U.S. Pacific Marine Mammal Stock Assessments: 2024

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NOAA Technical Memorandum TM-PIFSC-182

May 2026

U.S. Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Pacific Islands Fisheries Science Center

Acknowledgements

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Edited by Revere Wood & Jill Coyle

Cover page photo caption: Monk seal sleeping on a beach with NOAA Ship *Oscar Elton Sette* in the background. Credit: NOAA Fisheries / Mark Sullivan. Permit # 22677.

Recommended citation

Carretta, J. V., Baker, J., Orr, A. J., Melin, S., Hanson, B., Van Ee, J., Wallen, M., Chinn, S., & Mahoney, P. J. (2026). *U.S. Pacific Marine Mammal Stock Assessments: 2024* (PIFSC Technical Memorandum Series, TM-PIFSC-182). Pacific Islands Fisheries Science Center. https://doi.org/10.25923/cxs2-8t22

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Guadalupe Fur Seal *Arctocephalus townsendi*

Stock Definition and Geographic Range

Commercial sealing during the 19th century reduced the once abundant Guadalupe fur seal to near extinction in 1894 (Townsend, 1931). Prior to the harvest it ranged from Monterey Bay, California, to the Revillagigedo Islands, Mexico (Hanni et al., 1997; Repenning et al., 1971; Figure 1). The prehistoric distribution of Guadalupe fur seals during the Holocene was apparently quite different from today; the archeological record indicates Guadalupe fur seal remains accounted for 40–80% of all pinniped bones at the California Channel Islands (Rick et al., 2009). The live capture of two adult males (and killing of ~60 more animals) at Guadalupe Island in 1928 established the continued existence of the species (Townsend, 1931).

Guadalupe fur seals pup and breed mainly at Isla Guadalupe, Mexico. In 1997, a second rookery was discovered at Isla Benito del Este, Baja California (Maravilla-Chavez & Lowry, 1999) and a pup was born at San Miguel Island, California (Melin & DeLong, 1999). Since 2008, individual adult females, subadult males, and between one and three pups have been observed annually on San Miguel Island (NMFS, unpublished data). The population at Isla Benito del Este is now well-established, though very few pups are observed there. Population increases at Isla San Benito are attributed to immigration of animals from Isla Guadalupe (Aurioles-Gamboa et al., 2010; García-Capitanachi, 2011).

Along the U.S. West Coast, strandings occur annually in California waters and animals are increasingly observed in Oregon and Washington waters. During 2015–2021, a total of 715 Guadalupe fur seals were observed stranded in U.S. West Coast waters and NMFS declared an [Unusual Mortality Event] (UME), which closed in 2021 (see section below on other factors that may be causing a decline or impeding recovery). Individuals have stranded or been sighted in the Gulf of California and south to Zihuatanejo, Mexico (Hanni et al., 1997; Aurioles-Gamboa & Hernadez-Camacho, 1999) Cerro Hermoso, Oaxaca, Mexico (Esperon-Rodriguez & Gallo-Reynoso, 2012), and the Galapagos Islands (Páez-Rosas et al., 2020). Multiple sightings have been reported from the Bering Sea and Aleutian Islands region (Pace et al., 2022).

There are several records of Guadalupe fur seals being hooked in the mouth by longline gear in the Hawai‘i shallow set longline fishery (Carretta et al., 2024). Guadalupe fur seals that stranded in central California and treated at rehabilitation centers were fitted with satellite tags and documented to travel as far north as Graham Island and Vancouver Island, British Columbia, Canada (Norris et al., 2015). Some satellite-tagged animals traveled far offshore outside the U.S. Exclusive Economic Zone (EEZ) to areas 700 nmi west of the California / Oregon border. The population is considered to be a single stock because all are recent descendants from one breeding colony at Isla Guadalupe, Mexico.

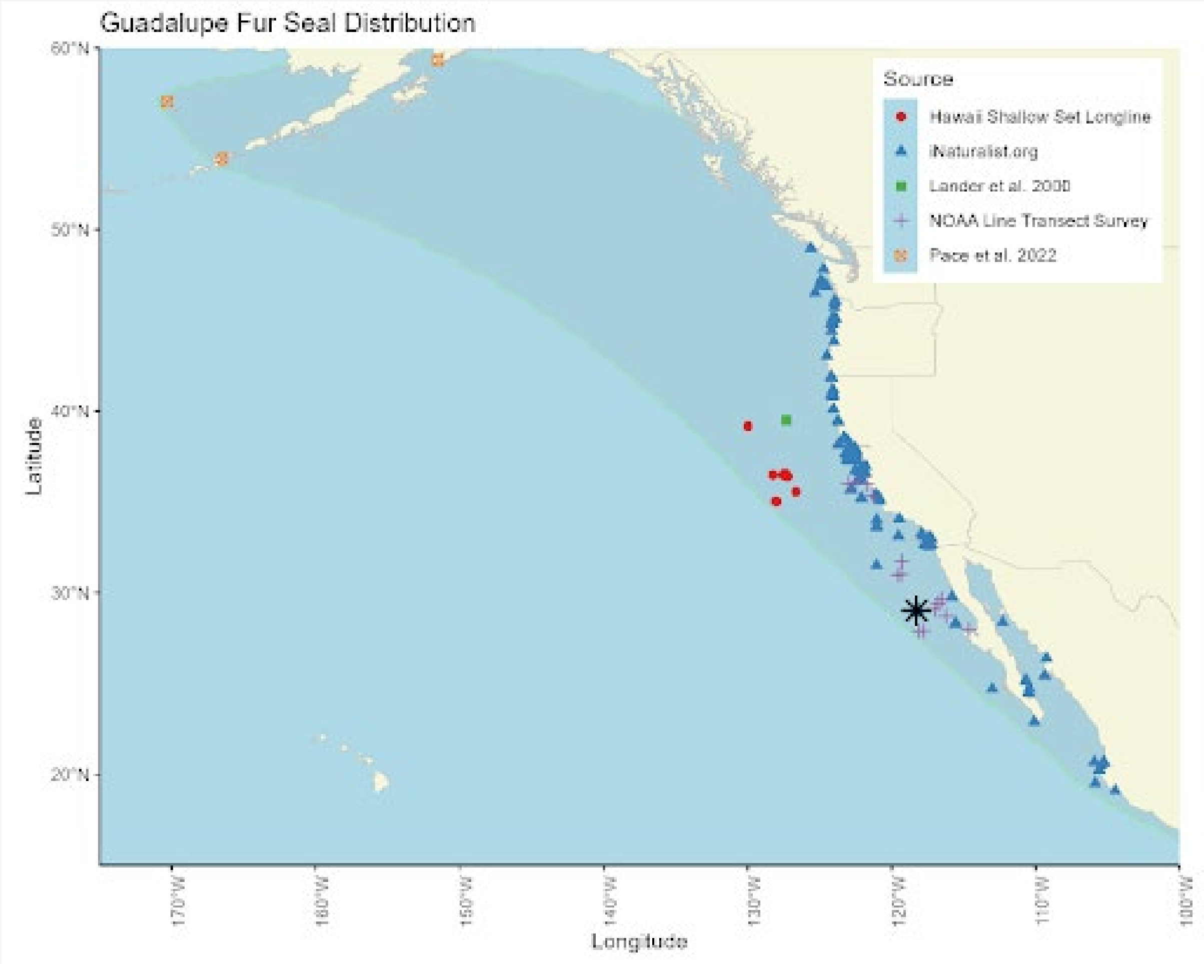

Figure 1. Sighting data and approximate range of the Guadalupe fur seal. The location of Guadalupe Island is noted with the large black asterisk.

Population Size

Population size prior to commercial harvests in the 19th century is unknown, but estimates range from 20,000 to 200,000 animals (Fleischer, 1987; Hubbs, 1979). Juárez-Ruiz et al. (2022) estimated pup production for the period 1991–2019 and estimated total population size in 2019 at 63,850 animals (range: 57,199–72,631), based on a total population to pup ratio of 4:1 (estimated for the northern fur seal, (Johnson, 1975). This expansion factor is the mean of previous expansion factors used by García-Aguilar et al. (2018) to estimate total Guadalupe fur seal population size from pup counts, based on the work of Harwood and Prime (1978). Additional, unpublished estimates of total population size appear in a Guadalupe fur seal workshop report (Marine Mammal Commission, 2023), but until new estimates are published, the best estimate of Guadalupe fur seal abundance is considered to be the 2019 estimate of 63,850 animals reported by Juárez-Ruiz et al. (2022) (Figure 2).

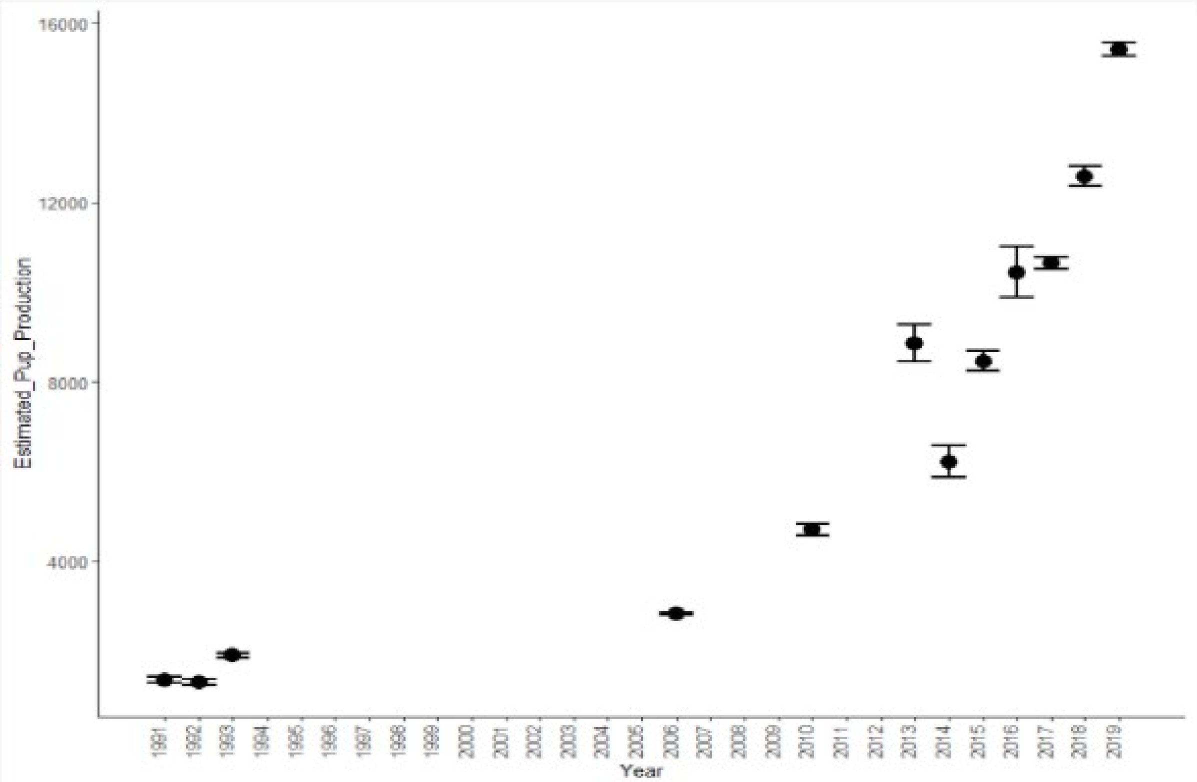

Figure 2. Estimated pup production of Guadalupe fur seals reported by Juárez-Ruiz et al. (2022) for 1991–2019. Horizontal lines represent point estimates ± 2 standard deviations.

Minimum Population Estimate

The minimum population size is taken as the lower bound of the estimate provided by Juárez-Ruiz et al. (2022), or 57,199 animals.

Current Population Trend

Juárez-Ruiz et al. (2022) estimated an annual growth rate of 8.4% (range: 8–8.8) from 1991–2019, which is higher than the 5.9% estimated for 1984–2013 (García-Aguilar et al., 2018).

