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# U.S. PACIFIC MARINE MAMMAL STOCK ASSESSMENTS: 2023

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Reports revised in 2023 are highlighted; all others can be found at the NOAA marine mammal stock assessment homepage and previously published reports are also contained in this document.

**PINNIPEDS**



**CETACEANS - U.S. WEST COAST**



**CETACEANS – HAWAI'I & WESTERN PACIFIC**

APPENDICES

# PREFACE

Under the 1994 amendments to the Marine Mammal Protection Act (MMPA), the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) are required to publish Stock Assessment Reports for all stocks of marine mammals within U.S. waters, to review new information every year for strategic stocks and every three years for non-strategic stocks, and to update the stock assessment reports when significant new information becomes available.

Pacific region stock assessments include those studied by the Southwest Fisheries Science Center (SWFSC, La Jolla, CA), the Pacific Islands Fisheries Science Center (PIFSC, Honolulu, HI), the Marine Mammal Laboratory (NMML, Seattle, WA), and the Northwest Fisheries Science Center (NWFSC, Seattle, WA). The 2023 Pacific marine mammal stock assessments include revised reports for 27 stocks under NMFS jurisdiction, including 8 "strategic" stocks: Hawaiian monk seal, Southern Resident killer whale, CA-OR-WA Sperm Whale, Eastern North Pacific blue whale, CA-OR-WA fin whale, Eastern North Pacific sei whale, Hawaiʻi pelagic false killer whale, and Hawaiian Islands Insular false killer whale. Information on sea otters, manatees, walrus, and polar bears are published separately by the US Fish and Wildlife Service.

This is a working document and individual stock assessment reports will be updated as new information on marine mammal stocks and fisheries becomes available. Background information and guidelines for preparing stock assessment reports are reviewed in NOAA (2023). The authors solicit any new information or comments which would improve future stock assessment reports. Draft versions of the 2023 stock assessment reports were reviewed by the Pacific Scientific Review Group (PSRG) at the March 2023 meeting. These Stock Assessment Reports summarize information from a wide range of original data sources and an extensive bibliography of published sources are provided in each report. We recommend users of this document refer to and *cite original literature* sources cited within the stock assessment reports rather than citing this report or previous Stock Assessment Reports.

References:

NMFS. 2023. Guidelines for Preparing Stock Assessment Reports Pursuant to the Marine Mammal Protection Act. Protected Resources Policy Directive 02-204-01.

Revised 3/18/2019

# CALIFORNIA SEA LION (*Zalophus californianus*): U.S. Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

California sea lions breed on islands located in southern California, western Baja California, and the Gulf of California (Fig. 1). Mitochondrial DNA analysis identified five genetically distinct geographic populations: (1) Pacific Temperate, (2) Pacific Subtropical, (3) Southern Gulf of California, (4) Central Gulf of California and (5) Northern Gulf of California (Schramm *et al*. 2009). The Pacific Temperate population includes rookeries within U.S. waters and the Coronados Islands just south of U.S./Mexico border. Animals from the Pacific Temperate population range into Canadian and Baja California waters. Males from western Baja California rookeries may spend most of the year in the United States.

International agreements between the U.S., Mexico, and Canada for joint management of California sea lions do not exist, and sea lion numbers at the Coronado Islands is not monitored. Consequently, this report considers only the U.S. Stock, i.e. sea lions at rookeries north of the U.S./Mexico border. Pup production at the Coronado Islands is minimal (between 12 and 82 pups annually; Lowry and Maravilla-Chavez 2005) and does not represent a significant contribution to the overall size of the Pacific Temperate population.

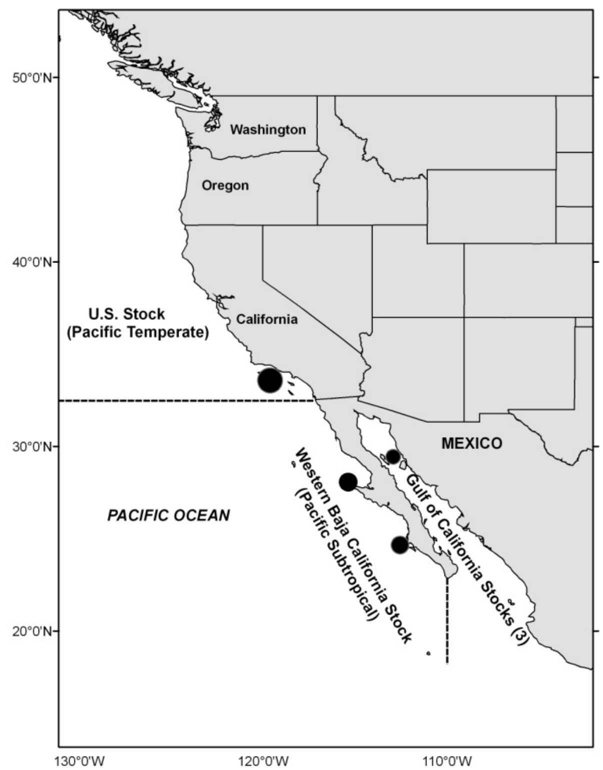


**Figure 1.** Geographic range of California sea lions showing stock boundaries and locations of major rookeries. The U.S. stock also ranges north into Canadian waters.

## POPULATION SIZE

California sea lion population size was estimated from a 1975-2014 time series of pup counts (Lowry *et al*. 2017), combined with mark-recapture estimates of survival rates (DeLong *et al*. 2017, Laake *et al*. 2018). Population size in 2014 was estimated at 257,606 animals, which corresponded with a pup count of 47,691 animals along the U.S. west coast (Lowry *et al*. 2017, Laake *et al*. 2018).

### Minimum Population Estimate

Minimum population size for 2014 is taken as the lower 95% confidence interval (CI) of the 2014 population size estimate, or 233,515 animals (Laake *et al.* 2018). The lower 95% CI is used as an estimate of $N_{min}$ in this report because the lower 20$^{th}$ percentile of the estimated population size is not calculated in Laake *et al*. 2018. The lower 95% CI is a more conservative estimate of minimum population size and is superior to previous approaches that simply used 2x the annual pup count, which were negatively-biased because not all age classes were represented.

### Current Population Trend

Population size trends from 1975 through 2014 are shown in Fig. 2. The time series of population estimates are derived from 3 primary data sources: 1) annual pup counts (Lowry *et al*. 2017); 2) annual survivorship estimates from mark-recapture data (DeLong *et al*. 2017); and 3) estimates of human-caused serious injuries, mortalities, and bycatch (Carretta and Enriquez 2012a, 2012b, Carretta *et al*. 2016, Carretta *et al*. 2018a, 2018b). These 3 data sources were combined to reconstruct the population size estimates shown in Fig. 2 (Laake *et al*. 2018). Age- and sex-specific survival rates of California sea lions were estimated by DeLong *et al*. (2017), and female survivorship exceeds that of males. Annual pup survival was 0.600 and 0.574 for females and males, respectively. Maximum annual survival rates corresponded to animals 5 years of age (0.952 and 0.931 for females and males, respectively). Survival of pups and yearlings declined with increasing sea surface temperatures (SST). For each 1 degree C increase in SST, the estimated odds of survival declined by 50% for pups and yearlings, while negative SST anomalies resulted in higher survival estimates (DeLong *et al.* 2017). Such declines in survival are related to warm oceanographic conditions (e.g. El Niño) that limit prey availability to pregnant and lactating females (DeLong *et al*. 2017).

