Therefore, the minimum population estimate is 3,272 animals (3,194 along the mainland and 78 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 past 5 years is 3.2 percent per year (Tinker and Hatfield 2016; Figure 2).

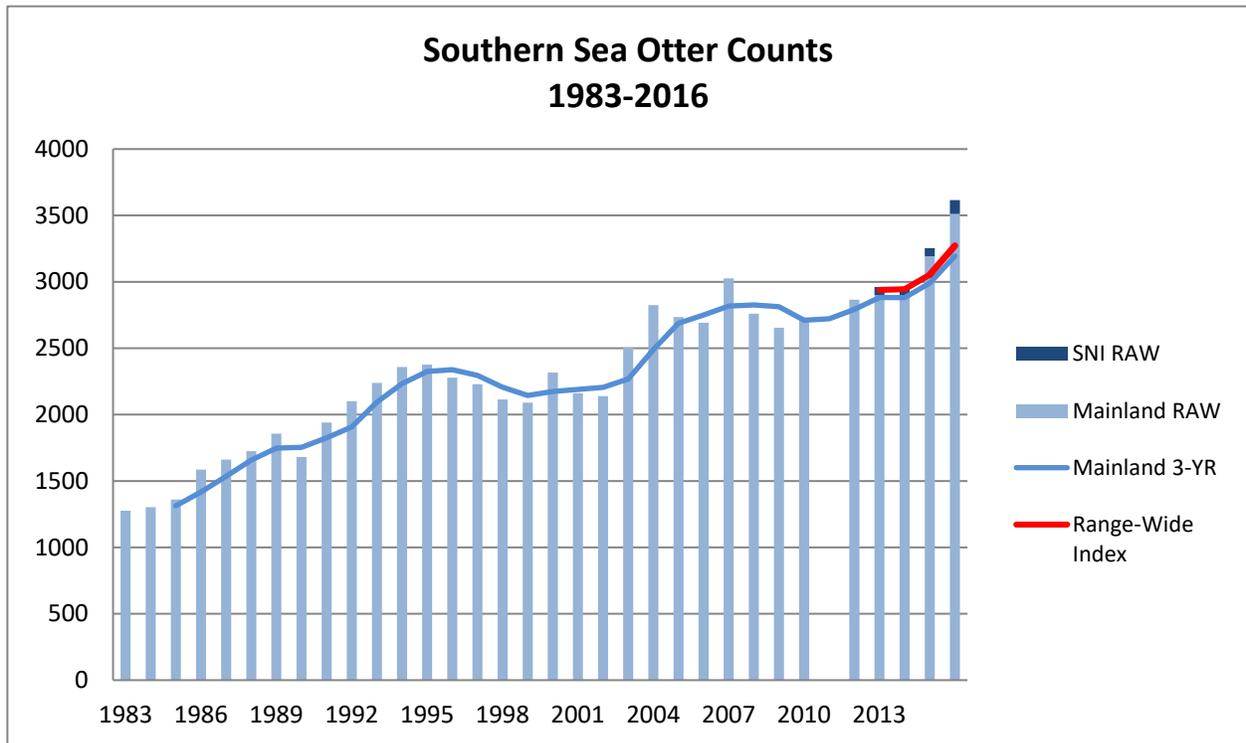


Figure 2. Southern sea otter counts 1983-2016. Bars show raw counts for each year, whereas lines represent 3-year running averages. The mainland census was not completed in 2011. Data source: Tinker and Hatfield 2016.

The rangewide trend is strongly influenced by high counts in 2015 and 2016 in the center portion of the mainland range (Seaside to Cayucos), an area that was formerly considered to be at or near carrying capacity. In the central region, population growth over the past 5 years has averaged 6 percent per year. The increase in sea otter numbers in this area may be due to increased juvenile survival in response to an unusually high abundance of sea urchins since 2013 (Tinker and Hatfield 2016).

In the portions of the mainland range to the north and south of the central region, trends are negative, with declines over the past 5 years averaging 2.5 percent per year in the northern region (Pigeon Point to Seaside) and 0.6 percent per year in the southern region (Cayucos to Gaviota) (Tinker and Hatfield 2016). These regional declining trends coincide with areas of increased shark-bite mortality (Tinker et al. 2016) and are the probable reason for the lack of range expansion over the past several years (Tinker and Hatfield 2016).

The small subpopulation at San Nicolas Island continues the strong growth trend it has exhibited since approximately 2010. Annual growth has averaged 13 percent over the past 5 years.

CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

We use the 5-year population trend to characterize current net productivity rates. During the past 5 years, growth of the mainland population averaged 3 percent per year, whereas growth of the San Nicolas Island population averaged 13 percent per year (Tinker and Hatfield 2016). Because most of the population occurs along the mainland coastline, the rangewide population growth trend of 3.2 percent per year is heavily influenced by the mainland population trend.

The maximum growth rate (R_{\max}) for southern sea otters along the mainland coastline since the early 1980s (when reliable trend data first become available) appears to be 6 percent per year, although localized sub-populations have been observed to grow at much higher rates immediately after re-colonization (Lafferty and Tinker 2014). In contrast, 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).

Although there has been speculation that the slower rate of population growth observed for the southern sea otter reflects a fundamental difference in survival or reproduction relative to northern sea otter populations, recent data and analyses indicate that the emphasis on differential rates of population growth may be misplaced. First, a variety of evidence supports the conclusion that sea otters throughout much of central California have been at or near the carrying capacity of the local environment for several years. The population's status with respect to carrying capacity explains the lack of growth in these areas prior to 2015 and 2016 (*i.e.*, further growth was limited by available food resources) (Tinker et al. 2006b, Tinker et al. 2008a). Second, radio-tagging studies report age- and sex-specific rates of survival and reproduction that are comparable for southern sea otters and northern sea otters when status with respect to carrying capacity is accounted for (Monson et al. 2000, Tinker et al. 2006b). Finally, recent modeling analyses indicate that the spatial configuration of available habitat (the long narrow strip of coastal shelf characteristic of the mainland California sea otter range versus the bays, islands, and complex matrices of inland channels characteristic of the habitat in British Columbia and Alaska), combined with the high degree of spatial structure in sea otter populations (due to the limited mobility of reproductive females), result in greatly different expected population growth rates over the long term and may account in large part for the differences in trends between the southern sea otter and northern sea otter populations (Tinker 2014).

From the early 1900s to the mid-1970s, the southern sea otter population is thought to have increased at about 5 percent annually (Estes 1990), although consistent surveys and trend data from early years are lacking. From 1983 to 1995, annual growth averaged about 6 percent. The population declined during the late 1990s, resumed growth in the early 2000s, and ceased growth again beginning in 2008. Growth resumed in 2010, resulting in an average increase of 3 percent annually over the past 5 years. Growth rates at San Nicolas Island averaged approximately 9 percent annually from the early 1990s to the mid-2000s and approximately 13 percent over the past 5 years.

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: $PBR = (N_{\min})(\frac{1}{2} \text{ of } R_{\max})(F_r)$.

For the southern sea otter stock, N_{\min} is 3,272 (3,194 along the mainland and 78 at San Nicolas Island). Because the maximum population growth rate appears to be tightly constrained by habitat configuration, we use an R_{\max} of 6 percent for the mainland portion of the population and an R_{\max} of 13 percent for the island portion of the population. We use a recovery factor of 0.1 for the southern sea otter stock because, although the population appears to be stable or growing, 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 10.09 $[(3,194 \times 0.5 \times 0.06 \times 0.1) + (78 \times 0.5 \times 0.13 \times 0.1)]$, which when rounded down to the nearest whole animal is 10. It is 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 50 vessels participate in the CA halibut/white seabass and other

species set gillnet (>3.5" mesh) fishery (81 FR 20550, April 8, 2016). Approximately 30 vessels participate in the CA yellowtail, barracuda, and white seabass drift gillnet fishery (mesh size ≥ 3.5 " and <14") (81 FR 20550, April 8, 2016). Approximately 18 vessels participate in the CA thresher shark/swordfish drift gillnet fishery (≥ 14 " mesh) (81 FR 20550, April 8, 2016).

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.¹ 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 (81 FR 20550, April 8, 2016).

The potential exists for sea otters to drown in traps set for crabs, lobsters, and finfish, but 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. 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.²

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

¹ Personal communication, Lyle Enriquez, 2006. Southwest Regional Office, NOAA, U.S. National Marine Fisheries Service, 501 West Ocean Boulevard, Long Beach, CA 90802.

² 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.

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 Conception, 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 (570); CA rock crab pot (124); CA spiny lobster (194); and CA nearshore finfish live trap/hook-and-line (93) (81 FR 20550, April 8, 2016).

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.