Current and maximum net productivity rates

Reported annual growth rates of 21% at Isla San Benito over an 11-year period are too high for intrinsic growth and likely result from immigration from Isla Guadalupe (Esperón-Rodríguez & Gallo-Reynoso, 2012). The maximum net productivity rate is assumed to be equal to the maximum annual growth rate observed between 1955 and 1993 (13.7%) when the population was at a very low level and should have been growing at nearly its maximum rate (Gallo, 1994). The current maximum net productivity rate is taken as the most-recent growth estimate provided by Juárez-Ruiz et al. (2022), or 8.4%.

Potential Biological Removal

The potential biological removal (PBR) for this stock is calculated as the minimum population size (57,199) **x** one half the maximum net growth rate observed for this species (½ of 13.7%) **x** a recovery factor of 0.5 (for a threatened species, Wade & Angliss, 1997), resulting in a PBR of 1,959 Guadalupe fur seals per year. The vast majority of this PBR would apply towards incidental mortality in Mexico as most of the population occurs outside of U.S. waters. The fraction of this stock that occurs in U.S. waters and the amount of time spent in U.S. waters is unknown, thus, a proration factor for calculating a PBR in U.S. waters is not available.

Fisheries Information

Table 1. Summary of available information on the incidental mortality and serious injury (MSI) of Guadalupe fur seals in commercial fisheries and other unidentified fisheries.

Fishery Name	**Year(s)**	**Data Type**	**Percent Observer Coverage**	**Sum of Observed Mortality and Serious Injury (MSI)**	**Estimated Mortality and Serious Injury (CV)**	**Mean Annual Takes (CV)**
CA driftnet fishery for sharks and swordfish	2018–2022	Observer	20%–25%	0	0	0
CA set gillnet fishery for halibut/white seabass and other species	2013–2017	Observer	<10%	0	0	0
Hawaiʻi Shallow Set Longline Fishery	2018–2022	Observer	100%	4	4	0.8 (n/a)
Unidentified fishery interactions, including gillnets and trawls of unknown origin	2018–2022	Strandings	n/a	32	≥32	≥6.4
Minimum total annual takes						≥7.2 (n/a)

No Guadalupe fur seals have been observed entangled in California gillnet fisheries between 1990 and 2022 (Julian & Beeson, 1998; Carretta, 2023), although stranded animals have been found entangled in gillnet of unknown origin (Other mortality and serious injury). Gillnets have been documented to entangle marine mammals off Baja California (Sosa-Nishizaki et al., 1993), but no recent bycatch data from Mexico are available.

Guadalupe fur seals occasionally are observed hooked in the Hawaiʻi shallow set longline fishery (100% observer coverage, Table 1). Between 2018 and 2022, there was 1 death, 3 serious injuries, and 5 non-serious injuries involving this species (Carretta et al., 2024). These interactions occurred outside of the U.S. EEZ, west of the California Current.

Most records of serious injury and mortality of Guadalupe fur seals with fishing gear is derived from opportunistic stranding data, which includes 32 mortality / serious injury cases involving gillnet and trawl fisheries (Table 1).

Other mortality and serious injury

Other human-related mortality and serious injury during 2018–2022 included entanglement in marine debris (∑MSI = 8), shootings (∑MSI = 3), unidentified human interactions (∑MSI = 2) and oil/tar (∑MSI = 1). The average annual observed human-caused mortality and serious injury of Guadalupe fur seals for 2018–2022 from non-fishery sources is 2.8 animals (14 animals / 5 years). Observed human-caused mortality and serious injury for this stock very likely represents a fraction of the true impacts because not all cases are documented. No correction factors to account for undetected mortality and injury are currently available for pinnipeds along the U.S. West Coast.

Status of Stock

The Endangered Species Act (ESA) lists the Guadalupe fur seal as a threatened species, which automatically qualifies this stock as "depleted" and "strategic" stock under the Marine Mammal Protection Act (MMPA). There is insufficient information to determine levels of human-caused mortality and serious injury in Mexico. The total U.S. commercial fishery mortality and serious injury for this stock (≥7.2 animals per year) is less than 10% of the calculated PBR (1,959) for the entire stock, but it is not currently possible to calculate a prorated PBR for U.S. waters with which to compare serious injury and mortality from U.S. fisheries.

Therefore, it is unknown whether total U.S. fishery mortality is insignificant and approaching zero mortality and serious injury rate. The combined annual serious injury and mortality from commercial fisheries (≥7.2) and other sources (≥2.8) is 10 animals per year, which is less than the range-wide PBR of 1,959 animals for this stock. The population was estimated to grow at 8.4% annually for the period 1991–2019 (Juárez-Ruiz et al., 2022).

Other factors that may be causing a decline or impeding recovery

Guadalupe fur seals may be negatively affected by marine heatwaves (Cavole et al., 2016; Gálvez et al., 2023), including reduction and changes in prey availability, with impacts to pup and juvenile survival. The 2015–2021 UME coincided with multiple marine heatwaves.

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Hawaiian Monk Seal *Neomonachus schauinslandi*

Stock Definition and Geographic Range

Hawaiian monk seals are distributed throughout the Northwestern Hawaiian Islands (NWHI), with subpopulations at French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Atoll, Kure Atoll, and Necker and Nihoa Islands. They also occur throughout the main Hawaiian Islands. Genetic variation among monk seals is extremely low and may reflect a long-term history at low population levels and more recent human influences (Kretzmann et al., 1997, 2001; Schultz et al., 2009). Though monk seal subpopulations often exhibit asynchronous variation in demographic parameters—such as abundance trends and survival rates—they are connected by animal movement throughout the species' range (Johanos et al., 2013). Genetic analysis (Schultz et al., 2011) indicates the species is a single panmictic population. The Hawaiian monk seal is, therefore, considered a single stock. Scheel et al. (2014) established a new genus, *Neomonachus*, comprising the Caribbean and Hawaiian monk seals, based upon molecular and skull morphology evidence.

Population Size

The best estimate of the total population size is 1,605 (95% confidence interval 1,512–1,743; CV = 0.05), (Table 2; Johanos, 2024a, 2024b, 2024c). Obtaining abundance estimates for all NWHI subpopulations requires sea-going vessel support for approximately 56 days. Methods for abundance estimation vary by site and year depending on the type and quantity of data available. Total enumeration is the favored method, but it requires sufficient field presence to convincingly identify all the seals present. This is typically not achieved at most sites (Baker et al., 2006).

When total enumeration is not possible, capture-recapture estimates (using the program CAPTURE) are conducted (Baker, 2004; Otis et al., 1978; Rexstad & Burnham, 1991; White et al., 1982). When no reliable estimator is obtainable in CAPTURE (i.e., the model selection criterion is <0.75, following Otis et al., 1978), total non-pup abundance is estimated using pre-existing information on the relationship between proportion of the population identified and field effort hours expended (referred to as discovery curve analysis). At rarely visited sites (Necker, Nihoa, Niʻihau, and Lehua Islands), where data are insufficient to use any of the above methods, beach counts are corrected for the proportion of seals at sea. In the main Hawaiian Islands, other than Niʻihau and Lehua Islands, abundance is estimated as the minimum tally of all individuals identified by an established sighting network during the calendar year. At all sites, pups are tallied.

Finally, site-specific abundance estimates and their uncertainty are combined using Monte Carlo methods to obtain a range-wide abundance estimate distribution. All the above methods are described or referenced in Baker et al. (2016) and Harting et al. (2017). Since some of the abundance estimation methods utilize empirical distributions, which are updated as new data accrue, previous years' estimates can change slightly when recalculated using these updated distributions.

In 2022, total enumeration was not achieved at any subpopulation. Consequently, capture-recapture estimates were obtained at Lisianski Island, Pearl and Hermes Reef, and Midway Atoll. Discovery curve analysis was used to generate abundance estimates at French Frigate Shoals, Laysan Island, and Kure Atoll (Table 2). Counts at Necker and Nihoa Islands are typically conducted from zero to a few times per year. Pups are born over the course of many months and have very different haulout patterns compared to older animals. Therefore, pup production at Necker and Nihoa Islands is estimated as the mean of the total pups observed in the past 5 years, excluding counts occurring early in the pupping season when most have yet to be born.

In the main Hawaiian Islands, NOAA Fisheries collects information on seal sightings reported throughout the year by a variety of sources, including a volunteer network, the public, and directed NOAA Fisheries observation effort. A small number of surveys of Niʻihau and nearby Lehua Islands are conducted through a collaboration between NMFS, Niʻihau residents, and the U.S. Navy.

Total main Hawaiian Islands monk seal abundance is estimated by adding the number of individually identifiable seals documented during a calendar year on all main Hawaiian Islands other than Niʻihau and Lehua to an estimate for these latter two islands based on counts expanded by a haulout correction factor. A telemetry study (Wilson et al., 2017) found that main Hawaiian Islands monk seals (N=23) spent a greater proportion of time ashore than Harting et al. (2017) estimated for NWHI seals. Therefore, the total non-pup estimate for Niʻihau and Lehua Islands was the total beach count at those sites (fewer individual seals already counted at other main Hawaiian Islands) divided by the mean proportion of time hauled out in the main Hawaiian Islands (Wilson et al., 2017). The total pups observed at Niʻihau and Lehua Islands were added to obtain the total (Table 2).

Table 2. Total and minimum estimated abundance (*N*min) of Hawaiian monk seals by location in 2022. The estimation method is indicated for each site. Methods used include DC: discovery curve analysis, EN: total enumeration; CR: capture-recapture; CC: counts corrected for the proportion of seals at sea; Min: minimum tally. Median values are presented. The median range-wide abundance is not equal to the total of the individual sites' medians, because the median of sums may differ from the sum of medians for non-symmetrical distributions. *N*min for individual sites are either the minimum number of individuals identified or the 20th percentile of the abundance distribution (the latter applies to Necker, Nihoa, Niʻihau / Lehua, and range-wide).

-	**Total**			***N*min**			-
Location	**Non-pups**	**Pups**	**Total**	**Non-pups**	**Pups**	**Total**	**Method**
French Frigate Shoals	198	45	243	195	45	240	DC
Laysan	197	53	250	195	53	248	DC
Lisianski	143	19	162	138	19	157	CR
Pearl & Hermes Reef	126	27	153	116	27	143	CR
Midway	65	10	75	61	10	71	CR
Kure	89	23	112	89	23	112	DC
Necker	85	8	93	71	8	79	CC
Nihoa	75	2	77	63	2	65	CC
MHI Kauai to Hawaiʻi	190	25	215	190	25	215	Min
Niʻihau /Lehua	191	17	208	161	17	178	CC
Range-wide total	1376	229	1605	1279	229	1508	-

Minimum Population Estimate

The total numbers of seals identified at the NWHI subpopulations other than Necker and Nihoa, and in the main Hawaiian Islands other than Niʻihau and Lehua, are the best estimates of minimum population size at those sites. Minimum population sizes for Necker, Nihoa, Niʻihau, and Lehua Islands are estimated as the lower 20th percentiles of the non-pup abundance distributions generated using haulout corrections as described above, plus the pup estimates. The minimum abundance estimates for each site and for all sites combined (1,508) are presented in Table 2.

Current Population Trend

Range-wide abundance estimates are available from 2013 to 2022 (Table 2, Figure 3). While these estimates remain somewhat negatively-biased for reasons explained in Baker et al. (2016), they provided a robust and consistent assessment of status and trends. A Monte Carlo approximation of the annual multiplicative rate of realized population growth during 2013–2022 was generated by fitting 10,000 log-linear regressions to randomly selected values from each year's abundance distributions. The median rate (and 95% confidence limits) is 1.02 (1.02, 1.03). Thus, the best estimate is that the population grew at an average rate of about 2% per year from 2013–2022.

Current and maximum net productivity rates

Mean non-pup beach counts are used as a long-term index of abundance for years when data are insufficient to estimate total abundance as described above. Prior to 1999, beach count increases of up to 7% annually were observed at Pearl and Hermes Reef, and this is the highest estimate of the maximum net productivity rate (R_{max}) observed for this species (Johanos, 2024a). Consistent with this value, a life table analysis representing a time when the main Hawaiian Islands monk seal population was apparently expanding, yielded an estimated intrinsic population growth rate of 1.07 (Baker et al., 2011).