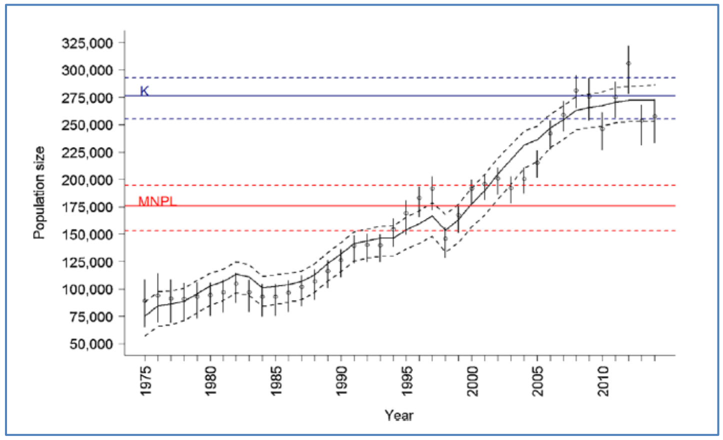


**Figure 2.** Fitted logistic growth curve (solid line) and 95% bootstrap intervals (dashed line) for reconstructed California sea lion annual population sizes in the United States, 1975–2014. Vertical lines indicate 95% confidence intervals for reconstructed annual population sizes. We also present estimated carrying capacity (K; solid blue line) with 95% confidence intervals (dashed blue line) and maximum net productivity level (MNPL; red solid line) with 95% confidence intervals (dashed red line). Figure from Laake *et al*. 2018.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Using a logistic growth model and reconstructed population size estimates from 1975-2014, Laake *et al*. (2018) estimated a net productivity rate of 7% per year. This estimate includes periods of sharp population declines associated with El Niño events and excludes undocumented levels of anthropogenic removals through bycatch and other sources (Carretta *et al*. 2016). The net productivity rate estimate of 7% per year is not considered a maximum net productivity rate, and Laake *et al*. (2018) note that the population is capable of faster growth rates. Therefore, we use the default maximum net productivity rate for pinnipeds of 12% per year (Wade and Angliss 1997). Laake *et al*. (2018) also estimated the population size at maximum net productivity level (MNPL) to be 183,481 animals.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (233,515) <u>times</u> one half the default maximum net growth rate for pinnipeds (½ of 12%) <u>times</u> a recovery factor of 1.0 (for a stock within OSP, Laake *et al*. 2018, Wade and Angliss 1997); or 14,011 sea lions per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Historical Depletion

Historic exploitation of California sea lions include harvest for food by native Californians in the Channel Islands 4,000-5,000 years ago (Stewart *et al*. 1993) and for oil and hides in the mid-1800s (Scammon 1874). Other exploitation of sea lions for pet food, target practice, bounty, trimmings, hides, reduction of fishery depredation, and sport are reviewed in Helling (1984), Cass (1985), Seagers *et al*. (1985), and Howorth (1993). There are few historical records to document the effects of such exploitation on sea lion abundance (Lowry *et al*. 1992).

## Fisheries Information

California sea lions are killed in a variety of trawl, purse seine, and gillnet fisheries along the U.S. west coast (Barlow *et al*. 1994, Carretta and Barlow 2011, Carretta *et al*. 2018a, 2018b, Julian and Beeson 1998, Jannot *et al*. 2011, Stewart and Yochem 1987). Sources with recent observations or estimates of bycatch mortality are summarized in Table 1. In addition to bycatch estimates from fishery observer programs, data on fishery-related sea lion deaths and serious injuries comes largely from stranding data (Carretta *et al*. 2018b). Stranding data represent a minimum number of animals killed or injured, as many entanglements are unreported or undetected.

California sea lions are also killed and injured by hooks from recreational and commercial fisheries. Sea lion deaths due to hook-and-line fisheries can result from complications involving hook ingestion, perforation of body cavities leading to infections, or the inability of the animal to feed. Many animals die post-stranding during rehabilitation or are euthanized as a result of their injuries. Between 2012 and 2016, there were 146 California sea lion deaths / serious injuries attributed to hook and line fisheries, or an annual average of 29 animals (Carretta *et al*. 2018b).

**Table 1.** Summary of available information on the mortality and serious injury of California sea lions in commercial fisheries that might take this species (Carretta *et al.* 2012a, 2012b, 2014a, 2018a, 2018b;). Mean annual takes are based on 2012-2016 data unless noted otherwise. Bycatch estimates for 2 additional years, 2010 and 2011, have been included for the CA halibut and white seabass set gillnet fishery because this fishery has not been observed recently or lacks estimates of bycatch.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish large mesh drift gillnet fishery** | 2012<br>2013<br>2014<br>2015<br>2016<br>2012-2016 | observer | 19%<br>37%<br>24%<br>20%<br>18%<br>23% | 6<br>3<br>3<br>0<br>0<br>12 | 16.1 (0.58)<br>11.6 (0.35)<br>10.9 (0.59)<br>6.2 (0.92)<br>17 (0.67)<br>62.3 (0.24) | 12.5 (0.24) |
| **CA halibut and white seabass set gillnet fishery** | 2010<br>2011<br>2012<br>2013<br>2014<br>2015<br>2016 | observer | 12.5%<br>8.0%<br>5.5%<br>n/a<br>0%<br>0%<br>0% | 25<br>6<br>18<br>0<br>n/a<br>n/a<br>n/a | 199 (0.30)<br>74 (0.39)<br>326 (0.33)<br>0 (n/a)<br>n/a<br>n/a<br>n/a | 150 (0.28) |
| **CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna** | 2010<br>2011<br>2012 | observer | 0.7%<br>3.3%<br>4.6% | 0<br>0<br>0 | 0 (n/a)<br>0 (n/a)<br>0 (n/a) | 0 (n/a) |
| **CA anchovy, mackerel, sardine, and tuna purse- seine fishery** | 2004-2008 | observer | ~5% | 2 | n/a | ≥2 (n/a) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)** | 2012-2016 | observer | 98% to 100% of tows in at-sea hake fishery<br><br>Generally less than 30% of landings observed in other groundfish sectors | 95 | n/a | ≥ 19 (n/a) |
| **Unknown entangling net fishery** | 2012-2016 | stranding | n/a | 55 | n/a | ≥ 11 (n/a) |
| **Unidentified fishery interactions** | 2012-2016 | stranding | n/a | 11 | n/a | ≥ 2.2 |
| **Minimum total annual takes** | | | | | | ≥ 197 (0.23) |

**Other Mortality**

California sea lions strand with evidence of human-caused mortality and serious injury from a variety of non-commercial fishery sources, including shootings, hook and line fisheries, power plant entrainment, marine debris entanglement, oil exposure, vessel strikes, and dog attacks (Carretta *et al*. 2018b). Between 2012 and 2016, there were 485 mortality and serious injuries documented from these sources, or an annual average of 97 sea lions (Carretta *et al*. 2018b). The most common sources of mortality and serious injury were shootings (n=155), hook and line fisheries (n=146), entanglements in marine debris (n=65), and oil exposure (n=58), which accounted for 87% of all cases. These values represent a minimum accounting of impacts, because an unknown number of dead or injured animals are unreported or undetected.

Under authorization of MMPA Section 120, individually identifiable California sea lions have been euthanized or relocated since 2008 in response to their predation on endangered salmon and steelhead stocks in the Columbia River. Relocated animals are transferred to aquaria and/or zoos. Between 2012 and 2016, 122 California sea lions were removed from this stock (115 lethal removals and 7 relocations to aquaria and/or zoos). The average annual mortality due to direct removals for the 2012-2016 period is 24.4 animals per year (Carretta *et al*. 2018b). Relocations to aquaria/zoos are treated equivalent to mortality because animals are effectively removed from the environment.

Mortality and serious injury may occasionally occur incidental to marine mammal research activities authorized under NMFS protected species permits issued to government, academic, and other research organizations, including research trawls and animal studies that require handling and tagging of individuals. From 2012-2016, nine mortalities were reported during research activities, resulting in a mean annual mortality and serious injury rate of 1.8 sea lions (Carretta *et al*., 2018b).

NOAA declared an unusual mortality event (UME) for California sea lions during 2013-2017. High mortality of pup and juvenile age classes were documented during this time and NOAA identified changes in the availability of sea lion prey species, particularly sardines, as a contributing factor. Changes in prey abundance and distribution have been linked to warm-water anomalies in the California Current that have impacted a wide range of marine taxa (Cavole *et al*. 2016).