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 | 2011 2012 2013 2014 2015 | 50 | observer observer observer n/a n/a | 8.3% 14.6% 11.3% not observed not observed | 0 0 0 n/a n/a | n/a | n/a |
| CA yellowtail, barracuda, and white seabass drift gillnet (≥3.5" and <14") | 2 | 2011 2012 2013 2014 2015 | 30 | observer observer n/a n/a n/a | 3.3% 0.7% not observed not observed not observed | n/a 0 0 n/a n/a | n/a | n/a |
| CA thresher shark/swordfish drift gillnet fishery (≥14") | 1 | 2011 2012 2013 2014 2015 | 18 | observer | 19.5% 18.6% 37.2% 23.7% 20.0% | 0 0 0 0 0 | 0 | 0 |
| CA anchovy, mackerel, and sardine purse seine | 3 | 2011-2015 | 65 | n/a | not observed | n/a | n/a | n/a |
| CA Dungeness crab pot | 2 | 2011-2015 | 570 | n/a | not observed | n/a | n/a | n/a |
| CA rock crab pot ³ | 3 | 2011-2015 | 124 | n/a | not observed | n/a | n/a | n/a |
| CA spiny lobster ^{3,5} | 3 | 2011-2015 | 194 | n/a | not observed | n/a | n/a | n/a |
| CA nearshore finfish live trap/hook and line ³ | 3 | 2011-2015 | 93 | n/a | not observed | n/a | n/a | n/a |
| Unknown hook and line | n/a | 2011 2012 2013 2014 2015 | n/a | stranding data | — | 0 0 1 0 1 ⁶ | ≥1 | ≥0.2 |
| Unknown net | n/a | 2011 2012 2013 2014 2015 | n/a | stranding data | — | 1 ⁴ 0 0 0 0 | ≥1 | ≥0.2 |
| TOTAL | | | | | | | n/a | n/a |

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 2016 (81 FR 20550, April 8, 2016).

² Personal communication, Jim Carretta, 2010, 2011, 2013, 2016, 2017. Southwest Fisheries Science Center, NOAA, U.S. National Marine Fisheries Service, 8604 La Jolla Shores Drive, La Jolla, CA 92037.

³ This fishery is classified as a Category III fishery (81 FR 20550, April 8, 2016). 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 was also shot, presumably after becoming entangled in the net.

⁵ Observer coverage data are currently available only through 2015. Therefore, this table does not include the sea otter mortality reported by CDFW wardens in 2016 and described in the text. When additional observer coverage data become available, the table will be updated to include this mortality and other available information through 2016.

⁶ This sea otter was seriously injured, rehabilitated, and released.

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 coastline by the number of sea otters recorded in the spring count for that same year. This value represents an index rather than a true per capita mortality rate because less than 50

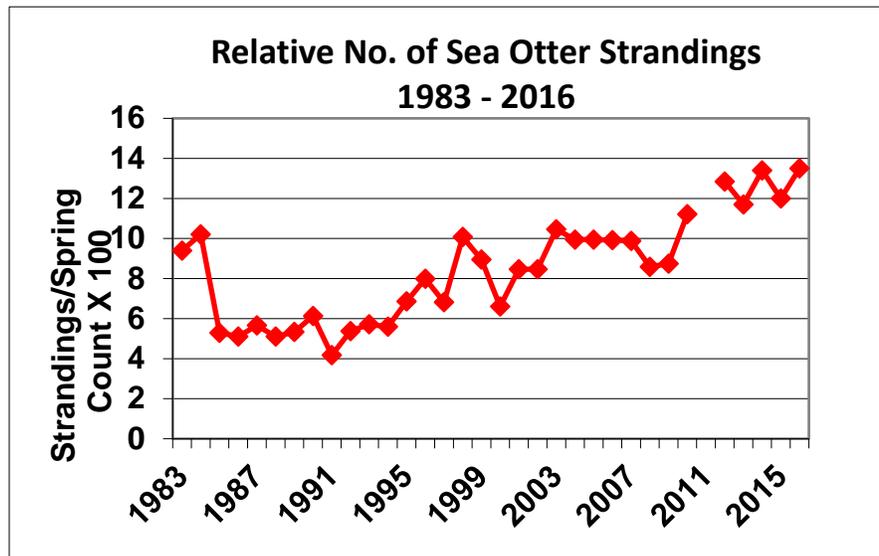


Figure 3. Strandings of southern sea otters relative to the spring count, 1983-2016. The entry for 2011 is missing because the spring survey was not completed that year. Source: U.S. Geological Survey unpublished data.

percent 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.

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, 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 even 2003 levels, averaging 12.7 percent from 2011-2016 (Figure 3). These increases in relative mortality 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). 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. The reasons for the increase in shark bites in areas of the sea otter range not previously subject to high rates of shark-related mortality are not well understood, but they may reflect growing white shark (*Carcharodon carcharias*) numbers or changes in white shark behavior and distribution associated with increasing populations of northern elephant seals (*Mirounga angustirostris*) and California sea lions (*Zalophus californianus*) along the California coastline (Tinker et al. 2016). 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 (Tinker and Hatfield 2016).