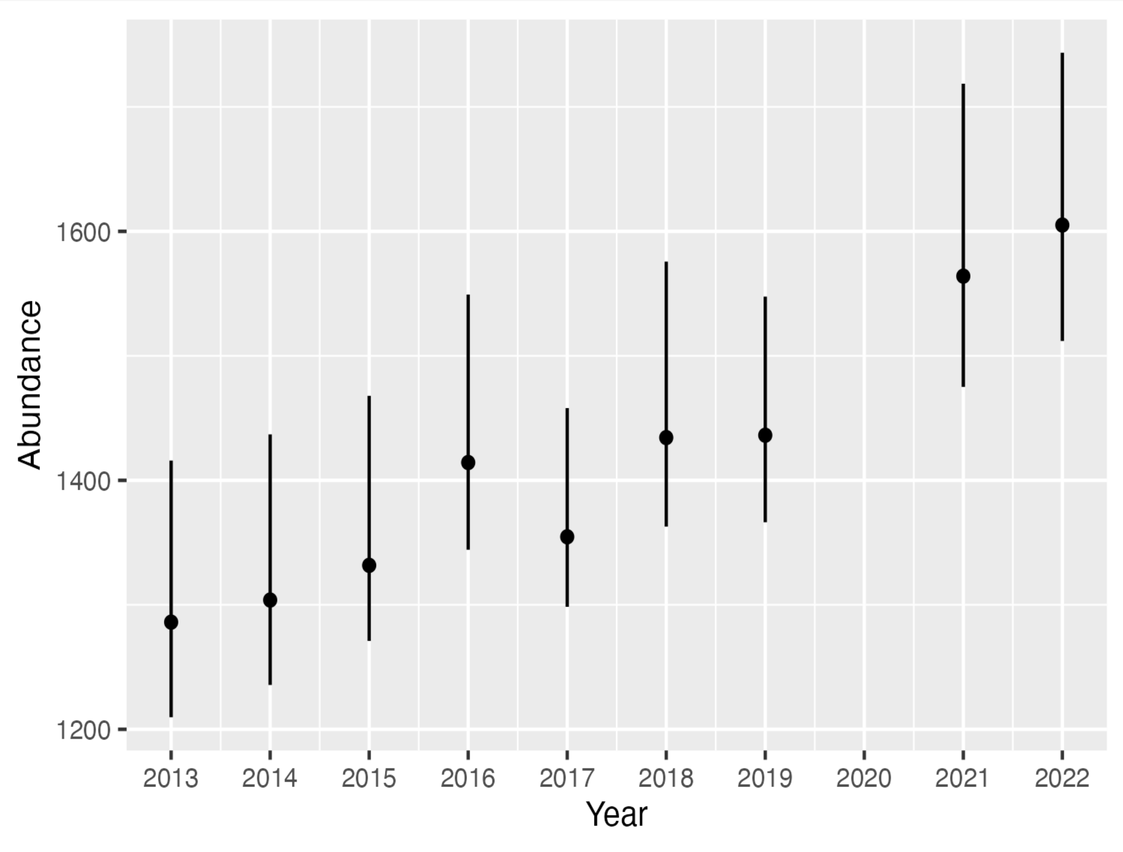

Figure 3. Range-wide abundance of Hawaiian monk seals, 2013-2022. Medians and 95% confidence limits are shown. Estimates prior to 2022 are re-estimated based on new data and represent negligible changes compared with values reported in the previous final stock assessments. (Table 2).

Potential Biological Removal

Using current minimum population size (1,508), R_{max} (0.07) and a recovery factor (F_r) for ESA endangered stocks (0.1), yields a Potential Biological Removal (PBR) of 5.3.

Human-Caused Mortality and Serious Injury

Human-related mortality has caused two major declines of the Hawaiian monk seal (Ragen, 1999). In the 1800s, this species was decimated by sealers, crews of wrecked vessels, and guano and feather hunters (Dill & Bryan, 1912; Wetmore, 1925; Bailey, 1952; Clapp & Woodward, 1972). Following a period of at least partial recovery in the first half of the 20th century (Rice, 1960), most subpopulations again declined. This second decline has not been fully explained, but long-term trends at several sites appear to have been driven both by variable oceanic productivity (represented by the Pacific Decadal Oscillation) and by human disturbance (Baker et al., 2012; Ragen, 1999; Kenyon, 1972; Gerrodette & Gilmartin, 1990). Currently, human activities in the NWHI are limited and human disturbance is relatively rare, but human-seal interactions are an important issue in the main Hawaiian Islands. Intentional killing of seals in the main Hawaiian Islands is an ongoing and serious concern (Table 3). In 2022, no intentional seal killings were documented.

Table 3. Intentional and potentially intentional killings of main Hawaiian Islands monk seals, and anthropogenic mortalities not associated with fishing gear during 2018–2022 (Johanos, 2024d; Mercer, 2024a, b). There were no confirmed cases in 2019, 2020, or 2022.

Year	Age/sex	Island	Cause of Death	Comments
2018	Juvenile female	Molokai	Blunt force trauma	Intentional
2021	Subadult male	Molokai	Blunt force trauma	Intentional
2021	Subadult male	Molokai	Blunt force trauma	Intentional
2021	Juvenile female	Molokai	Gunshot	Intentional

Harting et al. (2021) found that 46% of carcasses of monk seals which died in the main Hawaiian Islands during 2004–2019 were detected. Consequently, the cases in Table 3 must be considered a minimum representation of intentional killings.

Fishery Information

Fishery interactions with monk seals can include direct interaction with gear (hooking or entanglement), seal consumption of discarded or depredated catch, and competition for prey. Entanglement of monk seals in derelict fishing gear, which is believed to originate outside the Hawaiian archipelago, is described in a separate section. Fishery interactions are a serious concern in the main Hawaiian Islands, especially involving nearshore fisheries managed by the State of Hawaiʻi (Gobush et al., 2016). There are no fisheries operating in or near the NWHI. In 2022, 22 seal hookings were documented; one was classified as serious injuries and 21 as non-serious injuries. Of the non-serious injuries, 8 would have been deemed serious had they not been mitigated (Mercer, 2024a, 2024b). Monk seals also interact with nearshore gillnets, and several confirmed deaths have resulted. In 2022, one seal’s death was attributed to peracute underwater entrapment (drowning most likely due to net entanglement; Moore et al., 2013). No mortality or injuries have been attributed to the main Hawaiian Islands bottomfish handline fishery, and no interactions with longline fisheries have occurred since 1991.

Consequently, these fisheries are not included in Table 4. Published studies on monk seal prey selection based upon scat / spew analysis and video from seal-mounted cameras revealed evidence that monk seals fed on families of bottomfish which contain commercial species (many prey items recovered from scats and spews were identified only to the level of family; Goodman-Lowe, 1998; Longenecker et al., 2006; Parrish et al., 2000). Quantitative fatty acid signature analysis (QFASA) results support previous studies illustrating that monk seals consume a wide range of species (Iverson et al., 2011). However, deepwater-slope species, including two commercially targeted bottomfishes and other species not caught in the fishery, were estimated to comprise a large portion of the diet for some individuals. Similar species were estimated to be consumed by seals regardless of location, age or gender, but the relative importance of each species varied. Diets differed considerably between individual seals. These results highlight the need to better understand potential ecological interactions with the main Hawaiian Islands bottomfish handline fishery.

Table 4. Summary of mortality, serious and non-serious injury of Hawaiian monk seals due to fisheries and calculation of annual mortality rate. N/a indicates that sufficient data are not available. Total non-serious injuries are presented as well as, in parentheses, the number of those injuries that would have been deemed serious had they not been mitigated (e.g., by de-hooking or disentangling). Nearshore fisheries injuries and mortalities include seals entangled/drowned in nearshore gillnets and hooked/entangled in hook-and-line gear, recognizing that it is not possible to determine whether the nets or hook-and-line gear involved were being used for commercial purposes.

Fishery Name	Year	Data Type	% Obs. Coverage	Observed / Reported Mortality / Serious Injury	Estimated Mortality / Serious Injury	Non-serious (Mitigated serious)	Mean Takes (CV)
Nearshore	2018 2019 2020 2021 2022	Incidental observations of seals	None	0 3 4 2 2	n/a	11(3) 17(5) 29(4) 30(4) 21(8)	≥2.2
Mariculture	2018 2019 2020 2021 2022	Incidental Observation	None	0 0 0 0 0	n/a	0 0 0 1 0	0(0)
Minimum total annual takes	-					-	≥2.2

Fishery Mortality Rate

Total fishery mortality and serious injury is not considered to be insignificant and approaching a rate of zero. Monk seals are regularly hooked and entangled in the main Hawaiian Islands and the resulting deaths have substantially reduced the population growth rate (Harting et al., 2021). Monk seals also die from entanglement in fishing gear and other debris throughout their range (likely originating from various sources outside of Hawaiʻi). NOAA Fisheries and partners are working to mitigate entanglement (see below).

Entanglement in Marine Debris

Hawaiian monk seals become entangled in fishing and other marine debris at rates higher than reported for other pinnipeds (Henderson, 2001). Several hundred cases of debris entanglement have been documented in monk seals (nearly all in the NWHI), including ten documented deaths (Henderson, 2001; Johanos, 2024d; Mercer, 2024a, b). The number of marine debris entanglements documented in the past five years (Table 5) is an underestimate of the total impact of this threat because no people are present to document nor mitigate entanglements at most of the NWHI for the majority of the year. Moreover, seals that become entangled at sea and are unable to return to shore are very unlikely to be detected.

The low number of entanglements documented in 2020 is due to limited or no surveillance conducted at NWHI subpopulations due to the COVID pandemic. The fishing gear fouling the reefs and beaches of the NWHI and entangling monk seals only rarely includes types used in Hawaiʻi fisheries. For example, trawl net and monofilament gillnet accounted for approximately 35% and 34%, respectively, of the debris removed from reefs in the NWHI by weight, and trawl net alone accounted for 88% of the debris by frequency (Donohue et al., 2001), despite the fact that trawl fisheries have been prohibited in Hawaiʻi since the 1980s.

Table 5. Summary of documented marine debris entanglements of Hawaiian monk seals during the most recent five years. Total non-serious injuries are presented as well as, in parentheses, the number of those injuries that would have been deemed serious had the seals not been disentangled.

Year	Observed/Reported Mortality/Serious Injury	Non-serious (Mitigated serious)
2018	1	15(6)
2019	0	16(10)
2020	0	5(1)
2021	0	11(6)
2022	1	9(5)
Minimum total annual takes	≥ 0.4	-

NOAA Fisheries and partners continue to mitigate impacts of marine debris on monk seals as well as turtles, coral reefs, and other wildlife. Marine debris is removed from beaches and seals are disentangled during population assessment activities in the NWHI. Since 1996, annual debris survey and removal efforts in the NWHI coral reef habitat have been ongoing (Donohue et al., 2000, 2001; Dameron et al., 2007).

Toxoplasmosis

Land-to-sea transfer of *Toxoplasma gondii,* a protozoal parasite shed in the feces of cats, is of growing concern. Although the parasite can infect many species, felids are the definitive host, meaning it can only reproduce in cats. There are no native felids in Hawaiʻi, but several hundred thousand feral and domestic cats occur throughout the main Hawaiian Islands. As such, all monk seal deaths attributable to toxoplasmosis are considered human caused. A case definition for toxoplasmosis and other protozoal-related mortalities was developed and retrospectively applied to 306 cases of monk seal mortality from 1982–2015 (Barbieri et al., 2016).

During the past five years (2018–2022), seven monk seal deaths (representing a minimum average of 1.4 deaths per year) have been directly attributed to toxoplasmosis (Mercer, 2024a). Five of the seven deaths involved female seals. The number of deaths from this pathogen are likely underrepresented, given that more seals disappear each year than are found dead and examined (Harting et al., 2021), and the potential for chronic infections remains poorly understood in this species. Furthermore, *T. gondii* can be transmitted vertically from dam to fetus, and failed pregnancies are difficult to detect in wild, free-ranging animals.