**Habitat Concerns**

The algal neurotoxin domoic acid has been linked to mortality of California sea lions since 1998 (Scholin *et al*. 2000, Brodie *et al*. 2006, Ramsdell and Zabka 2008). Future mortality is expected to occur, due to the repeated occurrence of such harmful algal blooms.

Exposure to anthropogenic sound may impact individual sea lions. Experimental exposure of captive California sea lions to simulated mid-frequency sonar (Houser *et al*. 2013) and acoustic pingers (Bowles and Anderson 2012) resulted in a wide variety of behavioral responses, including increases in respiration, refusal to participate in food reward tasks, evasive hauling out, and prolonged submergence.

Expanding pinniped populations have resulted in increased human-caused serious injury and mortality, due to shootings, entrainment in power plants, interactions with hook and line fisheries, separation of mothers and pups due to human disturbance, dog bites, and vessel and vehicle strikes (Carretta *et al*. 2018b).

Increasing sea-surface temperatures in the California Current negatively impact prey species availability and reduce survival rates of California sea lions (DeLong *et al*. 2017, Laake *et al*. 2018, Lowry *et al*. 1991, Melin *et al*. 2008, 2010). Increasing ocean temperatures may continue to limit the population size of the California sea lion stock within the California Current (Cavole *et al*. 2016, DeLong *et al*. 2017, Laake *et al*. 2018).

**STATUS OF STOCK**

California sea lions in the U.S. are not listed as "endangered" or "threatened" under the Endangered Species Act or as "depleted" under the MMPA. The stock is estimated to be approximately 40% above its maximum net productivity level (MNPL = 183,481 animals), and it is therefore considered within the range of its optimum sustainable population (OSP) size (Laake *et al*. 2018). The carrying capacity of the population was estimated at 275,298 animals in 2014 (Laake *et al*. 2018). Mean annual commercial fishery mortality is 197 animals per year (Table 1). Other sources of human-caused mortality (shootings, direct removals, recreational hook, research-related and line fisheries, entrainment in power plant intakes) average 97 animals per year. Human-caused mortality and serious injury of this stock is $\geq$ 321 animals annually, which does not include undetected and unreported cases. California sea lions are not considered "strategic" under the MMPA because human-caused mortality is less than the PBR (14,011). The fishery mortality and serious injury rate (197 animals/year) for this stock is less than 10% of the calculated PBR and, therefore, is considered to be insignificant and approaching a zero mortality and serious injury rate.

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# HARBOR SEAL (*Phoca vitulina richardii*): California Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Harbor seals (*Phoca vitulina*) are widely distributed in the North Atlantic and North Pacific. Two subspecies exist in the Pacific: *P. v. stejnegeri* in the western North Pacific, near Japan, and *P. v. richardii* in the eastern North Pacific. The latter subspecies inhabits coastal and estuarine areas from Mexico to Alaska. These seals do not make extensive pelagic migrations, but do travel 300-500 km to find food or suitable breeding areas (Herder 1986; Harvey and Goley 2011). In California, approximately 400-600 harbor seal haulout sites are widely distributed along the mainland and on offshore islands, including intertidal sandbars, rocky shores and beaches (Hanan 1996; Lowry *et al*. 2008).

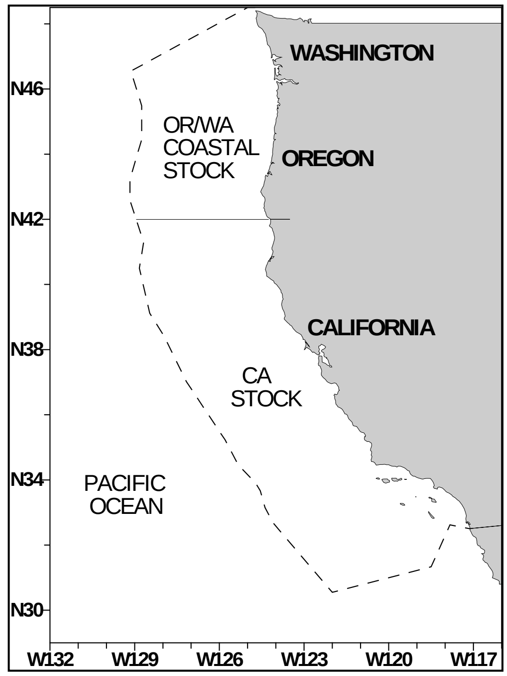


**Figure 1.** Stock boundaries for the California and Oregon/Washington coastal stocks of harbor seals. Dashed line represents the U.S. EEZ.

Within the subspecies *P. v. richardii*, abundant evidence of geographic structure comes from differences in mitochondrial DNA (Huber *et al*. 1994, 2010, 2012; Burg 1996; Lamont *et al*. 1996; Westlake and O'Corry-Crowe 2002; O'Corry-Crowe *et al*. 2003), mean pupping dates (Temte 1986), pollutant loads (Calambokidis *et al*. 1985), pelage coloration (Kelly 1981) and movement patterns (Jeffries 1985; Brown 1988). LaMont *et al.* (1996) identified four discrete subpopulation differences in mtDNA between harbor seals from Washington (two locations), Oregon, and California. Another mtDNA study (Burg 1996) supported the existence of three separate groups of harbor seals between Vancouver Island and southeastern Alaska. Three genetically distinct populations of harbor seals within Washington inland waters are also evident, based on work by Huber *et al.* (2010, 2012). Although geographic structure exists along an almost continuous distribution of harbor seals from California to Alaska, stock boundaries are difficult to draw because any rigid line is arbitrary from a biological perspective. Nonetheless, failure to recognize geographic structure in defining management stocks can lead to depletion of local populations. Previous assessments of the status of harbor seals have recognized three stocks along the west coast of the continental U.S.: 1) California, 2) Oregon and Washington outer coast waters, and 3) inland waters of Washington. Although the need for stock boundaries for management is real and is supported by biological information, the exact placement of a boundary between California and Oregon was largely a political/jurisdictional convenience. An unknown number of harbor seals also occur along the west coast of Baja California, at least as far south as Isla Asuncion, which is about 100 miles south of Punta Eugenia. Animals along Baja California are not considered to be a part of the California stock because it is not known if there is any demographically significant movement of harbor seals between California and Mexico and there is no international agreement for joint management of harbor seals. Lacking any new information on which to base a revised boundary, the harbor seals of California are treated as a separate stock in this report (Fig. 1). Other Marine Mammal Protection Act (MMPA) stock assessment reports cover the other stocks that are recognized along the U.S. west coast: (1) Southern Puget Sound (south of the Tacoma Narrows Bridge); (2) Washington Northern Inland Waters (including Puget Sound north of the Tacoma Narrows Bridge, the San Juan Islands, and the Strait of Juan de Fuca); (3) Hood Canal; and (4) Oregon/Washington Coast.

## POPULATION SIZE

A complete count of all harbor seals in California is impossible because not all animals are hauled out simultaneously. A complete pup count (as is done for other pinnipeds in California) is also not possible because harbor seal pups enter the water almost immediately after birth. Population size is estimated by counting the number of seals ashore during the peak haul-out period (May to July) and by multiplying this count by a correction factor equal to the inverse of the estimated fraction of seals on land. Harvey and Goley (2011) calculated a correction factor of 1.54 (CV=0.157), based on 180 radio-tagged seals in California. This correction factor is based on the mean of four date-specific correction factors (1.31, 1.38, 1.62, 1.84) calculated for central and northern California. Based on the most recent harbor seal counts during May-July of 2012 (20,109 animals) (NMFS unpublished data) and the Harvey and Goley (2011) correction factor, the harbor seal population in California in 2012 is estimated to number 30,968 seals (CV=0.157).