Variation in reproductive success and survival rates of sea otters in the central portion of the mainland range (Seaside to Cayucos) appears to be influenced primarily by density-dependent resource limitation (Tinker et al. 2013). 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. 2013). 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. 2013).

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. Other rare sources of anthropogenic mortality include debris entanglement and complications associated with research activities. Stranding data indicate that from 2011-2015, 4 were shot³ and 20 were struck by boats (U.S. Geological Survey and CDFW unpublished data). Total observed anthropogenic mortality for 2011-2015, excluding any fisheries-related mortality, is 24, yielding an estimated mortality of ≥ 24 and a mean annual mortality of ≥ 4.8 . Disease is an important proximate cause of death in sea otters, but due to several complicating factors (including the complexity of the pathways by which sea otters are being exposed to land-borne pathogens, the synergistic relationship between sea otter susceptibility to disease and density-dependent resource limitation, and other factors), the anthropogenic contribution to disease-related mortality in sea otters is not well understood. 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.⁴ 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,

³ An additional animal, not included in this total, was also shot, apparently after becoming entangled in a net in 2011 (see Table 1). Three sea otters died of gunshot wounds in 2016. These mortalities are not included in the current calculation of mean annual mortality because they occurred outside the 5-year analysis window (2011-2015).

⁴ 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.

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).⁵ 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. 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 that it still meets the definition of threatened and recommended no change in listing status (USFWS 2015).

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 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 activities⁶ conducted within these Areas are subject to certain exemptions under the ESA and MMPA.⁷

The status of the southern sea otter in relation to its optimum sustainable population (OSP) level has not been formally determined, but population counts are well below the estimated lower bound of the OSP level for southern sea otters in California, about 8,400 animals (U.S. Fish and Wildlife Service 2003), which is roughly 50 percent of the estimated carrying

⁵ 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.

⁶ 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.”

⁷ 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.

capacity of California (Laidre et al. 2001). This number does not account for habitat outside California but within the historic range of the subspecies. The formal determination of OSP will be developed with reference to the entire historic range of the subspecies.

Based on the currently available data, the minimum level of human caused mortality and serious injury is ≥ 5.2 sea otters per year (≥ 0.4 from fishery sources in Table 1 + ≥ 4.8 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 more recent curtailment of range expansion caused by high levels of shark-bite mortality at the range ends (Tinker et al. 2008b, Lafferty and Tinker 2014).

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. 2013). Poor body condition increases susceptibility to environmental stressors, such as pathogens, pollutants, and intoxicants produced during harmful algal blooms (Tinker et al. 2013). Important disease-causing pathogens include the protozoal parasite *Toxoplasma gondii*, which is shed in the feces of both wild and domestic cats (Dubey et al. 1970, Miller et al. 2002, Miller et al. 2004, Miller et al. 2008) and *Sarcocystis neurona*, which is shed in the feces of opossums (*Didelphis virginiana* and *D. albiventris*) (Kreuder et al. 2003, Miller et al. 2010). Both of these pathogens can cause severe encephalitis in sea otters. Protozoal encephalitis was identified as the primary cause of death in approximately 23 percent of beach-cast sea otter carcasses examined between 1998 and 2001 (Kreuder et al. 2003). Encephalitis caused by *T. gondii* in particular has been associated with shark attack and cardiac disease (Kreuder et al. 2003). A seroprevalence analysis updated through 2004 revealed that 52 percent of 305 freshly dead, beach-cast sea otters and 38 percent of 257 live sea otters sampled along the California coast were infected with *T. gondii* (Conrad et al. 2005). Infection by acanthocephalan parasites (*Proflicollis* spp.) was reported as the direct or indirect cause of mortality in 13 percent of 162 beach-cast carcasses sampled from 1997-2001 (Mayer et al. 2003) and the primary cause of death in 16.2 percent of 105 beach-cast carcasses sampled from 1998-2001 (Kreuder et al. 2003).

Harmful algal or cyanobacterial blooms, which are exacerbated in some cases by anthropogenic inputs of nitrogen or phosphorus into coastal watersheds and the nearshore marine environment (Mos 2001, Kudela et al. 2008, Vezie et al. 2002), 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 cardiac disease (Kreuder et al. 2005). Microcystin has been implicated as either a primary or contributing cause in the deaths of more than 40 sea otters through 2013 (with the earliest known case occurring in 1999 and the greatest number of cases occurring in 2007) (Miller et al. 2010, Tinker et al. 2013).