Unlike threats such as hook ingestion or malnutrition, which can often be mitigated through rehabilitation, options for treating seals with toxoplasmosis are challenging and two attempts have not been successful. The accumulating number of monk seal deaths from toxoplasmosis in recent years is a growing concern given the increasing geographic overlap between humans, cats, and Hawaiian monk seals in the main Hawaiian Islands.

Other Mortality

Sources of mortality that impede recovery include food limitation, single and multiple-male intra-species aggression (mobbing), shark predation, and disease. Male seal aggression has caused episodes of mortality and injury. Past interventions to remove aggressive males greatly mitigated, but have not eliminated, this source of mortality (Johanos et al., 2010). Galapagos shark predation on monk seal pups has been a chronic and significant source of mortality at French Frigate Shoals since the late 1990s, despite mitigation efforts by NOAA Fisheries (Gobush, 2010).

Besides toxoplasmosis, infectious disease effects on monk seal demographic trends are low relative to other stressors. However, a disease outbreak could be catastrophic to the immunologically naive monk seal population. Key disease threats include West Nile virus, morbillivirus and influenza. NOAA Fisheries is vaccinating wild seals with the aim of reducing the impact of potential morbillivirus outbreaks.

Status of Stock

In 1976, the Hawaiian monk seal was designated as depleted under the MMPA and as endangered under the ESA (Musbach, et. al., 2007). Therefore, the Hawaiian monk seal is a strategic stock. The species is well below its optimum sustainable population and has not recovered from past declines. Annual human-caused mortality for the most recent 5-year period (2018–2022) was greater than 4.8 animals, including fishery-related mortality in nearshore gillnets, hook-and-line gear, and mariculture (≥2.2 / year, Table 4); intentional killings and other human-caused mortalities (≥0.8 / year, Table 3); entanglement in marine debris (≥0.4 / year, Table 5); and deaths due to toxoplasmosis (≥1.4 / year). The minimum rate of annual human-caused mortality was slightly below PBR (5.3).

Other factors that may be causing a decline or impeding recovery

Poor juvenile survival rates and variability in the relationship between weaning size and survival suggest that prey availability has limited recovery of NWHI monk seals (Baker & Thompson, 2007; Baker et al., 2007; Baker, 2008). Multiple strategies for improving juvenile survival, including translocation and rehabilitation are being implemented (Baker & Littnan, 2008; Baker et al., 2013; Norris, 2013).

A testament to the effectiveness of past actions to improve survival, Harting et al. (2014) demonstrated that approximately one-third of the monk seal population alive in 2012 was made up of seals that either had been intervened with to mitigate life-threatening situations or were descendants of such seals. In 2014, NOAA Fisheries produced a final Programmatic Environmental Impact Statement (PEIS) on current and future anticipated research and enhancement activities and issued a permit covering the activities described in the PEIS preferred alternative. Loss of terrestrial habitat at French Frigate Shoals is a serious threat to the viability of the resident monk seal population (Baker et al., 2020). Prior to 2018, pupping and resting islets had shrunk or virtually disappeared (Antonelis et al., 2006). In 2018, the two remaining primary islands where pups were born at French Frigate Shoals (Trig and East Islands) were obliterated due to progressive erosion and hurricane Walaka (in September 2018). Projected increases in global average sea level are expected to further significantly reduce terrestrial habitat for monk seals in the NWHI (Baker et al., 2006; Reynolds et al., 2012).

The seawall at Tern Island, French Frigate Shoals, continues to degrade and poses an entrapment hazard for monk seals and other fauna. The situation worsened after 2012, when the U.S. Fish and Wildlife Service ceased operations on Tern Island, leaving the island unmanned for most of the year. Previously, daily surveys were conducted throughout the year to remove entrapped animals. Now, this only occurs when NOAA Fisheries staff are on site. Furthermore, seawall breaches are allowing sections of the island to erode and undermine buildings and other infrastructure. Several large water tanks have collapsed, exposing pipes and wiring that may entangle or entrap seals. In September 2018, hurricane Walaka exacerbated this situation by largely destroying remaining structures and strewing the resulting debris around the island.

Strategies to mitigate these threats are currently under consideration. In 2020, the Papahānaumokuākea Marine Debris Project (PMDP), a non-profit organization, conducted an extensive cleanup operation at Tern Island, removing over 80,000 lb of debris and cutting multiple gaps in the seawall to provide escape routes for seals.

Goodman-Lowe (1998) provided information on prey selection using hard parts in scats and spewings. Information on at-sea movement and diving is available for seals at all six main subpopulations in the NWHI using satellite telemetry (Stewart et al., 2006). Cahoon (2011) and Cahoon et al. (2013) described diet and foraging behavior of main Hawaiian Islands monk seals and found no striking difference in prey selection between the NWHI and the main Hawaiian Islands.

Monk seal juvenile survival rates are favorable in the main Hawaiian Islands (Baker et al., 2011). Further, the excellent condition of pups weaned on these islands suggests that there are ample prey resources available, perhaps in part due to fishing pressure that has reduced monk seal competition with large fish predators (sharks and jacks) (Baker & Johanos, 2004). However, there are many challenges that may limit the potential for growth in this region.

The human population in the main Hawaiian Islands is approximately 1.4 million, compared to fewer than 100 in the NWHI; thus, anthropogenic threats in the main Hawaiian Islands are considerable. Intentional killing of seals is a very serious concern. Additionally, the same fishing pressure that may have reduced the monk seal's competitors is a source of injury and mortality. Vessel traffic in the populated islands entails risk of collision with seals and impacts from oil spills. A mortality in 2015 was deemed most likely due to boat strike. Finally, as noted above, toxoplasmosis is now recognized as a serious anthropogenic threat to seals in the main Hawaiian Islands.

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Northern Elephant Seal *Mirounga angustirostris*: California Breeding Stock

Stock Definition and Geographic Ranges

Northern elephant seals breed and give birth primarily on offshore islands (Stewart et al., 1994) in California (U.S.) and Baja California (Mexico) from December to March (Stewart and Huber, 1993). Spatial segregation in foraging areas between males and females is evident from satellite tag data (Le Boeuf et al., 2000). Males migrate to the Gulf of Alaska and western Aleutian Islands along the continental shelf to feed on benthic prey, while females migrate to the central North Pacific and the Gulf of Alaska to feed on pelagic prey (Le Boeuf et al., 2000; Figure 4). Adults return to land between March and August to molt; males return later than females. Adults return to their feeding areas again between their spring / summer molting and their winter breeding season.

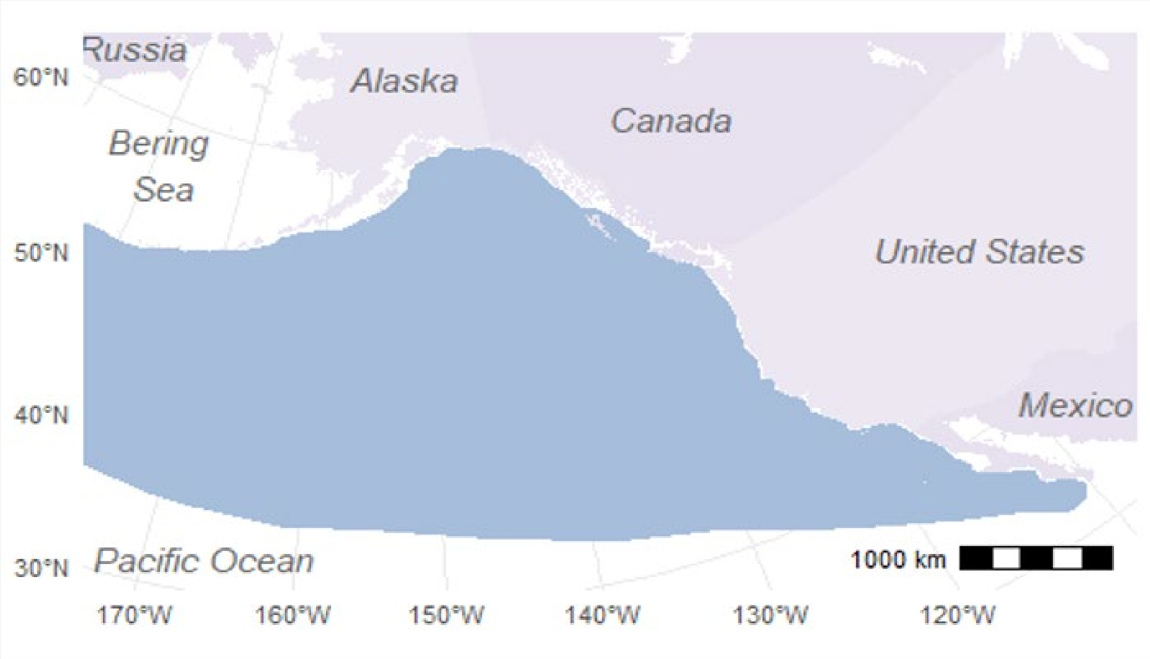

Figure 4. Approximate pelagic range of northern elephant seals including the eastern North Pacific Ocean. Major breeding rookeries occur along the west coast of Baja, California, and the California coast, as described in Lowry et al. (2014).

Populations of northern elephant seals in the U.S. and Mexico have recovered from near extinction following a severe population bottleneck after hunting reduced their numbers to an estimated 10 to 30 individuals, resulting in a substantial loss of genetic diversity (Hoelzel et al., 2002). Although movement and genetic exchange continues between rookeries, most elephant seals return to natal rookeries to breed as they reach reproductive maturity (Huber et al., 1991). The California breeding population is considered demographically distinct from the Baja California population. Therefore, the

California breeding population (subsequently referred to as the U.S. breeding population) is considered to be a separate stock. No international agreements exist for the joint management of this species between the U.S. and Mexico.

Population Size

A census of elephant seals is not possible since all age classes are not ashore simultaneously. The U.S. stock size has been historically estimated by counting the number of pups produced and multiplying by the inverse of the expected ratio of pups to total animals (McCann, 1985). More recently, pup births were estimated using aerial and / or ground counts of adult females present on the rookery during the breeding season (Le Boeuf et al., 2011; Lowry et al., 2014). The number of adult females was estimated based on rookery arrival dates and tenure (Condit et al., 2007). Total number of births were estimated by multiplying the estimated number of adult females by the reported fecundity rate (F=0.975; Le Boeuf et al., 2011) derived from the Año Nuevo rookery. The U.S. stock abundance estimate has been historically extrapolated from the estimated total births, which are multiplied by a correction factor based on the best available data at the time (Barlow et al., 1993; Boveng, 1998).

Starting in 2013, correction factors (c_{Pop}) were based on life table data (Lowry et al., 2020) constructed from elephant seal fecundity and survival rates, where approximately 23% of the population is comprised of pups (Cooper & Stewart, 1983; Hindell, 1991; Huber et al., 1991; Reiter & Le Boeuf, 1991; Clinton & Le Boeuf, 1993; Le Boeuf et al., 2019; Pistorius & Bester, 2002; McMahon et al., 2003; Pistorius et al., 2004; Condit et al., 2014). In years when ground counts are only completed at the Channel Islands (i.e., excludes un-surveyed areas in central and northern California), estimates of the total population are calculated as the sum of live and dead pups multiplied by the inverse of the U.S. population that resides at the Channel Islands (81.5%; Lowry et al., 2014).

In 2023, a range-wide survey was conducted during the elephant seal breeding season to count individuals of all age classes. Unlike previous population estimates, the estimate for 2023 implemented an updated model that derives true adult female attendance at rookeries ($N_{Adult\ Females}$ = 45,536) from counts of adult females during specific days of the year based on turnover of reproductive females throughout the breeding season (Condit et al., 2022). The number of births (N_{Births} = 44,398; 95% CL 42,876–46,308) was calculated using the estimated adult female population and mean fecundity rate (F=0.975). As with previous population estimates, the estimated number of births was multiplied by a correction factor (c_{Pop} = 4.39; 95% CL 3.87–4.92), assuming a population growth rate of 1.038 (Lowry et al., 2014) along the U.S. Pacific coast. Thus, total population size was estimated as $N_{Total} = c_{Pop} \times N_{Births}$, where $N_{births} = N_{Adult\ Females} \times F$, resulting in an estimate for 2023 of 194,907 (95% CI 170,185–233,677).