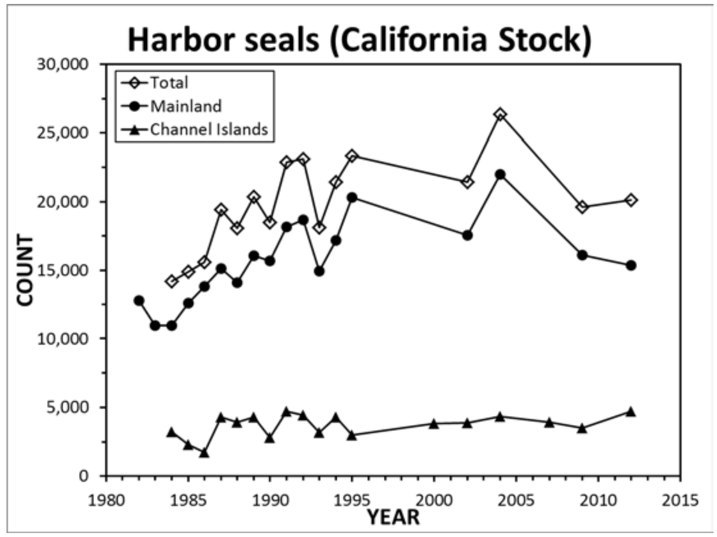


**Figure 2.** Harbor seal haulout counts in California during May to July (Hanan 1996; R. Read, CDFG unpubl. data; Lowry *et al*. 2008, NMFS unpubl. data from 2009-2012 surveys).

## Minimum Population Estimate

The minimum population size is estimated from the number of hauled out seals counted in 2012 (20,109), multiplied by the lower 20th percentile of the correction factor (1.36), or 27,348 seals.

## Current Population Trend

Counts of harbor seals in California increased from 1981 to 2004 when the statewide maximum count was recorded. Subsequent surveys conducted in 2009 and 2012 have been lower than the 2004 maximum count (Fig. 2).

# CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Historically, the largest known source of human-caused mortality of California harbor seals was the California halibut set gillnet fishery (Julian and Beeson 1998), where estimates of bycatch mortality were approximately 1-2% of the estimated population size between 1990 and 1995. Since 1996, that fishery been observed infrequently and at low observer coverage levels, though fishing effort levels have declined. Any estimate of current net productivity level should account for human-caused mortality, otherwise estimated net productivity will be negatively-biased. At this time, there are insufficient data on bycatch (only 3 of the last 5 years have observations from the fishery, with low observer coverage) and uncertainty regarding the degree of negative biases for other sources of human-caused mortality to reliably estimate the current net productivity level. An assessment of *maximum net productivity levels* is not possible, because abundance estimates were not available when the population was very small and presumably recovering from past exploitation (Bonnot 1928).

# POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (27,348) times one half the default maximum net productivity rate for pinnipeds (½ of 12%) times a recovery factor of 1.0 (for a stock of unknown status that is growing or for a stock at OSP, Wade and Angliss 1997), resulting in a PBR of 1,641 animals per year.

# HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Serious Injury Guidelines**

NMFS uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to distinguish serious from non-serious injury (Angliss and DeMaster 1998, Andersen et al. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*".

## Historical Takes

Prior to state and federal protection and especially during the nineteenth century, harbor seals along the west coast of North America were greatly reduced by commercial hunting (Bonnot 1928, 1951; Bartholomew and Boolootian 1960). Only a few hundred individuals survived in a few isolated areas along the California coast (Bonnot 1928). In the last half of the last century, the population increased dramatically.

## Fishery Information

A summary of known commercial fishery mortality and serious injury for this stock of harbor seals for the period 2008-2012 is given in Table 1. Historically, the set gillnet fishery for halibut and white seabass was the largest source of fishery mortality and remains the most likely fishery in California to interact with harbor seals. Julian and Beeson (1998) reported a range of annual mortality estimates from 227 to 1,204 seals (mean = 584) from 1990 to 1994, based on 5% to 15% fishery observer coverage and representing between 1-2% of the estimated population size. This fishery has been observed infreqently since 1995 and fishing effort has declined from approximately 5,000 trips in the early 1990s to 1,300 trips in 2012 (Carretta *et al*. 2014a.).

**Table 1.** Summary of available information on the mortality and serious injury of harbor seals (California stock) in commercial fisheries that might take this species (Carretta and Enriquez 2006, 2009, Carretta *et al*. 2014a; Heery *et al*. 2010); n/a indicates that data are not available. Mean annual takes are based on 2008-2012 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) |
|---|---|---|---|---|---|---|
| **CA halibut and white seabass set gillnet fishery** | 2008<br>2009<br>2010<br>2011<br>2012 | observer | 0%<br>0%<br>12.5%<br>8.0%<br>5.5% | 0<br>0<br>3<br>0<br>0 | n/a<br>n/a<br>23 (0.59)<br>n/a<br>n/a | 23 (0.59) |
| **CA small-mesh drift gillnet fishery for white seabass, yellowtail, barracuda, and tuna** | 2010<br>2011<br>2012 | observer | 0.7%<br>3.3%<br>4.6% | 0<br>0<br>0 | 0 (n/a)<br>0 (n/a)<br>0 (n/a) | 0 (n/a) |
| **WA, OR, CA groundfish trawl (includes at-sea hake and other limited-entry groundfish sectors)** | 2005<br>2006<br>2007<br>2008<br>2009 | observer | 99% to 100% of tows in at-sea hake fishery; 18%-26% of landings in other groundfish sectors | 1<br>1<br>0<br>4<br>1 | 1 (n/a)<br>1 (n/a)<br>0 (n/a)<br>29 (n/a)<br>1 (n/a) | 6.4 (n/a) |
| **Unknown net fisheries** | 2008-2012 | stranding | n/a | 5 | n/a | ≥ 1.0 |
| **Total annual takes** | | | | | | 30 (0.59) |

## Other Mortality

NMFS stranding records for California for the period 2008-2012 include the following human-caused mortality and serious injury not included in Table 1: shootings (1), ship/vessel strikes (3), entrainment in power plants (40), hook and line fisheries (6), human-induced abandonment of pups or harassment (9), marine debris entanglement (2), stabbing/gaff wounds (2), and research-related deaths (1) (Carretta *et al*. 2014b.). The total non-fishery related mortality and serious injury for the period totals 64 harbor seals, or an annual average of 12.8 seals.

## STATUS OF STOCK

A review of harbor seal dynamics through 1991 concluded that their status relative to OSP could not be determined with certainty (Hanan 1996). California harbor seals are not listed as "endangered" or "threatened" under the Endangered Species Act nor designated as "depleted" under the MMPA. Annual human-caused mortality from commercial fisheries (30/yr) and other human-caused sources (12.8/yr) is 42.8 animals, which is less than the calculated PBR for this stock (1,641), and thus they are not considered a "strategic" stock under the MMPA. The average annual rate of incidental commercial fishery mortality (30 animals) is less than 10% of the calculated PBR (1,641 animals); therefore, fishery mortality is considered insignificant and approaching zero mortality and serious injury rate. The population size has increased since the 1980s when statewide censuses were first conducted. The highest population counts occurred in 2004 and subsequent counts in 2009 and 2012 have been lower. 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. 2014b). All west-coast harbor seals that have been tested for morbilliviruses were found to be seronegative, indicating that this disease is not endemic in the population and that this population is extremely susceptible to an epidemic of this disease (Ham-Lammé *et al*. 1999).

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Revised 7/15/2014

# HARBOR SEAL (*Phoca vitulina richardii*): Oregon/Washington Coast Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Harbor seals inhabit coastal and estuarine waters off Baja California, north along the western coasts of the continental U.S., British Columbia, and Southeast Alaska, west through the Gulf of Alaska and Aleutian Islands, and in the Bering Sea north to Cape Newenham and the Pribilof Islands. They haul out on rocks, reefs, beaches, and drifting glacial ice and feed in marine, estuarine, and occasionally fresh waters. Harbor seals generally are non-migratory, with local movements associated with tides, weather, season, food availability, and reproduction (Scheffer and Slipp 1944; Fisher 1952; Bigg 1969, 1981). Harbor seals do not make extensive pelagic migrations, though some long distance movement of tagged animals in Alaska (900 km) and along the U.S. west coast (up to 550 km) have been recorded (Brown and Mate 1983, Herder 1986, Womble 2012). Harbor seals have also displayed strong fidelity to haulout sites (Pitcher and Calkins 1979, Pitcher and McAllister 1981).