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 (in unknown ways) the frequency of algal blooms in both freshwater and the marine environment. Increasing ocean temperatures may increase the incidence and spread of disease among marine organisms (Burge et al. 2014), with potentially negative or positive effects on sea otters depending on the particular ecological relationships affected. In addition to increasing ocean temperatures, changes in the carbonate chemistry of the oceans due to increasing atmospheric CO₂ levels (ocean acidification) may pose a serious threat to marine organisms, particularly calcifying organisms (Kroeker et al. 2010, Kurihara et al. 2004, Kurihara et al. 2008, Stumpp et al. 2011, Gazeau et al. 2013), 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.

REFERENCES

- Babcock, E.A., E.K. Pikitch, and C.G. Hudon. 2003. How much observer coverage is enough to adequately estimate bycatch? Pew Institute for Ocean Science and Oceana, 36 pp.
- Bacon, C.E. 1994. An ecotoxicological comparison of organic contaminants in sea otters among populations in California and Alaska. M.S. thesis, University of California, Santa Cruz.
- Bacon, C.E., W.M. Jarman, J.A. Estes, M. Simon, and R.J. Norstrom. 1999. Comparison of organochlorine contaminants among sea otter (*Enhydra lutris*) populations in California and Alaska. *Environ. Toxicology and Chemistry* 18:452-458.
- Barlow, J. 1989. Estimating sample size required to monitor marine mammal mortality in California gillnet fisheries. Southwest Fisheries Science Center Administrative Report LJ-89-08, 8 pp.
- Bentall, G.B. 2005. Morphological and behavioral correlates of population status in the southern sea otter: a comparative study between central California and San Nicolas Island. Master's Thesis, University of California, Santa Cruz, CA, unpublished.
- Burge, C.A., C.M. Eakin, C.S. Friedman, B. Froelich, P.K. Hershberger, E.E. Hofmann, L.E. Petes, K.C. Prager, E. Weil, B.L. Willis, S.E. Ford, and C.D. Harvell. 2014. Climate change influences on marine infectious diseases: implications for management and society. *Annual Review of Marine Science* 6:249-277.
- Bryant, H.C. 1915. Sea otters near Point Sur. *California Department of Fish and Game Bulletin*. 1:134-135.
- Cameron, G.A. and K.A. Forney. 2000. Preliminary estimates of cetacean mortality in California/Oregon gillnet fisheries for 1999. Paper SC/S2/O24 presented to the International Whaling Commission, 2000 (unpublished), 12 pp. Available from NMFS, Southwest Fisheries Science Center, P.O. Box 271, La Jolla, California, 92038.
- Carretta, J.V. 2001. Preliminary estimates of cetacean mortality in California gillnet fisheries for 2000. Paper SC/53/SM9 presented to the International Whaling Commission, 2001 (unpublished), 21 pp. Available from NMFS, Southwest Fisheries Science Center, P.O. Box 271, La Jolla, California, 92038.
- Chinn, S.M., M.A. Miller, M.T. Tinker, M.M. Staedler, F.I. Batac, E.M. Dodd, L.A. Henkel. 2016. The high cost of motherhood: end-lactation syndrome in southern sea otters. *Journal of Wildlife Diseases* 52:307-318. doi: 10.7589/2015-06-158.
- Conrad P.A., M.A. Miller, C. Kreuder, E.R. James, J. Mazet, H. Dabritz, D.A. Jessup, F. Gulland, M.E. Grigg. 2005. Transmission of toxoplasma: clues from the study of sea otters as sentinels of *Toxoplasma gondii* flow into the marine environment. *International Journal for Parasitology* 35:1155-1168.
- Cronin, M.A., J. Bodkin, B. Bellachey, J.A. Estes, and J.C. Patton. 1996. Mitochondrial-DNA variation among subspecies and populations of sea otters (*Enhydra lutris*). *Journal of Mammalogy* 77:546-557.
- Dubey, J.P., N.L. Miller, and D.K. Frenkel. 1970. *Toxoplasma gondii* life cycle in cats. *Journal of the American Veterinary Medical Association* 157:1767-1770.
- Estes, J.A. 1990. Growth and equilibrium in sea otter populations. *J. Anim. Ecol.* 59:385-401.
- Estes, J.A. and R.J. Jameson. 1988. A double-survey estimate for sighting probability of sea otters in California. *Journal of Wildlife Management* 52:70-76.