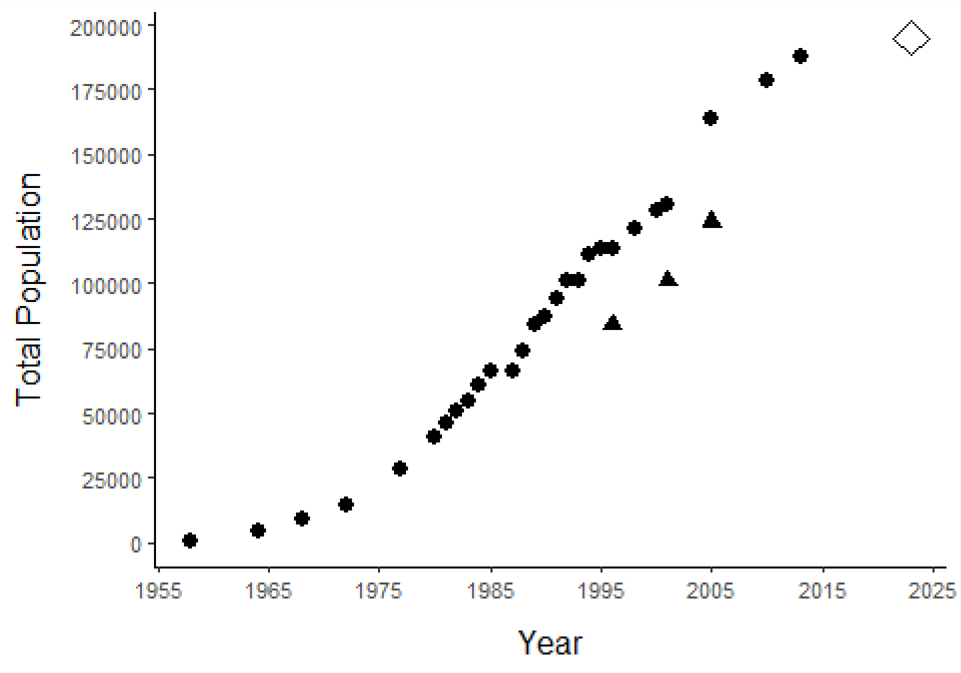

Figure 5. Estimated total U.S. stock abundance of northern elephant seals in California, 1958–2023. Circles represent estimates from a population growth rate of 17% (correction factor 4.4) from 1958–1987 and 3.8% (correction factor 4.39) from 1988–present. Pup birth estimates used to derive the total U.S. population estimate are from Stewart et al. (1994), Lowry et al. (1996), Lowry (2002), Lowry et al. (2014), and unpublished data from Sarah Allen, Dan Crocker, Brian Hatfield, Ron Jameson, Bernie Le Boeuf, Mark Lowry, Pat Morris, Guy Oliver, Derek Lee, and William Sydeman. Triangles represent estimates from previous stock assessment reports that used a correction of 3.5 (Barlow et al., 1993; Boveng, 1988). The open diamond represents the population estimate derived from an updated published model (Condit et al., 2022) to estimate the number of adult females and an assumed population growth rate of 3.8% (Lowry et al., 2014).

Minimum Population Estimate

The minimum population size for northern elephant seals in 2023 can be estimated conservatively as 88,794 seals, which is equal to twice the estimated pup count (to account for the pups and their mothers).

Current Population Trend

The California population is reported to have grown at 3.8% annually since 1988 (Lowry et al., 2014).

Current and Maximum Net Productivity Rate

An annual growth rate of 17% for elephant seals in the U.S. from 1958–1987 is reported by Lowry et al. (2014), but some of this growth is likely due to immigration of animals from Mexico and the consequences of a small population recovering from past exploitation. From 1988–2013, the population is estimated to have grown 3.8% annually (Lowry et al., 2014), which is assumed to have continued through 2023. For this stock assessment report, we use the default maximum theoretical net productivity rate for pinnipeds, or 12% (NMFS, 2023).

Potential Biological Removal

The PBR level for this stock is calculated as the minimum population size (88,794) multiplied by one half the observed maximum net growth rate for this stock (½ of 12%) **x** a recovery factor of 1.0 (for a stock of unknown status that is increasing (NMFS, 2023), resulting in a PBR of 5,328 animals per year.

Fisheries Information

A summary of known commercial fishery mortality and serious injury for this stock of northern elephant seals is given in Table 6. Total estimated commercial fishery mortality is ≥6.8 elephant seals annually. Although the mortality and serious injury shown in Table 6 occurred in U.S. waters, some may be of seals from Mexico's breeding population that are migrating through U.S. waters.

Other Mortality

Total mortality and serious injury from sources other than commercial fisheries for 2018–2022 includes the following: shootings (4); marine debris entanglement (2); hook and line fisheries (2); research-related (2), dog attack (2); unidentified human interaction (1); harassment (1); vehicle collision (1); tar / oil (6); and vessel strike (1) (Carretta et al., 2024). These other sources of mortality and serious injury total 22 animals, or an average of 4.4 elephant seals annually.

Table 6. Summary of available information on the mortality and serious injury of northern elephant seals (California breeding stock) in commercial fisheries that might take this species (Carretta, 2023; Carretta et al., 2024; Jannot et al., 2022; NMFS-MML, unpublished estimates based on methods in Breiwick, 2013,). N/a indicates information is not available. Mean annual takes are based on 2018-2022 data unless noted otherwise.

Fishery Name	Year(s)	Data Type	Percent Observer Coverage	Observed Mortality	Estimated Mortality (CV in parentheses)	Mean Annual Takes (CV in parentheses)
Alaska Bering Sea and Aleutian Islands Atka Mackerel Trawl	2018–2022	Observer	99%	2	2 (CV=0.085)	0.4 (CV=0.04)
Alaska Bering Sea and Aleutian Islands Flatfish Trawl	2018–2022	Observer	99%	1	1 (n/a)	0.2 (n/a)
CA thresher shark / swordfish drift gillnet fishery	2018–2022	Observer	20–25%	0	2.6 (>0.8)	0.52 (>0.8)
Dungeness Crab Pot Fishery (California)	2020	Stranding	n/a	1	1	0.2
Gillnet fishery, unidentified	2018–2022	Stranding	n/a	3	3	0.6
WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)	2015 2016 2017 2018 2019	Observer	98% to 100% of tows in at-sea hake fishery	n/a	9.2 (>0.8) 4.6 (>0.8) 3.7 (>0.8) 3.4 (>0.8) 3.5 (>0.8)	4.9 (>0.8)
Total annual takes						≥ 6.8 (CV>0.8)

Status of Stock

Northern elephant seals are not listed as "endangered" or "threatened" under the ESA nor designated as "depleted" under the MMPA. Total annual human-caused mortality (commercial fishery (6.8) + other human-caused sources (4.4 = 11.2) is less than the calculated PBR for this stock (5,328); thus, northern elephant seals are not considered a "strategic" stock under the MMPA.

The average rate of incidental commercial fishery related mortality for this stock over the last five years (11.2) is less than 10% of the calculated PBR (533); therefore, the total commercial fishery serious injury and mortality appears to be insignificant and approaching a zero mortality and serious injury rate. The population growth rate between 1958–1987 was 17% annually (Lowry et al., 2014). From 1988–2010, the population grew at an annual rate of 3.8% (Lowry et al., 2014). The population continues to grow, with most births occurring at southern California rookeries (Lowry et al., 2014).

No estimate of carrying capacity is available for this population, and the population status relative to optimal sustainable population (OSP) is unknown. There are no known habitat issues that are of concern for this stock. However, expanding pinniped populations in general have resulted in increased human-caused serious injury and mortality, due to shootings, entrainment in power plants, interactions with recreational hook and line fisheries, separation of mothers and pups due to human disturbance, dog bites, and vessel and vehicle strikes (Carretta et al., 2024).

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Northern Fur Seal *Callorhinus ursinus*: California Stock

Stock Definition and Geographic Range

Northern fur seals occur from southern California north to the Bering Sea and west to the Okhotsk Sea and Honshu Island, Japan (Figure 6). The worldwide population has been in a long-term decline since the 1950s (York & Hartley, 1981; Towell et al., 2006; Gelatt et al., 2015); however, it remains in excess of a million individuals. The global population has experienced inconsistent abundance trends at various breeding sites, with the most pronounced change occurring at the largest breeding colony, St. Paul Island, one of the Pribilof Islands, Alaska (Gelatt et al., 2015). During the breeding season, northern fur seals are primarily located at eight main colonies. Most of the worldwide population is found on the Pribilof (St. Paul and St. George) Islands, United States, the southern Bering Sea, and the Commander (Bering and Medny) Islands, Russia. Smaller rookeries are located at Tuleny (Robben) Island and the Kuril Islands in Russia, as well as Bogoslof Island, San Miguel Island, and Farallon Islands, United States (Gelatt et al., 2015). Northern fur seals may temporarily haul out on land at other sites in Alaska, British Columbia, and on islets along the coast of the continental United States, but generally this occurs outside of the breeding season (Fiscus, 1983).

Figure 6. Geographic distribution of the Northern Fur Seal.

Due to differing requirements during the annual reproductive season, adult males and females typically occur ashore at different, though overlapping, times. Adult males occur ashore and defend reproductive territories during a 3-month period from June–August, though some may be present until November (well after giving up their territories). Adult females are found ashore for as long as 6 months (June–November). After their respective times ashore, fur seals of both sexes spend the next 7–8 months at sea (Roppel, 1984). Adult females and pups from the Pribilof Islands migrate through the Aleutian Islands into the North Pacific Ocean, often to waters off Washington, Oregon, and California. Many pups may remain at sea for 22 months before returning to their natal rookery. Adult females and pups from San Miguel Island and the Farallon Islands migrate northward to these same areas (Lea et al., 2009). Adult males from the Pribilof Islands generally migrate only as far south as the Gulf of Alaska (Kajimura, 1984). Little is known about where adult males from San Miguel Island and the Farallon Islands migrate.

The following information was considered in classifying stock structure based on the Dizon et al. (1992) phylogeographic approach: (1) Distributional data: continuous geographic distribution during feeding, geographic separation during the breeding season, and high natal site fidelity (DeLong, 1982); (2) Population response data: substantial differences in population dynamics between the Pribilof and San Miguel Islands (DeLong, 1982; DeLong & Antonelis, 1991; NMFS, 2007); (3) Phenotypic data: unknown; and (4) Genotypic data: little evidence of genetic differentiation among breeding islands (Ream, 2002; Dickerson et al., 2010). Based on this information, two separate stocks of northern fur seals are recognized within U.S. waters: an Eastern Pacific stock (including the Pribilof and Bogoslof Islands) and a California stock (including San Miguel and the Farallon Islands). The Eastern Pacific stock is reported separately in the stock assessment reports for the U.S. Alaska Region.

This stock assessment report assesses the California stock of northern fur seals, which resides at their southernmost breeding rookeries in U.S. waters.

Population Size

The population estimate for northern fur seals on San Miguel Island (including Castle Rock, an islet 1 km from the main island) is calculated as the estimated number of pups at rookeries multiplied by an expansion factor. Based on research conducted on the Eastern Pacific stock of northern fur seals, Lander’s (1981) life table analysis was used to estimate the number of yearlings—two-year-old, three-year-old, and animals at least four-years-old. The resulting population estimate was equal to the pup count multiplied by 4.475.