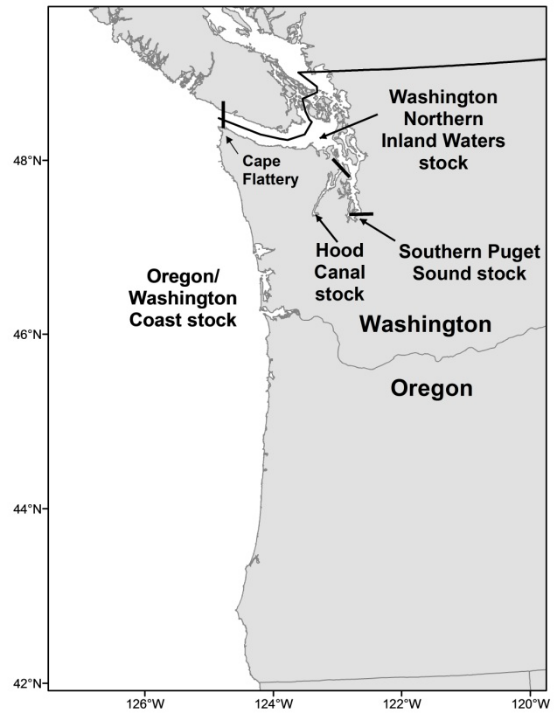


**Figure 1.** Harbor seal stocks in the U.S. Pacific Northwest

Until recently, differences in mean pupping date (Temte 1986), movement patterns (Jeffries 1985, Brown 1988), pollutant loads (Calambokidis et al. 1985), and fishery interactions led to the recognition of three separate harbor seal stocks along the west coast of the continental U.S. (Boveng 1988): 1) inland waters of Washington State (including Hood Canal, Puget Sound, and the Strait of Juan de Fuca out to Cape Flattery), 2) outer coast of Oregon and Washington, and 3) California. Recent genetic evidence suggests that the population of harbor seals in Washington inland waters has more structure than previously recognized. Studies of pupping phenology, mitochondrial DNA, and microsatellite variation of harbor seals in Washington and Canada-U.S. transboundary waters confirm the currently recognized stock boundary between the Washington Coast and Washington Inland Waters harbor seal stocks, but three genetically distinct populations of harbor seals within Washington inland waters are also evident (Huber et al. 2010, 2012). Within U.S. west coast waters, five stocks of harbor seals are recognized: 1) Southern Puget Sound (south of the Tacoma Narrows Bridge); 2) Washington Northern Inland Waters (including Puget Sound north of the Tacoma Narrows Bridge, the San Juan Islands, and the Strait of Juan de Fuca); 3) Hood Canal; 4) Oregon/Washington Coast; and 5) California. This report considers only the Oregon/Washington Coast stock. Stock assessment reports for California harbor seals and harbor seals in Washington inland waters (including the Southern Puget Sound, Washington Northern Inland Waters, and Hood Canal stocks) also appear in this volume. Harbor seal stocks that occur in the inland and coastal waters of Alaska are discussed separately in the Alaska Stock Assessment Reports. Harbor seals occurring in British Columbia are not included in any of the U.S. Marine Mammal Protection Act (MMPA) stock assessment reports.

## POPULATION SIZE

Aerial surveys of harbor seals in Oregon and Washington were conducted by personnel from the National Marine Mammal Laboratory (NMML) and the Oregon and Washington Departments of Fish and Wildlife (ODFW and WDFW) during the 1999 pupping season. Total numbers of hauled-out seals (including pups) were counted during these surveys. In 1999, the mean count of harbor seals occurring along the Washington coast was 10,430 (CV=0.14) animals (Jeffries et al. 2003). In 1999, the mean count of harbor seals occurring along the Oregon coast and in the Columbia River was 5,735 (CV=0.14) animals (Brown 1997; ODFW, unpublished data). Combining these counts results in 16,165 (CV=0.10) harbor seals in the Oregon/Washington Coast stock.

Radio-tagging studies conducted at six locations (three Washington inland waters sites and three Oregon and Washington coastal sites) collected information on haulout patterns from 63 harbor seals in 1991 and 61 harbor seals in 1992. Haulout data from coastal and inland sites were not significantly different and were thus pooled, resulting in a correction factor of 1.53 (CV=0.065) to account for animals in the water which are missed during the aerial surveys (Huber et al. 2001). Using this correction factor results in a population estimate of 24,732 (16,165 x 1.53; CV=0.12) for the Oregon/Washington Coast stock of harbor seals in 1999 (Jeffries et al. 2003; ODFW, unpublished data). However, because the most recent abundance estimate is >8 years old, there is no current estimate of abundance available for this stock.

**Minimum Population Estimate**

No current information on abundance is available to obtain a minimum population estimate for the Oregon/Washington Coast stock of harbor seals.

**Current Population Trend**

Historical levels of harbor seal abundance in Oregon and Washington are unknown. The population apparently decreased during the 1940s and 1950s due to state-financed bounty programs. Approximately 17,133 harbor seals were killed in Washington by bounty hunters between 1943 and 1960 (Newby 1973). More than 3,800 harbor seals were killed in Oregon between 1925 and 1972 by bounty hunters and a state-hired seal hunter (Pearson 1968). The population remained relatively low during the 1960s but, since the termination of the harbor seal bounty program and with the protection provided by the passage of the MMPA in 1972, harbor seal counts for this stock have increased from 6,389 in 1977 to 16,165 in 1999 (Jeffries et al. 2003; ODFW, unpublished data). Based on the analyses of Jeffries et al. (2003) and Brown et al. (2005), both the Washington and Oregon portions of this stock were reported as reaching carrying capacity (Fig. 2). In the absence of recent abundance estimates, the current population trend is unknown.

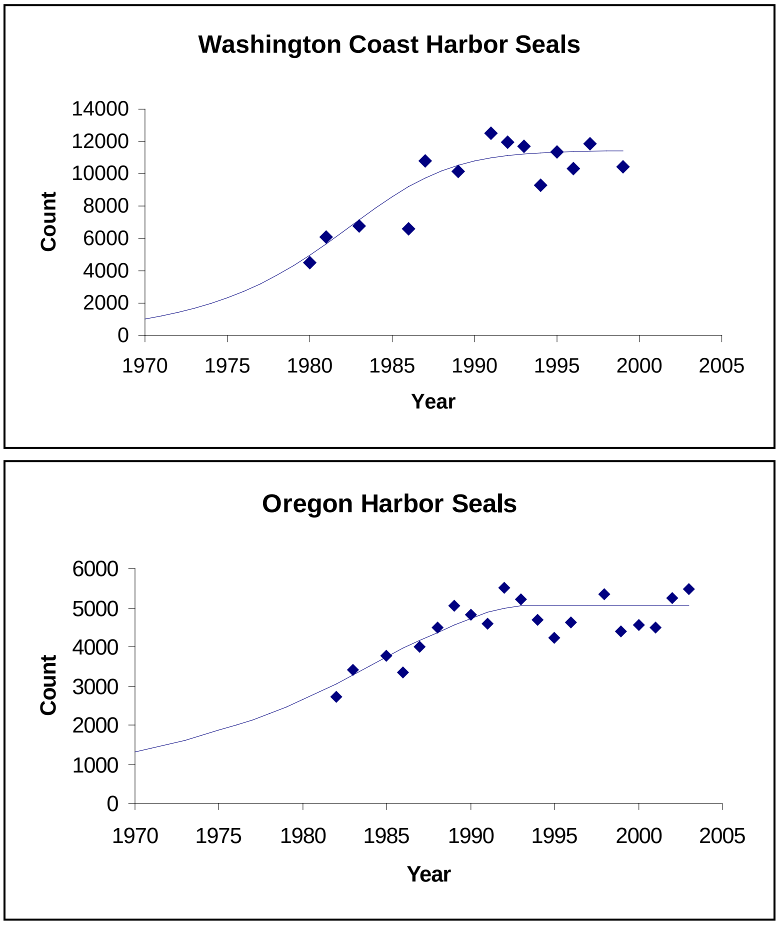


**Figure 2.** Generalized logistic growth curves of Washington Coast (Jeffries et al. 2003) and Oregon (Brown et al. 2005) harbor seals.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

The Oregon/Washington Coast harbor seal stock increased at an annual rate of 7% from 1983 to 1992 and at 4% from 1983 to 1996 (Jeffries et al. 1997). Because the population was not at a very low level by 1983, the observed rates of increase may underestimate the maximum net productivity rate ($R_{MAX}$). When a logistic model was fit to the Washington portion of the 1975-1999 abundance data, the resulting estimate of $R_{MAX}$ was 18.5% (95% CI = 12.9-26.8%) (Jeffries et al. 2003). When a logistic model was fit to the Oregon portion of the 1977-2003 abundance data, estimates of $R_{MAX}$ ranged from 6.4% (95% CI = 4.6-27%) for the south coast of Oregon to 10.1% (95% CI = 8.6-20%) for the north coast (Brown et al. 2005). Until a combined analysis for the entire stock is completed, the pinniped default maximum theoretical net productivity rate ($R_{MAX}$) of 12% will be used for this harbor seal stock (Wade and Angliss 1997).