- Estes, J.A., B.B. Hatfield, K. Ralls, and J. Ames. 2003. Causes of mortality in California sea otters during periods of population growth and decline. *Marine Mammal Science* 19:198-216.
- Forney, K.A., S.R. Benson, and G.A. Cameron. 2001. Central California gill net effort and bycatch of sensitive species, 1990-1998. Pages 141-160 *in* *Seabird Bycatch: Trends, Roadblocks, and Solutions*, E.F. Melvin and J.K. Parrish, eds. Proceedings of an International Symposium of the Pacific Seabird Group, University of Alaska Sea Grant, Fairbanks, Alaska, 212 pp.
- Gazeau F., L.M. Parker, S. Comeau, J.-P. Gattuso, W.A. O'Connor, S. Martin, H.-O. Pörtner, and P.M. Ross. 2013. Impacts of ocean acidification on marine shelled molluscs. *Marine Biology* 160:2207-45.
- Gerber, L.R., M.T. Tinker, D.F. Doak, J.A. Estes, and D.A. Jessup. 2004. Mortality sensitivity in life-stage simulation analysis: a case study of southern sea otters. *Ecological Applications* 14:1554-1565.
- Hatfield, B.B. and J.A. Estes. 2000. Preliminary results of an evaluation of the potential threat to sea otters posed by the nearshore finfish trap fishery. Unpublished, 6 pp. + appendices.
- Hatfield, B.B., J.A. Ames, J.A. Estes, M.T. Tinker, A.B. Johnson, M.M. Staedler, and M.D. Harris. 2011. Sea otter mortality in fish and shellfish traps: estimating potential impacts and exploring possible solutions. *Endangered Species Research* 13:219–229.
- Herrick, S.F. Jr. and D. Hanan. 1988. A review of California entangling net fisheries, 1981-1986. National Oceanic and Atmospheric Administration Technical Memorandum. National Marine Fisheries Service. NOAA-TM-NMFS-SWFC-108, 39 pp.
- Jameson, R.J. 1989. Movements, home range, and territories of male sea otters off central California. *Marine Mammal Science* 5:159-172.
- Jameson, R.J. and S. Jeffries. 1999. Results of the 1999 survey of the Washington sea otter population. Unpublished report, 5 pp.
- Jameson, R.J. and S. Jeffries. 2005. Results of the 2005 survey of the reintroduced Washington sea otter population. Unpublished report, 6 pp.
- Jessup D.A., M.A. Miller, M. Harris, B.B. Hatfield, and J.A. Estes. 2004. The 2003 southern sea otter (*Enhydra lutris nereis*) unusual mortality event: A preliminary report to NOAA and USFWS. Unpublished report, 38 pp.
- Johnson, C.K., M.T. Tinker, J.A. Estes, P.A. Conrad, M. Staedler, M.A. Miller, D.A. Jessup and J.A.K. Mazet. 2009. Prey choice and habitat use drive sea otter pathogen exposure in a resource-limited coastal system. *PNAS* 106:2242-2247.
- Kannan, K., E. Perrotta, and N.J. Thomas. 2006. Association between perfluorinated compounds and pathological conditions in southern sea otters. *Environmental Science & Technology* 40:4943-4948.
- Kannan, K., E. Perrotta, N.J. Thomas, and K.M. Aldous. 2007. A comparative analysis of polybrominated diphenyl ethers and polychlorinated biphenyls in southern sea otters that died of infectious diseases and noninfectious causes. *Archives of Environmental Contamination and Toxicology* 53:293–302.
- Kannan K., K.S. Guruge, N.J. Thomas, S. Tanabe, J.P. Giesy. 1998. Butyltin residues in southern sea otters (*Enhydra lutris nereis*) found dead along California coastal waters. *Environmental Science and Technology* 32:1169-1175.