The expansion factors were based on a sex and age distribution estimated after the commercial harvest of juvenile males was terminated in 1984. A more appropriate expansion factor for San Miguel Island is 4.0, because immigration of recruitment-aged females is occurring in the population (DeLong, 1982), as well as mortality and possible emigration of adults associated with the El Niño events in 1982–1983 and 1997–1998 (Melin et al., 2008). A 1998 pup count resulted in an 80% decrease from the 1997 count (Melin et al., 2005). In 1999, the population began to recover, and in 2017 the highest total pup count of 4,491 was recorded (Figure 6).

There could be multiple possible causes for the decline in total pup counts after 2017. No counts were conducted during 2020 due to travel restrictions resulting from the Coronavirus disease (COVID-19) pandemic. The abundance of territorial males decreased dramatically in 2015 and 2016, perhaps due to warm environmental conditions, such as the North Pacific marine heat wave of 2014–2015 (Cavole et al., 2015; Di Lorenzo & Mantua, 2016) and the El Niño of 2015–2016. The North Pacific marine heat wave (formerly termed “the blob”; Bond et al., 2015) consisted of a large area of abnormally high sea-surface temperature anomalies that started in the Gulf of Alaska in late 2013 (Bond et al., 2015). The North Pacific heat wave interacted with an El Niño in 2015, resulting in an abnormally long period of exceptionally high-temperature anomalies in the California Current system from 2014 to mid-2016 (McClatchie et al., 2016). The warm-water conditions that prevailed in 2015 and parts of 2016 may have adversely affected the foraging ecology and condition of territorial bulls during the non-breeding season and their subsequent return to San Miguel Island. As with adult male northern fur seals, adult females may not have returned San Miguel Island perhaps because they were nutritionally challenged to sustain their pregnancy. The decrease in reproductive animals may have led to subsequent decreases in recruited cohorts and decreases in pup production after 2017.

Another possible explanation may be due to habitat loss (erosion) on the island. During the past 5–10 years, the sandy beach where the northern fur seal rookery is primarily located has been subject to larger and more frequent storm generated waves. Subsequently, a large amount of substrate has been removed, resulting in flood ponds in the rookery ranging from a day to over a week. During these conditions, the pups might be thermally challenged, leading to death in some cases. Additionally, their carcasses might get swept out to sea and never accounted for during mortality surveys, resulting in lower production numbers. Based on the 2019 count (the most recent complete data set) and the expansion factor, the most recent population estimate of northern fur seals at San Miguel Island is 12,548 (3,137 × 4.0). Currently, a coefficient of variation (CV) for the expansion factor is unavailable; however, studies are underway to determine the accuracy and precision of the expansion factor.

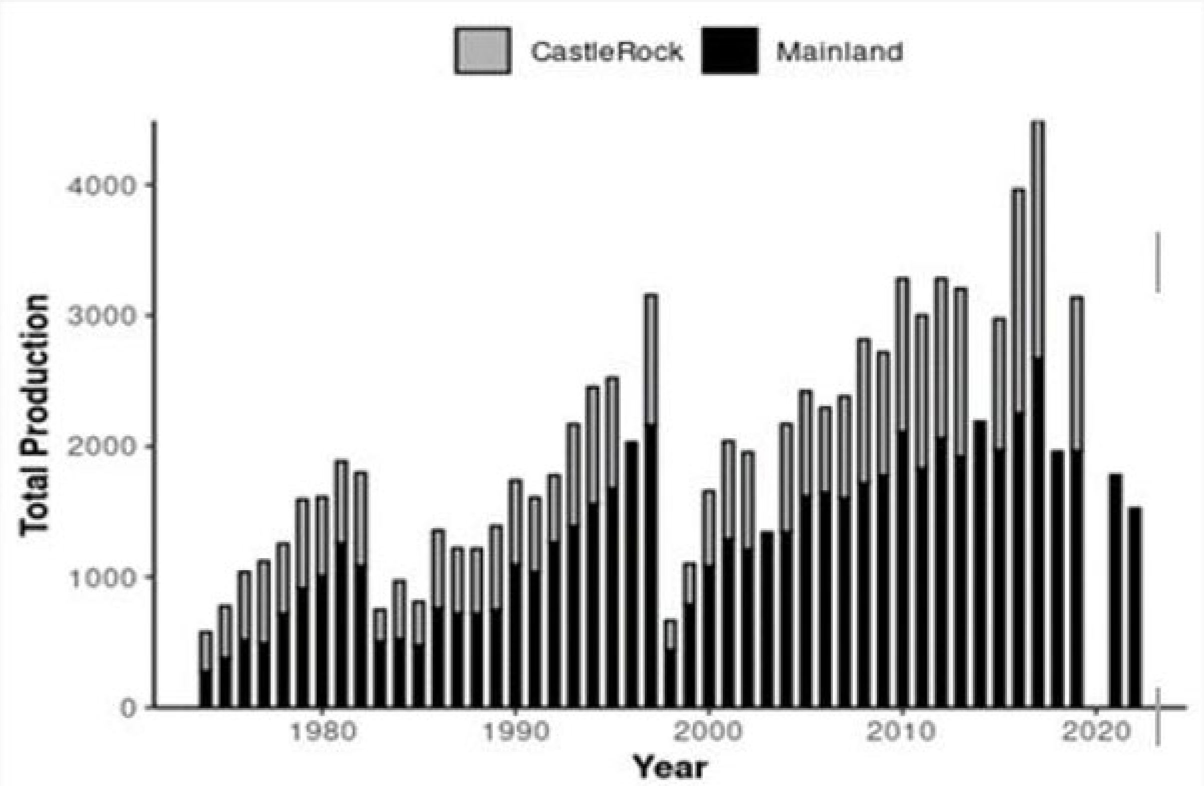

Figure 7. Estimated total production of northern fur seal pups counted on San Miguel Island, California, including the mainland rookeries (primarily Adam's Cove) and the offshore islet (Castle Rock), 1972–2022.

The population estimate for northern fur seals on the Farallon Islands (more specifically, the rookery on Southeast Farallon Island) is calculated as the highest number of pups, juveniles, and adults counted at the rookery. The long-term population estimate at the Farallon Islands should be regarded as an index of abundance rather than a precise indicator of population size for several reasons: (1) population censuses are incomplete because researchers do not enter rookery areas until the end of the breeding / pupping season to reduce human disturbance to other breeding pinnipeds and nesting seabirds, and counts are conducted from a lighthouse at the highest point of the island but part of the rookery area is obstructed from view; (2) mortality occurring early in the season is not accounted for; and (3) estimates of the number of pups are compromised because by the time counts are conducted, many pups have learned to swim and may not be present at the rookery. Additionally, yearlings may be present at rookeries and misidentified as pups.

Keeping these factors in mind, the peak counts of northern fur seals increased steadily from 1995 to 2006 and have increased exponentially from 2008–2022 (Figure 8; Tietz, 2012; Berger et al., 2013; Point Blue Conservation, unpubl. data). Based solely on the count, the population estimate of northern fur seals at the Farallon Islands was 1,464 in

2013 and increased to 7,086 in 2022. These estimates were based on applying a correction factor of 0.336 to the actual count. This number was derived from taking the annual maximums from ground surveys at West End Island and corrected using the mean correction factor estimated from aerial surveys for northern fur seal pups (Lee et al., 2018). This correction factor might not be applicable for non-pups, but it is the only one currently available. Efforts are being made to obtain more accurate counts at the Farallon Islands closer to the period of peak production.

The most recent population estimate for the entire stock of California northern fur seals, which incorporates estimates from San Miguel and the Farallon Islands in 2022, is 19,634 (12,548 + 7,086).

Minimum Population Estimate

Minimum population size is calculated as the sum of the minimum number of animals at San Miguel Island and the Farallon Islands in 2022 (NMFS unpubl. data, Point Blue Conservation, unpubl. data). The minimum number of animals at San Miguel Island is twice the pup count (3,137 × 2 = 6,274), to account for pups and mothers, plus the number of territorial males (133) counted the same year (i.e., 2022), or 6,407 fur seals. The minimum number at the Farallon Islands is the total number of individuals (2,381; raw count) counted during the survey in 2022. The total minimum population size is the sum of the minimum population sizes at San Miguel Island (6,407) and the Farallon Islands (2,381) in 2022, or 8,788 northern fur seals.

Current Population Trend

Northern fur seals were extirpated on San Miguel and the Farallon Islands during the late 1700s and early 1800s. Immigrants from the Pribilof Islands and Russian populations recolonized San Miguel Island during the late 1950s or early 1960s (DeLong, 1982). The colony has increased steadily, since its discovery in 1968, except for severe declines in 1983 and 1998 associated with El Niño events in 1982–1983 and 1997–1998 (DeLong & Antonelis, 1991; Melin et al., 2005). El Niño events impact population growth of northern fur seals at San Miguel Island and are an important regulatory mechanism for this population (DeLong & Antonelis, 1991; Melin & DeLong, 1994, 2000; Melin et al., 1996, 2005, 2008; Orr et al., 2012, 2016).

Live pup counts increased about 24% annually from 1972 through 1982 (Figure 8), partly due to immigration of females from the Bering Sea and the western North Pacific Ocean (DeLong, 1982). The 1982–1983 El Niño event resulted in a 60% decline in the northern fur seal population at San Miguel Island (DeLong & Antonelis, 1991). It took the population 7 years to recover from this decline, because adult female mortality or emigration occurred in addition to pup mortality (Melin & DeLong, 1994). The 1992–

1993 El Niño resulted in reduced pup production in 1992, but the population recovered in 1993 and increased during 1994 (Melin et al., 1996).

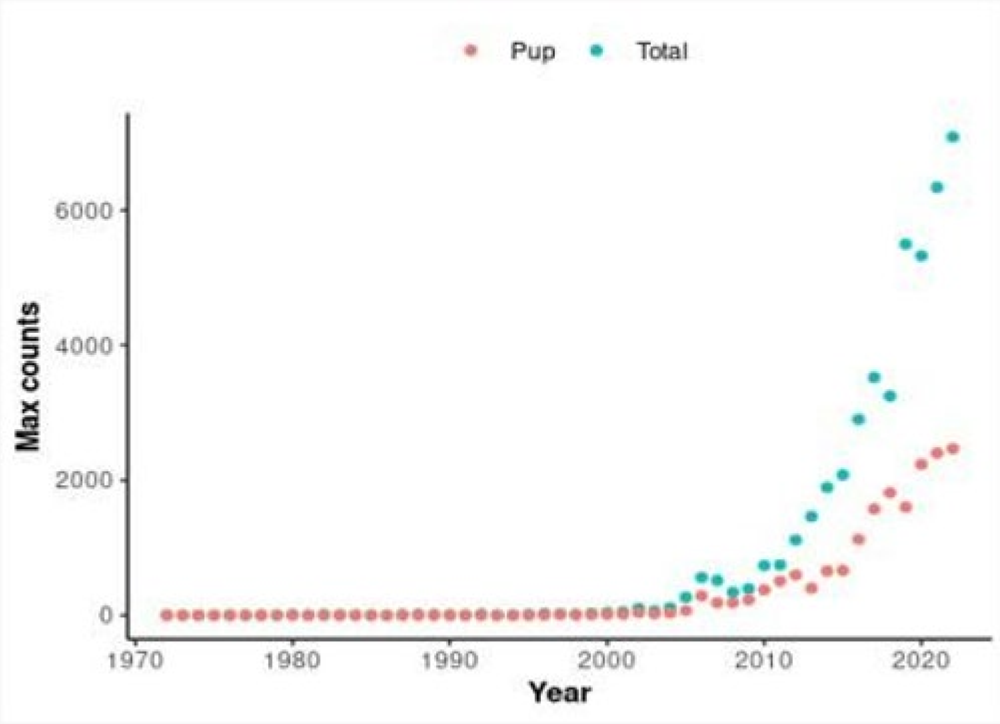

Figure 8. The peak counts of northern fur seals from 1970–2020.

The northern fur seal population appears to be greatly affected by El Niño events. These events cause changes in marine communities by altering sea-level height, sea-surface temperature, thermocline and nutricline depths, current-flow patterns, and upwelling strength. Fur seal prey generally move to more productive areas farther north and deeper in the water column and, thereby, become less accessible for fur seals. Consequently, fur seals at San Miguel Island are in poor physical condition during El Niño events and the population experiences reduced reproductive success and high mortality of pups and, occasionally, adults.