**POTENTIAL BIOLOGICAL REMOVAL**

Because there is no current estimate of minimum abundance, a potential biological removal (PBR) cannot be calculated for this stock.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**New Serious Injury Guidelines**

NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998, Andersen *et al*. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

**Fisheries Information**

Fishing effort in the northern Washington marine gillnet tribal fishery is conducted within the range of the Oregon/Washington Coast and Washington Northern Inland Waters stocks of harbor seals. Movement of animals between Washington's coastal and inland waters is likely, although tagging data do not show movement of harbor seals between the two locations (Huber et al. 2001). For the purposes of this report, animals taken in waters south and west of Cape Flattery, WA, are assumed to belong to the Oregon/Washington Coast stock and Table 1 includes data only from that portion of the fishery. Fishing effort in the coastal marine set gillnet tribal fishery has declined since 2004. A test set gillnet fishery, with 100% observer coverage, was conducted in coastal waters in 2008 and 2010. This test fishery required the use of nets equipped with acoustic alarms, and observers reported one harbor seal death in 2008 and three in 2010 (Makah Fisheries Management, unpublished data). The mean annual mortality for the marine set gillnet tribal fishery in 2007-2011 is 0.8 (CV=0) harbor seals from observer data.

The U.S. West Coast groundfish fishery was monitored for incidental takes in 2005-2009 (Jannot et al. 2011). Harbor seal deaths were observed in the groundfish trawl fishery (Pacific hake at-sea processing component) in 2005, 2006, and 2008; the nearshore fixed gear fishery in 2006 and 2008; and the non-nearshore fixed gear (limited entry non-primary sablefish) fishery in 2009. The mean annual mortality for each of these fisheries in 2005-2009 is 1.0 (CV=0.24) harbor seals for the groundfish trawl fishery, 5.6 (CV=0.68) for the nearshore fixed gear fishery, and 0.2 for the non-nearshore fixed gear fishery.

**Table 1.** Summary of available information on the incidental mortality and serious injury of harbor seals (Oregon/Washington Coast stock) in commercial and tribal fisheries that might take this species and calculation of the mean annual mortality rate; n/a indicates that data are not available. Mean annual takes are based on 2007-2011 data unless otherwise noted.

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Northern WA marine set gillnet (tribal test fishery in coastal waters) | 2007<br>2008<br>2009<br>2010<br>2011 | observer data | no fishery<br>100%<br>no fishery<br>100%<br>no fishery | 0<br>1<br>0<br>3<br>0 | 0 (0)<br>1 (0)<br>0 (0)<br>3 (0)<br>0 (0) | 0.8 (0) |
| West Coast groundfish trawl (Pacific hake at-sea processing component) | 2005<br>2006<br>2007<br>2008<br>2009 | observer data | 67%[1]<br>83%[1]<br>73%[1]<br>76%[1]<br>79%[1] | 1<br>1<br>0<br>2<br>0 | 1 (0.52)<br>1 (0.42)<br>0<br>3 (0.34)<br>0 | 1.0 (0.24) |
| West Coast groundfish nearshore fixed gear | 2005<br>2006<br>2007<br>2008<br>2009 | observer data | 5%[2]<br>11%[2]<br>9%[2]<br>7%[2]<br>4%[2] | 0<br>1<br>0<br>2<br>0 | 0<br>n/a[3]<br>0<br>27 (0.68)<br>0 | 5.6 (0.68) |

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| West Coast groundfish non-nearshore fixed gear (limited entry non-primary sablefish) | 2009 | observer data | n/a | 1 | n/a[3] | >0.2 (n/a) |
| WA Grays Harbor salmon drift gillnet[2] | 1991-1993 | observer data | 4-5% | 0, 1, 1 | 0, 10, 10 | see text[24] |
| WA Willapa Bay drift gillnet[2] | 1991-1993 | observer data | 1-3% | 0, 0, 0 | 0, 0, 0 | see text[24] |
| WA Willapa Bay drift gillnet[2] | 1990-1993 | fisherman self-reports | n/a | 0, 0, 6, 8 | n/a | see text[24] |
| Unknown West Coast fisheries | 2007-2011 | stranding data | n/a | 0, 0, 0, 0, 3 | n/a | >0.6 (n/a) |
| Minimum total annual takes | | | | | | >8.2 (0.52) |

[1]Percent hauls observed for marine mammals.
[2]Percent observed landings of target species.
[3]Bycatch estimate not provided due to high CV (>80%) for estimate; minimum bycatch of one observed harbor seal is included in the calculation of mean annual take.
[4]This fishery has not been observed since 1993 (see text); these data are not included in the calculation of recent minimum total annual takes.

Commercial salmon drift gillnet fisheries in Washington outer coast waters (Grays Harbor, Willapa Bay) were last observed in 1993 and 1994, with observer coverage levels typically less than 10% (Erstad et al. 1996, Pierce et al. 1994, Pierce et al. 1996, NWIFC 1995). Drift gillnet fishing effort in the outer coast waters has declined considerably since 1994 because fewer vessels participate today (NMFS NW Region, unpublished data), but entanglements of harbor seals likely continue to occur. The most recent data on harbor seal mortality from commercial and tribal gillnet fisheries is included in Table 1.

Combining recent estimates from commercial fisheries observer data for the West Coast groundfish trawl (1.0), West Coast groundfish nearshore fixed gear (5.6), and West Coast groundfish non-nearshore fixed gear (0.2) fisheries results in a mean annual mortality rate of 6.8 harbor seals from these fisheries. An additional 0.8 harbor seals per year were taken in the northern Washington marine set gillnet tribal fishery.

Strandings of harbor seals entangled in fishing gear or with serious injuries caused by interactions with gear are another source of fishery-related mortality. Based on stranding network data, there were three commercial fishery-related deaths of harbor seals from this stock reported in 2011 (listed as unknown West Coast fisheries in Table 1), resulting in a mean annual mortality of 0.6 harbor seals in 2007-2011. Fishery entanglements included two gillnet and one trawl net interaction. Hook and line gear is used by both commercial (salmon troll) and recreational fisheries in coastal waters. Two harbor seal deaths due to ingested hooks were reported in 2007-2011, resulting in an additional mean annual mortality of 0.4 seals from unknown hook and line fisheries. Estimates from stranding data are considered minimum estimates because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel). An additional harbor seal that stranded with a serious hook injury in 2011 was treated and released with non-serious injuries (Carretta et al. 2013); therefore, it was not included in the mean annual mortality in this report.

Data on fisheries mortality reported in Table 1 likely represent minimum estimates, particularly for fisheries where observer coverage is low and bycatch events are too infrequent to be documented by fishery observers. The magnitude of negative bias in mortality estimates is unknown and methods to correct for such negative biases in these fisheries have not been developed.

**Other Mortality**

During 2007-2011, one harbor seal from this stock was incidentally killed during scientific halibut longline operations in 2011, resulting in a mean annual research-related mortality of 0.2 animals.