- Kooyman, G.L. and D.P. Costa. 1979. Effects of oiling on temperature regulation in sea otters. Yearly progress report, Outer Continental Shelf Energy Assessment Program.
- Kreuder, C., M.A. Miller, D.A. Jessup, L.J. Lowenstein, M.D. Harris, J.A. Ames, T.E. Carpenter, P.A. Conrad, and J.A.K. Mazet. 2003. Patterns of mortality in southern sea otters (*Enhydra lutris nereis*) from 1998-2001. *Journal of Wildlife Diseases* 39:495-509.
- Kreuder, C., M.A. Miller, L.J. Lowenstein, P.A. Conrad, T.E. Carpenter, D.A. Jessup, and J.A.K. Mazet. 2005. Evaluation of cardiac lesions and risk factors associated with myocarditis and dilated cardiomyopathy in southern sea otters (*Enhydra lutris nereis*). *American Journal of Veterinary Research* 66:289-299.
- Kroeker, K. J., R.L. Kordas, R. N. Crim, and G.G. Singh. 2010. Meta-analysis reveals negative yet variable effects of ocean acidification on marine organisms. *Ecology Letters* 13:1419-1434.
- Kudela, R.M., J.Q. Lane, W.P. Cochlan. 2008. The potential role of anthropogenically derived nitrogen in the growth of harmful algae in California, USA. *Harmful Algae* 8:103-110.
- Kurihara, H., and Y. Shirayama. 2004. Effects of increased atmospheric CO₂ on sea urchin early development. *Marine Ecology Progress Series* 274:161-169.
- Kurihara, H., T. Asai, S. Kato, and A. Ishimatsu. 2008. Effects of elevated pCO₂ on early development in the mussel *Mytilus galloprovincialis*. *Aquatic Biology* 4:225-233.
- Lafferty, K.D. M.T. and Tinker. 2014. Sea otters are recolonizing southern California in fits and starts. *Ecosphere* 5:50. <http://dx.doi.org/10.1890/ES13-00394.1>
- Laidre, K.L., R.J. Jameson, and D.P. DeMaster. 2001. An estimation of carrying capacity for sea otters along the California coast. *Marine Mammal Science* 17:294-309.
- Larson, S., R. Jameson, J. Bodkin, M. Staedler, and P. Bentzen. 2002. Microsatellite DNA and mitochondrial DNA variation in remnant and translocated sea otter (*Enhydra lutris*) populations. *Journal of Mammalogy* 83:893-906.
- Mayer, K.A., M.D. Dailey, and M.A. Miller. 2003. Helminth parasites of the southern sea otter *Enhydra lutris nereis* in central California: abundance, distribution, and pathology. *Diseases of Aquatic Organisms* 53:77-88.
- Miller, M.A., M.E. Grigg, C. Kreuder, E.R. James, A.C. Melli, P.R. Crosbie, D.A. Jessup, J.C. Boothroyd, D. Brownstein, and P.A. Conrad. 2004. An unusual genotype of *Toxoplasma gondii* is common in California sea otters (*Enhydra lutris nereis*) and is a cause of mortality. *International Journal for Parasitology* 34:275-284.
- Miller, M.A., I.A. Gardner, C. Kreuder, D.M. Paradies, K.R. Worcester, D.A. Jessup, E. Dodd, M.D. Harris, J.A. Ames, A.E. Packham, and P.A. Conrad. 2002. Coastal freshwater runoff is a risk factor for *Toxoplasma gondii* infection of southern sea otters (*Enhydra lutris nereis*). *International Journal for Parasitology* 32:997-1006.
- Miller, M.A., W.A. Miller, P.A. Conrad, E.R. James, A.C. Melli, C.M. Leutenegger, H.A. Dabritz, A.E. Packham, D. Paradies, M. Harris, J. Ames, D.A. Jessup, K. Worcester, M.E. Grigg. 2008. Type X *Toxoplasma gondii* in a wild mussel and terrestrial carnivores from coastal California: new linkages between terrestrial mammals, runoff and toxoplasmosis of sea otters. *International Journal for Parasitology* 38:1319-28.
- Miller, M.A., R.M. Kudela, A. Mekebri, D. Crane, S.C. Oates, M.T. Tinker, M. Staedler, W.A. Miller, S. Toy-Choutka, C. Domink, D. Hardin, G. Langlois, M. Murray, K. Ward and D.A. Jessup. 2010. Evidence for a novel marine harmful algal bloom: cyanotoxin (Microcystin) transfer from land to sea otters. *PLoS ONE* 5:e12576.