From July 1997 through May 1998, the most severe El Niño event in recorded history affected California coastal waters (Lynn et al., 1998). In 1997, total fur seal pup production was the highest recorded since the colony has been monitored. However, it appears that up to 87% of the pups born in 1997 died before weaning, and total production in 1998 declined 80% from 1997 (Melin et al., 2005). Total production increased to a record high of 4,491 in 2017. Since then, the population has decreased, not due to acute responses of the North Pacific marine heat wave or the El Niño in 2015–2016, but rather a delayed response to these warm-water anomalies resulting in a decrease in recruitment and pup production after 2017.

Additionally, recent pup production decreases may be attributed to increases in storm generated wave heights and frequency reaching northern fur seals on the rookery. Despite recent declines, pup production in years with complete counts still exceed the 1997 levels; therefore, the San Miguel Island population has recovered from the 1997–1998 El Niño event. Hookworm disease has decreased pup survival for the past 25 years and is also a major factor affecting the population dynamics of this species at its southernmost rookery (Lyons et al., 2001).

Compared to San Miguel Island, less information is known about the population of northern fur seals on the Farallon Islands. Based on tag-resight data, it appears that the population originated from emigrants from the Pribilof and San Miguel Islands. The first pup was observed on the Farallon Islands in 1996 (Pyle et al., 2001). After this discovery, annual ground surveys were conducted in early fall to document population trends of the colony (Tietz, 2012). The colony increased steadily from 1996 to the early 2000s. However, the population has grown exponentially during the past several years, with an occasional decline (Tietz, 2012). Because counts are conducted during the fall after the breeding season, population trends and demographic information are less clear than for San Miguel Island.

Current and Maximum Net Productivity Rates

Currently, productivity rates for northern fur seals on the Farallon Islands are unknown. A growth rate of 20% was calculated for northern fur seals on San Miguel Island in 1972–1982 by linear regression of the natural logarithm of pup count against year. However, it is clear that this rate of increase was due in part to immigration of females from Russian and Pribilof Islands populations (DeLong, 1982). Immigration was also occurring from the early 1980s to 1997. In the absence of a reliable estimate of the maximum net productivity rate for the California stock of northern fur seals, the pinniped default maximum theoretical net productivity rate (R_{MAX}) of 12% (Wade and Angliss 1997) is used as an estimate of R_{MAX}.

Potential Biological Removal

The PBR level for this stock is calculated as the minimum population estimate (8,788) times one-half the default maximum net growth rate (½ of 12%) times a recovery factor of 1.0 (for stocks of unknown status that are increasing in size: Wade & Angliss (1997), resulting in a PBR of 527 northern fur seals from the California stock per year.

Fisheries Information

Northern fur seals taken by commercial fisheries during the winter / spring along the U.S. West Coast could be from either the Eastern Pacific or California stock; therefore, any mortality or serious injury of northern fur seals reported off the coasts of California, Oregon, or Washington during December–May is assigned to both the Eastern Pacific and California stocks of northern fur seals. During 2018–2022, there were two deaths reported from trawl fisheries along the U.S. West Coast (Carretta et al., 2024). Jannot et al. (2022) report that two northern fur seals were observed killed or seriously injured in observed U.S. groundfish fisheries along the U.S. West Coast during the 18-year period from 2002–2019, with annual estimates of bycatch between zero and one animal per year. Updated bycatch estimates for U.S. West Coast groundfish fisheries are forthcoming. No northern fur seals have been observed entangled in the California large-mesh drift gillnet fishery for swordfish since observations began in 1990 (Carretta, 2023).

Table 7. Summary of available information on the incidental mortality and serious injury of the California stock of northern fur seals in commercial fisheries that might take this species and calculation of the mean annual mortality and serious injury rate. N/a indicates that data are not available. Mean annual takes are based on 2018–2022 data unless noted otherwise.

Fishery name	Years	Data type	Percent observer coverage	Observed mortality	Estimated mortality	Mean annual takes (CV in parentheses)
Rockfish midwater trawl	2018-2022	Observer	n/a	0, 0, 0, 1, 0	n/a	≥0.2
Unidentified trawl fishery	2018-2022	Stranding	n/a	0, 0, 0, 1, 0	n/a	≧0.2
Minimum total annual takes	-	-	-	-	-	≥0.4

Other Mortality

Stranding records for California, Oregon, and Washington (Carretta et al., 2024) include 5 non-fishery related injuries or deaths during 2018–2022. One seal was covered in oil and tar and was considered a serious injury due to its poor condition. Two dead seals showed evidence of blunt force trauma to their heads that was considered to be human-related. One seal died during research activities at a rookery. The mean annual mortality and serious injury rate from non-fishery sources in 2018–2022 includes three deaths and one serious injury, or 0.8 California northern fur seals annually. This estimate is considered a minimum because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel).

Status of Stock

The California northern fur seal stock is not considered to be “depleted” under the MMPA or listed as “threatened” or “endangered” under the ESA. Based on currently available data, the minimum annual level of total human-caused mortality and serious injury (1.2) does not exceed the PBR (527). Therefore, the California stock of northern fur seals is not classified as a “strategic” stock. The minimum annual commercial fishery mortality and serious injury rate for this stock (0.4) is not known to exceed 10% of the calculated PBR (53) and, therefore, appears to be insignificant and approaching zero mortality and serious injury rate. The stock (based on San Miguel Island data) decreased 80% from 1997 to 1998 but is considered to have recovered from that event. The status of this stock relative to its Optimum Sustainable Population (OSP) is unknown, unlike the Eastern Pacific northern fur seal stock which is formally listed as “depleted” under the MMPA.

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Killer Whale *Orcinus orca*: Eastern North Pacific Southern Resident Stock

Stock Definition and Geographic Ranges

Killer whales occur in all oceans and seas (Leatherwood & Dahlheim, 1978). Although they occur in tropical and offshore waters, killer whales prefer the colder waters of both hemispheres, with greatest abundances found within 800 km of major continents (Forney & Wade, 2006). Along the west coast of North America, killer whales occur along the entire Alaskan coast (Braham & Dahlheim, 1982), in British Columbia and Washington inland waterways (Bigg et al., 1990), and along the outer coasts of Washington, Oregon and California.

Seasonal and year-round occurrence is documented for killer whales throughout Alaska (Braham & Dahlheim, 1982) and in the intracoastal waterways of British Columbia and Washington, where three ecotypes have been recognized: "resident," "transient," and "offshore" (Bigg et al., 1990; Ford et al., 1994), based on aspects of morphology, ecology, genetics and behavior (Ford & Fisher, 1982; Baird & Stacey, 1988; Baird et al., 1992; Hoelzel et al., 1998; Morin et al., 2010; Ford et al., 2014). Recent genetic studies of killer whales globally suggested that residents and transient ecotypes warrant species recognition (Morin et al., 2024), and each were recently listed provisionally as named subspecies *Orcinus orca ater* (resident killer whale) and *O. orca rectipinnus* (Bigg's killer whale) by the Society for Marine Mammalogy's Taxonomy Committee (2024).

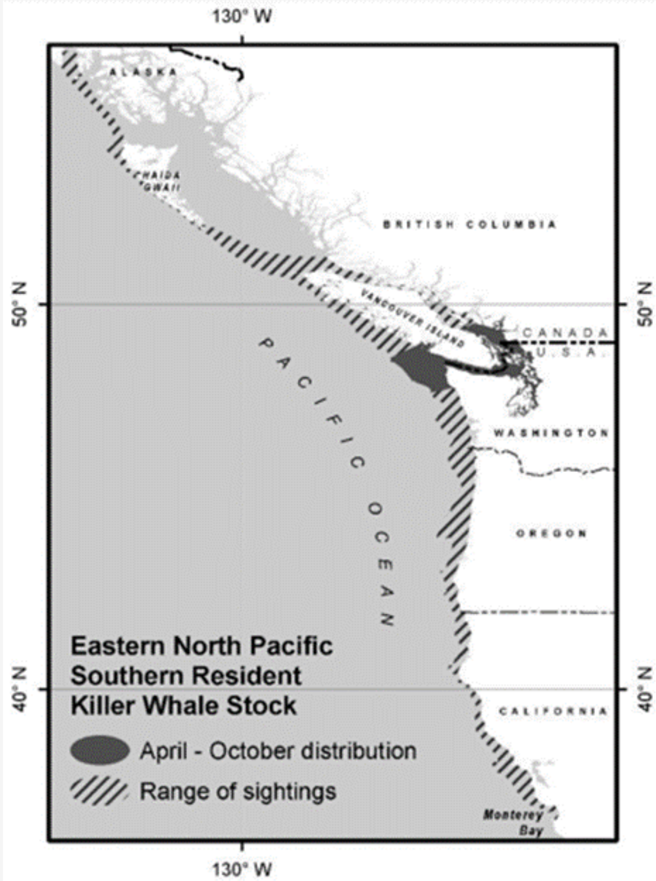

Figure 9. Approximate April–October distribution of the eastern North Pacific Southern Resident killer whale.

The range of Southern Resident killer whales is described in the biological report for the *Revision of the Critical Habitat Designation for Southern Resident Killer Whales* (NMFS, 2021a, 2021b):

> The three pods of the Southern Resident DPS, identified as J, K, and L pods, reside for part of the year in the inland waterways of Washington State and British Columbia known as the Salish Sea (Strait of Georgia, Strait of Juan de Fuca, and Puget Sound), principally during the late spring, summer, and fall (Ford et al., 2000: Krahn et al., 2004). The whales also occur in outer coastal waters, primarily in winter, off Washington and Vancouver Island, especially in the area between Grays Harbor and the Columbia River, and off Westport, WA (Ford et al., 2000; Hanson et al., 2017) but have been documented as far south as central California and as far north as Southeast Alaska. Although less is known about the whales' movements in outer coastal waters, satellite tagging, opportunistic sighting, and acoustic recording data suggest that Southern Residents spend nearly all of their time on the continental shelf, within 34 km (21.1 mi) of shore in water less than 200 m (656.2 ft) deep (Hanson et al., 2017).

Details of their winter range from satellite-tagging reveal whales use the entire Salish Sea (northern end of the Strait of Georgia and Puget Sound) in addition to coastal waters from the central west coast of Vancouver Island, British Columbia to Pt. Reyes in northern California. Animals from J pod were documented moving between the northern Strait of Georgia and the western entrance of the Strait of Juan de Fuca, with limited movement into coastal waters. In contrast, K and L pod movements were characterized by a coastal distribution from the western entrance to the Strait of Juan de Fuca to Pt. Reyes California (Hanson et al., 2017). Of the three pods comprising this stock, one (J) is commonly sighted in inshore waters in winter, while the other two (K and L) apparently spend more time offshore (Ford et al., 2000). Krahn et al. (2009) described sample pollutant ratios from K and L pod whales that were consistent with a hypothesis of time spent foraging in California waters, which is consistent with sightings of K and L pods as far south as Monterey Bay.

In June 2007, whales from L-pod were sighted off Chatham Strait, Alaska, the farthest north they have ever been documented (Hilborn et al., 2012). Southern Resident killer whale attendance in their core summer habitat in the Salish Sea appears to be declining, with occurrence well-below average since 2017 (Center for Whale Research, 2019) and shifting later in more recent years (Ettinger et al., 2022). Passive autonomous acoustic recorders have provided more information on the seasonal occurrence of these pods along the west coast of the U.S. (Hanson et al., 2013). In addition, satellite-linked tags were deployed in winter months on members of J, K, and L pods. Results were consistent with previous data, but provided much greater detail, showing wide-ranging use of inland waters by J pod whales and extensive movements in U.S. coastal waters by K and L pods.