According to Northwest Marine Mammal Stranding Network records, maintained by the NMFS Northwest Region (NMFS, Northwest Regional Office, unpublished data), a total of nine human-caused harbor seal deaths were reported from non-fisheries sources in 2007-2011. Six animals were shot, two animals were struck by boats, and one animal was killed by a dog, resulting in a mean annual mortality of 1.8 harbor seals from this stock. 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).

**Subsistence Harvests by Northwest Treaty Indian Tribes**

Tribal subsistence takes of this stock may occur, but no data on recent takes are available.

## STATUS OF STOCK

Harbor seals are not considered to be "depleted" under the MMPA or listed as "threatened" or "endangered" under the ESA. Based on currently available data, the minimum level of human-caused mortality and serious injury is 10.6 harbor seals per year: (8.2 from fishery sources in Table 1, plus 0.4 from unknown hook and line fisheries, plus 0.2 scientific takes annually, plus 1.8 non-fishery causes annually). A PBR cannot be calculated for this stock because there is no current abundance estimate. Human-caused mortality relative to PBR is unknown, but it is considered to be small relative to the stock size. Therefore, the Oregon/Washington Coast stock of harbor seals is not classified as a "strategic" stock. The minimum annual commercial fishery mortality and serious injury for this stock, based on recent observer data (6.8) and stranding data (0.6) is 7.4. Since a PBR cannot be calculated for this stock, fishery mortality relative to PBR is unknown. The stock was previously reported to be within its Optimum Sustainable Population (OSP) range (Jeffries et al. 2003, Brown et al. 2005), but in the absence of recent abundance estimates, this stock's status relative to OSP is unknown.

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Revised 7/15/2014

# HARBOR SEAL (*Phoca vitulina richardii*): Washington Inland Waters Stocks: (Hood Canal, Southern Puget Sound, Washington Northern Inland Waters)

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Harbor seals inhabit coastal and estuarine waters off Baja California, north along the western coasts of the continental U.S., British Columbia, and Southeast Alaska, west through the Gulf of Alaska and Aleutian Islands, and in the Bering Sea north to Cape Newenham and the Pribilof Islands. They haul out on rocks, reefs, beaches, and drifting glacial ice and feed in marine, estuarine, and occasionally fresh waters. Harbor seals generally are non-migratory, with local movements associated with such factors as tides, weather, season, food availability, and reproduction (Scheffer and Slipp 1944; Fisher 1952; Bigg 1969, 1981). Harbor seals do not make extensive pelagic migrations, though some long distance movement of tagged animals in Alaska (900 km) and along the U.S. west coast (up to 550 km) have been recorded (Brown and Mate 1983, Herder 1986, Womble 2012). Harbor seals have also displayed strong fidelity for haulout sites (Pitcher and Calkins 1979, Pitcher and McAllister 1981).

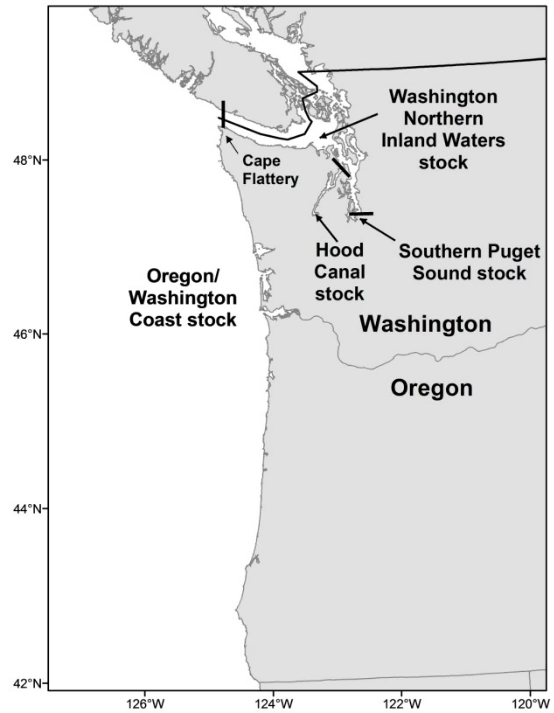


**Figure 1.** Approximate distribution of harbor seal stocks in the U.S. Pacific Northwest (shaded area). Stock boundaries separating the three stocks are shown.

Until recently, differences in mean pupping date (Temte 1986), movement patterns (Jeffries 1985, Brown 1988), pollutant loads (Calambokidis et al. 1985), and fishery interactions have led to the recognition of three separate harbor seal stocks along the west coast of the continental U.S. (Boveng 1988): 1) inland waters of Washington State (including Hood Canal, Puget Sound, and the Strait of Juan de Fuca out to Cape Flattery), 2) outer coast of Oregon and Washington, and 3) California. Recent genetic evidence suggests that the population of harbor seals in Washington inland waters has more structure than is currently was previously recognized . Studies of pupping phenology, mitochondrial DNA, and microsatellite variation of harbor seals in Washington and Canada-U.S. transboundary waters confirm the currently recognized stock boundary between the Washington Coast and Washington Inland Waters harbor seal stocks, but three genetically distinct populations of harbor seals within Washington inland waters are also evident (Huber et al. 2010, 2012). Within U.S. west coast waters, five stocks of harbor seals are recognized: 1) Southern Puget Sound (south of the Tacoma Narrows Bridge); 2) Washington Northern Inland Waters (including Puget Sound north of the Tacoma Narrows Bridge, the San Juan Islands, and the Strait of Juan de Fuca); 3) Hood Canal; 4) Oregon/Washington Coast; and 5) California. This report includes only the stocks in Washington's inland waters. Stock assessment reports for Oregon/Washington Coast and California harbor seals also appear in this volume. Harbor seal stocks that occur in the inland and coastal waters of Alaska are discussed separately in the Alaska Stock Assessment Reports. Harbor seals occurring in British Columbia are not included in any of the U.S. Marine Mammal Protection Act (MMPA) stock assessment reports.

## POPULATION SIZE

Aerial surveys of harbor seals in Washington were conducted during the pupping season in 1999, during which time the total numbers of hauled-out seals (including pups) were counted. In 1999, the mean count of harbor seals occurring in Washington's inland waters was 7,213 (CV=0.14) in Washington Northern Inland Waters, 711 (CV=0.14) in Hood Canal, and 1,025 (CV=0.14) in Southern Puget Sound (Jeffries et al. 2003).

Radio-tagging studies conducted at six locations (three Washington inland waters sites and three Oregon and Washington coastal sites) collected information on haulout patterns from 63 harbor seals in 1991 and 61 harbor seals in 1992. Data from coastal and inland sites were not significantly different and were thus pooled, resulting in a correction factor of 1.53 (CV=0.065) to account for animals in the water which are missed during the aerial surveys (Huber et al. 2001). Using this correction factor results in a population estimates of 11,036 (7,213 x 1.53; CV=0.15) for the Washington Northern Inland Waters stock; 1,088 (711 x 1.53; CV=0.15) for the Hood Canal stock; and 1,568 (1,025 x 1.53; CV=0.15) for the Southern Puget Sound stock of harbor seals (Jeffries et al. 2003). However, because the most recent abundance estimates are >8 years old, there are no current estimates of abundance for these stocks. Surveys of harbor seals in Washington inland waters are planned for 2013.

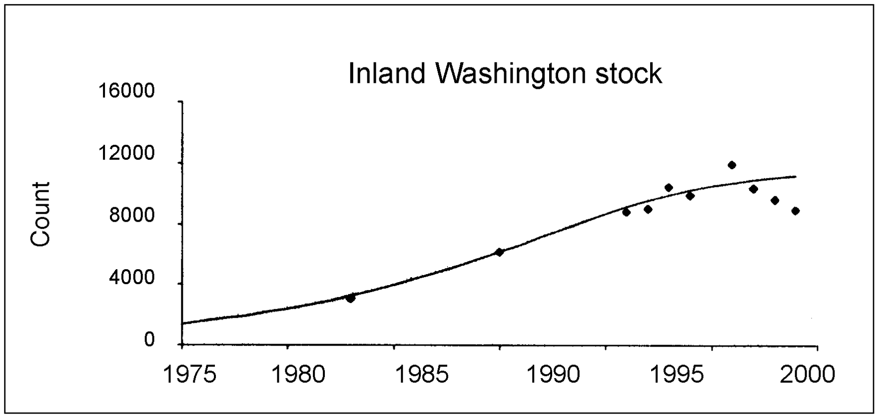


**Figure 2.** Generalized logistic population growth curve for the Washington Inland Waters stock of harbor seals, 1978-1999 (Jeffries et al. 2003).