- Monson, D.H., J.A. Estes, J.L. Bodkin, and D.B. Siniff. 2000. Life history plasticity and population regulation in sea otters. *Oikos* 90:457-468.
- Mos, L. 2001. Domoic acid: a fascinating marine toxin. *Environmental Toxicology and Pharmacology* 9:79-85.
- Nakata, H., K. Kannan, L. Jing, N. Thomas, S. Tanabe, and J.P. Giesy. 1998. Accumulation pattern of organochlorine pesticides and polychlorinated biphenyls in southern sea otters (*Enhydra lutris nereis*) found stranded along coastal California, USA. *Environmental Pollution* 103:45-53.
- Ralls, K., T.C. Eagle, and D.B. Siniff. 1996. Movement and spatial use patterns of California sea otters. *Canadian Journal of Zoology* 74:1841-1849.
- Sanchez, M.S. 1992. Differentiation and variability of mitochondrial DNA in three sea otter, *Enhydra lutris*, populations. M.S. Thesis, University of California Santa Cruz.
- Siniff, D.B., T.D. Williams, A.M. Johnson, and D.L. Garshelis. 1982. Experiments on the response of sea otters, *Enhydra lutris*, to oil contamination. *Biological Conservation* 2: 261-272.
- Stumpff, M., J. Wren, Frank Melzner, M. C. Thorndyke, and S. T. Dupont. 2011. CO₂ induced seawater acidification impacts sea urchin larval development I: Elevated metabolic rates decrease scope for growth and induce developmental delay. *Comparative Biochemistry and Physiology, Part A: Molecular & Integrative Physiology* 160:331-340.
- Taylor, B.L., M. Scott, J. Heyning, and J. Barlow. 2003. Suggested guidelines for recovery factors for endangered marine mammals. NOAA Technical Memorandum NOAA-TM-NMFS-SWFSC-354.
- Tinker, M.T. 2014. Models and sea otter conservation. Pp. 257-300 in Larson, S., G. VanBlaricom and J. Bodkin, eds., *Sea Otter Conservation*. New York: Elsevier.
- Tinker, M.T., and B.B. Hatfield. 2016. California sea otter (*Enhydra lutris nereis*) census results, spring 2016. U.S. Geological Survey Data Series 1018. 10 pp. <http://dx.doi.org/10.3133/ds1018>.
- Tinker, M.T., G. Bentall, and J.A. Estes. 2008a. Food limitation leads to behavioral diversification and dietary specialization in sea otters. *PNAS* 105:560-565.
- Tinker, M.T., D.F. Doak, and J.A. Estes. 2008b. Using demography and movement behavior to predict range expansion of the southern sea otter. *Ecological Applications* 18:1781-1794.
- Tinker, M.T., B.B. Hatfield, M.D. Harris, and J.A. Ames. 2016. Dramatic increase in sea otter mortality from white sharks in California. *Marine Mammal Science* 32:309–326.
- Tinker, M.T., J.A. Estes, K. Ralls, T.M. Williams, D. Jessup, and D.P. Costa. 2006a. Population Dynamics and Biology of the California Sea Otter (*Enhydra lutris nereis*) at the Southern End of its Range. MMS OCS Study 2006-007. Coastal Research Center, Marine Science Institute, University of California, Santa Barbara, California. MMS Cooperative Agreement Number 14-35-0001-31063.
- Tinker, M.T., D.F. Doak, J.A. Estes, B.B. Hatfield, M.M. Staedler, and J. Bodkin. 2006b. Incorporating diverse data and realistic complexity into demographic estimation procedures for sea otters. *Ecological Applications* 16:2293-2312.
- Tinker, M.T., D. Jessup, M. Staedler, M. Murray, M. Miller, T. Burgess, E. Bowen, K. Miles, J. Tomoleoni, N. Thometz, L. Tarjan, E. Golson, F. Batac, E. Dodd, E. Berberich, J. Kunz, G. Bentall, T. Nicholson, S. Newsome, H. MacCormick, A. Melli, A. Johnson, L. Henkel, C. Kreuder-Johnson, and P. Conrad. 2013. Sea otter population biology at Big

- Sur and Monterey California: investigating the consequences of resource abundance and anthropogenic stressors for sea otter recovery. Draft Final Report to California Coastal Conservancy and U.S. Fish and Wildlife Service. University of California, Santa Cruz, 243 pp.
- U.S. Fish and Wildlife Service. 2003. Final Revised Recovery Plan for the Southern Sea Otter (*Enhydra lutris nereis*). Portland, Oregon, xi + 165 pp.
- U.S. Fish and Wildlife Service. 2015. Southern Sea Otter (*Enhydra lutris nereis*) 5-Year Review: Summary and Evaluation. Ventura, California. 42 pp.
- Valentine, K., D.A. Duffield, L.E. Patrick, D.R. Hatch, V.L. Butler, R.L. Hall, and N. Lehman. 2008. Ancient DNA reveals genotypic relationships among Oregon populations of the sea otter (*Enhydra lutris*). *Conservation Genetics* 9:933-938.
- Vezie, C., J. Rapala, J. Vaitomaa, J. Seitsonen, and K. Sivonen. 2002. Effect of nitrogen and phosphorus on growth of toxic and nontoxic *Microcystis* strains and on intracellular microcystin concentrations. *Microbial Ecology* 43:443-454.
- Walther, G.-R, E. Post, P. Convey, A. Menzel, C. Parmesan, T.J.C. Beebee, J.-M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein. 2002. Ecological responses to recent climate change. *Nature* 416:389-395.
- Wendell, F.E., R.A. Hardy, and J.A. Ames. 1986. An assessment of the accidental take of sea otters, *Enhydra lutris*, in gill and trammel nets. California Department of Fish and Game, Mar. Res. Tech. Rep. No. 54, 31 pp.
- Wilson, D.E., M.A. Bogan, R.L. Brownell, Jr., A.M. Burdin, and M.K. Maminov. 1991. Geographic variation in sea otters, *Enhydra lutris*. *Journal of Mammalogy* 72:22-36.