Based on data regarding association patterns, acoustics, movements, genetic differences, and potential fishery interactions, eight killer whale stocks are recognized within the Pacific U.S. EEZ: (1) the eastern North Pacific Alaska Resident stock—occurring from Southeast Alaska to the Bering Sea, (2) the eastern North Pacific Northern Resident stock—occurring from British Columbia through Alaska, (3) the eastern North Pacific Southern Resident stock—occurring mainly within the inland waters of Washington State and southern British Columbia but extending from central California into southern Southeast Alaska (Figure 9), (4) the West Coast Transient stock—occurring from Alaska through California, (5) the Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock—occurring from southeast Alaska to the Bering Sea, (6) the AT1 Stock—found only in Prince William Sound, (7) the eastern North Pacific Offshore stock—occurring from Southeast Alaska through California, (8) the Hawaiian stock. The stock assessment reports for the Alaska Region contain information concerning the eastern North Pacific Alaska Resident, eastern North Pacific

Northern Resident and the Gulf of Alaska, Aleutian Islands, and Bering Sea, AT1, and eastern North Pacific transient stocks.

Population Size

The eastern North Pacific Southern Resident stock is a trans-boundary stock including killer whales in inland Washington and southern British Columbia waters. In 1993, the three pods comprising this stock totaled 96 killer whales (Ford et al., 1994). The population increased to 99 whales in 1995, then declined to 79 whales in 2001, and most recently numbered 73 whales in 2024 (Figure 10; Ford et al., 2000; Center for Whale Research, 2024). The most recent census, spanning July 1, 2023 through July 1, 2024, includes the deaths of two adult males (K34, L85). Since completion of the census, three additional calves were born (L128, J61, J62), but two have subsequently died, and one adult male is presumed dead (K26).

Minimum Population Estimate

The abundance estimate for this stock of killer whales is a direct count of individually identifiable animals. It is thought that the entire population is censused every year. This estimate therefore serves as both a best estimate of abundance and a minimum estimate of abundance. Thus, the minimum population estimate (N_{min}) for the eastern North Pacific Southern Resident stock of killer whales is 73 animals.

Figure 10. Line graph indicating the Southern Resident Killer Whale population census from 1993 to 2024, with the population declining from a peak of 99 in 1995 to 73 in 2024.

Current Population Trend

During the live-capture fishery that existed from 1967–1973, it is estimated that 47 killer whales, mostly immature, were taken out of this stock (Ford et al., 1994). Since the first complete census of this stock in 1974, when 71 animals were identified, the number of southern resident killer whales has fluctuated. Between 1974 and the mid-1990s, the Southern Resident stock increased approximately 35% (Ford et al., 1994), representing a net annual growth rate of 1.8% during those years. Following the peak census count of 99 animals in 1995, the population size has declined approximately 1% annually and currently stands at 73 animals as of the 2024 census (Ford et al., 2000; Center for Whale Research, 2024). Recent population models, based on an analysis of the entire SRKW genome, suggest that inbreeding depression is limiting population growth, and predicts further decline if the population remains genetically isolated and typical environmental conditions continue (Kardos et al., 2023). Recent work by Williams et al. (2024) indicates that reducing some of the threats faced by this population that were included in their model (Chinook availability, noise, polychlorinated biphenyls (PCB) accumulation), could allow population growth, but the baseline population dynamics model predicted gradual reduction followed by an accelerating decline.

Current and Maximum Net Productivity Rates

A reliable estimate of the maximum net productivity rate is currently unavailable for this stock of killer whales. Matkin et al. (2014) estimated a maximum population annual growth rate of 1.035 for southern Alaska resident killer whales. The authors noted that the 3.5% annual rate estimated for southern Alaska residents is higher than previously measured rates for British Columbia northern residents (2.9%) and “probably represents a population at r-max (maximum rate of growth)” (Olesiuk et al., 1990).

In the absence of published estimates of R_{max} for Southern Resident killer whales, the maximum annual rate of 3.5% found for Southern Alaska residents is used for this stock of Southern Resident killer whales. This reflects more information about the known life history of resident killer whales than the default R_{max} of 4% and results in a more conservative estimate of PBR.

Potential Biological Removal

The PBR level for this stock is calculated as the minimum population size (73) x ½ the maximum net growth rate for *Alaska* resident killer whales (½ of 3.5%) x a recovery factor of 0.1 (for an endangered stock; Wade and Angliss, 1997), resulting in a PBR of 0.13 whales per year, or approximately 1 animal every 7 years.

Fisheries Information

The only known case of Southern Resident killer whale mortality due to fisheries is an adult male, L8, who entangled in gillnet fishing gear and drowned in 1977 (Center for Whale Research, 2015). The entanglement occurred near southeastern Vancouver Island (Ford et al., 1998), and upon necropsy two pounds of recreational fishing lures and lines were found in the stomach. It was noted that some of the fishing gear found did not appear to be used locally at the time, and the ingestion of the gear did not cause the death of the animal. Salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994, and no killer whale entanglements were documented, though observer coverage levels were less than 10% (Erstad et al., 1996; Pierce et al., 1994; Pierce et al., 1996; NWIFC, 1995).

Fishing effort in the inland waters drift gillnet fishery has declined considerably since 1994 because far fewer vessels participate today. Past marine mammal entanglements in this fishery included harbor porpoise, Dall’s porpoise, and harbor seals. Coastal marine tribal set gillnets also occur along the outer Washington coast and no killer whale interactions have been reported in this fishery since the inception of the observer program in 1988, though the fishery is not active every year (Gearin et al., 1994; Gearin et al., 2000; Makah Fisheries Management). No fishery-related mortality from gillnet fisheries in California waters was documented between 2017–2021 (Carretta, 2022; Carretta et al., 2023).

An additional source of information on killer whale mortality and injury incidental to commercial fishery operations is the self-reported fisheries information required of vessel operators by the MMPA. No self-report records of killer whale mortality have been reported.

In 2015, J39, a young male Southern Resident killer whale, was found near False Bay, Washington with a recreational salmon flasher dangling from its mouth (Center for Whale Research, 2015). The whale was seen five days later without the gear attached and appeared energetic. The whale was monitored over the following weeks and there was no evidence of injury or behavioral changes (Center for Whale Research, 2015).

Due to a lack of observer programs, there are few data concerning the mortality of marine mammals incidental to Canadian commercial fisheries. Since 1990, there have been no reported fishery-related strandings of killer whales in Canadian waters. However, in 1994, one killer whale was reported to have contacted a salmon gillnet but did not entangle (Guenther et al., 1995). In 2014, a northern resident killer whale became entangled in a gillnet, and was released from the net, but died the next winter (Fisheries and Oceans Canada, 2018). Data regarding the level of killer whale mortality related to commercial fisheries in Canadian waters are not available.

The known total fishery mortality and serious injury for the Southern Resident stock of killer whales is zero, but undetected mortality and serious injury may occur.

Other Mortality

In 2012, a moderately decomposed juvenile female Southern Resident killer whale (L-112) was found dead near Long Beach, Washington. A full necropsy was performed, and the cause of death was determined to be blunt force trauma to the head; however, the source of the trauma (vessel strike, intraspecific aggression, or other unknown source) could not be established (NWFSC, 2014). There was documentation of a whale-boat collision in Haro Strait in 2005, which resulted in a minor injury to a whale.

In 2006, whale L98 was killed during a vessel interaction. It is important to note that L98 had become habituated to regularly interacting with vessels during its isolation in Nootka Sound.

In spring 2016, a young adult male, L95, was found to have died of a fungal infection related to a satellite tag deployment approximately 5 weeks prior to its death. The expert panel reviewing the stranding noted that “the tag loss, tag petal retention with biofilm formation or direct pathogen implantation, and development of a fungal infection at the tag site contributed to the illness, stranding, and death of this whale." (NMFS, 2016).

In fall 2016, another young adult male, J34, was found dead in the northern Georgia Strait. The necropsy indicated that "the animal had injuries consistent with blunt trauma to the dorsal side, and a hematoma indicating that it was alive at the time of injury and would have survived the initial trauma for a period of time prior to death" (Fisheries and Oceans Canada, 2019). The injuries are consistent with those incurred during a vessel strike. A recent summary of killer whale strandings in the northeastern Pacific Ocean and Hawai‘i noted the occurrence of human interactions across all age classes (Raverty et al., 2020).

Status of Stock

Total documented annual fishery mortality and serious injury for this stock from 2018–2022 (zero) is not known to exceed 10% of the calculated PBR (0.13). Given the low PBR level, a single undetected / undocumented fishery mortality or serious injury would exceed 10% of the PBR; thus, it is unknown if fishery mortality and serious injury is approaching zero mortality and serious injury rate. The documented annual level of human-caused mortality and serious injury for the most-recent 5-year period of 2018–2022 is zero. Southern Resident killer whales were formally listed as "endangered" under the ESA in 2005 and, consequently, the stock is automatically considered as a "strategic" stock under the MMPA. This stock was considered "depleted" (68 FR 31980, May 29, 2003) prior to its 2005 listing under the ESA (70 FR 69903, November 18, 2005).

Other factors that may be causing a decline or impeding recovery

A population viability analysis identified several risk factors to this population, including limitation of preferred Chinook salmon prey, anthropogenic noise and disturbance resulting in decreased foraging efficiency, and high levels of contaminants, including PCBs and dichlorodiphenyltrichloroethane (DDT) (Ebre, 2002; Clark et al., 2009; Krahn et al., 2007, 2009; Lacy et al., 2017; Nelson et al., 2024; Williams et al., 2024). The summer range of this population, the inland waters of Washington and British Columbia, are home to a large commercial whale watch industry, and high levels of recreational boating and commercial shipping. Potential for acoustic masking effects on the whales' communication and foraging due to vessel traffic remains a concern (Erbe 2002, Clark et al., 2009; Lacy et al., 2017; Holt et al., 2021a, 2021b).

In 2011, federal vessel approach regulations were implemented to restrict vessels from approaching closer than 200 m. Current regulations in Washington State restrict vessel approaches to 300 yards to the side or 400 yards in the path and will be expanded to

1000 yards in 2025. A genetic study of diet of Southern Resident killer whales from fecal remains collected during 2006–2011 noted that salmonids accounted for >98.6% of genetic sequences (Ford et al., 2016). Of six salmonid species documented, Chinook salmon accounted for 79.5% of the sequences, followed by coho salmon (15%). Chinook salmon dominate the diet in early summer, with coho salmon averaging >40% of the diet in late summer. Sockeye salmon were also found to be occasionally important (>18% in some samples). Non-salmonids were rarely observed.

These results are consistent with those obtained from surface prey remains and confirm the importance of Chinook salmon in this population's diet. These authors also noted the absence of pink salmon in the fecal samples. Prior studies note the prevalence of Chinook salmon in the killer whale diet, despite the relatively low abundance of this species in the region, supporting the thesis that Southern Resident killer whales are Chinook salmon specialists (Ford & Ellis, 2006; Hanson et al., 2010). Recent studies of diet in other seasons and regions of their range indicate that, although Chinook represent a major component of their diet almost year-round, other species also make potentially important contributions, likely when Chinook salmon are less available (Hanson et al., 2021). There is evidence that reduced abundance of Chinook salmon has negatively affected this population via reduced fecundity (Ayres et al., 2012; Ford et al., 2009; Ward et al., 2009; Wasser et al., 2017).

Studies on body condition and sizes of Southern Resident killer whales using aerial photogrammetry (Fearnbach et al., 2011; Fearnbach et al., 2020; Stewart et al., 2021) reflect hypotheses between Chinook salmon abundance and killer whale body condition and overall body size. In some cases (J pod), Chinook salmon abundance was found to have the greatest predictive power on Southern Resident body condition, while this relationship was absent for K pod (Stewart et al., 2021). In other studies (Fearnbach et al., 2011), authors suggest that nutritional stress is linked to a longer term decrease in body size in the population. In addition, the high trophic level and longevity of the population has predisposed them to accumulate high levels of contaminants that potentially impact health (Krahn et al., 2007, 2009). In particular, there is evidence of high levels of flame retardants in young animals (Krahn et al., 2007, 2009). High DDT / PCB ratios have been found in Southern Resident killer whales, especially in K and L pods (Krahn et al., 2007; NMFS, 2021b), which spend more time in California waters where DDTs still persist in the marine ecosystem (Sericano et al., 2014).

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