**Minimum Population Estimate**

No current information on abundance is available to obtain a minimum population estimate for the Washington Inland Waters stock of harbor seals.

**Current Population Trend**

Historical levels of harbor seal abundance in Washington are unknown. The population apparently decreased during the 1940s and 1950s due to a state-financed bounty program. Approximately 17,133 harbor seals were killed in Washington by bounty hunters between 1943 and 1960 (Newby 1973). The population remained relatively low during the 1970s but, since the termination of the harbor seal bounty program in 1960 and with the passage of the Marine Mammal Protection Act (MMPA) in 1972, harbor seal numbers in Washington have increased (Jeffries 1985).

Between 1983 and 1996, the annual rate of increase for this stock was 6% (Jeffries et al. 1997). The peak count occurred in 1996 and, based on a fitted generalized logistic model (Fig. 2), the population is thought to be stable (Jeffries et al. 2003). In the absence of recent abundance estimates, the current population trend is unknown.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

From 1991 to 1996, counts of harbor seals in Washington State have increased at an annual rate of 10% (Jeffries et al. 1997). Because the population was not at a very low level by 1991, the observed rate of increase may underestimate the maximum net productivity rate ($R_{MAX}$). When a logistic model was fit to the 1978-1999 abundance data, the resulting estimate of $R_{MAX}$ was 12.6% (95% CI = 9.4-18.7%) (Jeffries et al. 2003). This value of $R_{MAX}$ is very close to the default pinniped maximum theoretical net productivity rate of 12% ($R_{MAX}$), therefore, 12% will be employed for this harbor seal stock (Wade and Angliss 1997).

## POTENTIAL BIOLOGICAL REMOVAL

Because there is no current estimate of minimum abundance, a potential biological removal (PBR) cannot be calculated for this stock.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**New Serious Injury Guidelines**

NMFS updated its serious injury designation and reporting process, which uses guidance from previous serious injury workshops, expert opinion, and analysis of historic injury cases to develop new criteria for distinguishing serious from non-serious injury (Angliss and DeMaster 1998, Andersen *et al*. 2008, NOAA 2012). NMFS defines serious injury as an "*injury that is more likely than not to result in mortality*". Injury determinations for stock assessments revised in 2013 or later incorporate the new serious injury guidelines, based on the most recent 5-year period for which data are available.

**Fisheries Information**

Fishing effort in the northern Washington marine gillnet tribal fishery is conducted within the range of the Oregon/Washington Coast and Washington Northern Inland Waters stocks of harbor seals. Some movement of animals between Washington's coastal and inland waters is likely, although data from tagging studies have not shown movement of harbor seals between the two locations (Huber et al. 2001). For the purposes of this stock assessment report, the animals taken in waters east of Cape Flattery, WA, are assumed to have belonged to the Washington Northern Inland Waters stock, and Table 1 includes data only from that portion of the fishery. There was no observer coverage in the northern Washington marine set gillnet tribal fishery in inland waters in 2007-2011; however, there were two fishermen self-reports of harbor seal deaths in this fishery in 2008 and five in 2009 (Makah Fisheries Management, unpublished data). The mean annual mortality for this fishery in 2007-2011 is 1.4 harbor seals from self-reports. Fishing effort in the northern Washington marine drift gillnet tribal fishery in inland waters is also conducted within the range of the Washington Northern Inland Waters stock of harbor seals. This fishery is not observed; however, there was one self-report of a harbor seal death in 2008 (Makah Fisheries Management, unpublished data). The mean annual mortality for this fishery in 2007-2011 is 0.2 harbor seals from self-reports.

Commercial salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994, with observer coverage levels typically less than 10% (Erstad et al. 1996, Pierce et al. 1994, Pierce et al. 1996, NWIFC 1995). Drift gillnet fishing effort in the inland waters has declined considerably since 1994 because far fewer vessels participate today (NMFS NW Region, unpublished data), but entanglements of harbor seals likely continue to occur. The most recent data on harbor seal mortality from commercial gillnet fisheries is included in Table 1.

**Table 1.** Summary of available information on the incidental mortality and serious injury of harbor seals (Washington Northern Inland Waters, Hood Canal, and Southern Puget Sound stocks) in commercial and tribal fisheries that might take this species and calculation of the mean annual mortality rate; n/a indicates that data are not available. Mean annual takes are based on 2007-2011 data unless noted otherwise.

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Northern WA marine set gillnet (tribal fishery in inland waters) | 2008<br>2009 | fisherman self-reports | - | 2<br>5 | n/a<br>n/a | 1.4 (n/a) |
| Northern WA marine drift gillnet (tribal fishery in inland waters) | 2008 | fisherman self-reports | - | 1 | n/a | >0.2 (n/a) |
| WA Puget Sound Region salmon set/drift gillnet (observer programs listed below covered segments of this fishery): | - | - | - | - | - | - |
| Puget Sound non-treaty salmon gillnet (all areas and species) | 1993 | observer data | 1.3% | 2 | n/a | see text |
| Puget Sound non-treaty chum salmon gillnet (areas 10/11 and 12/12B)[1] | 1994 | observer data | 11% | 1 | 10 | see text[1] |
| Puget Sound treaty chum salmon gillnet (areas 12, 12B, and 12C)[1] | 1994 | observer data | 2.2% | 0 | 0 | see text[1] |
| Puget Sound treaty chum and sockeye salmon gillnet (areas 4B, 5, and 6C)[1] | 1994 | observer data | 7.5% | 0 | 0 | see text[1] |
| Puget Sound treaty and non-treaty sockeye salmon gillnet (areas 7 and 7A)[1] | 1994 | observer data | 7% | 1 | 15 | see text[1] |

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Unknown Washington Northern Inland Waters fisheries | 2007-2011 | stranding data | n/a | 1, 1, 1, 1, 2 | n/a | ≥1.2 (n/a) |
| Unknown Hood Canal fisheries | 2007-2011 | stranding data | n/a | 0, 0, 0, 0, 1 | n/a | > 0.2 (n/a) |
| Unknown Southern Puget Sound fisheries | 2007-2011 | stranding data | n/a | 0, 5, 0, 0, 0 | n/a | >1.0 (n/a) |
| Minimum total annual takes Washington Northern Inland Waters | | | | | | > 2.8 (n/a) |
| Minimum total annual takes Hood Canal | | | | | | > 0.2 (n/a) |
| Minimum total annual takes Southern Puget Sound | | | | | | >1.0 (n/a) |

[1]This fishery has not been observed since 1994 (see text); these data are not included in the calculation of recent minimum total annual takes.

Strandings of harbor seals entangled in fishing gear or with serious injuries caused by interactions with gear are a final source of fishery-related mortality information. As these strandings could not be attributed to a particular fishery, they have been included in Table 1 as occurring in unknown Washington inland waters fisheries. According to Northwest Marine Mammal Stranding Network records, maintained by the NMFS Northwest Region (NMFS, Northwest Regional Office, unpublished data), 12 fishery-related harbor seal deaths and serious injuries were reported in Washington inland waters in 2007-2011: six from the Washington Northern Inland Waters stock, one from the Hood Canal stock, and five from the Southern Puget Sound stock, resulting in mean annual takes of 1.2 harbor seals in Washington Northern Inland Waters, 0.2 in Hood Canal, and 1.0 in Southern Puget Sound. Fishery interactions included two gaff injuries, two gillnet entanglements, in one fishing net entanglement, and one entanglement in fishing gear in Washington Northern Inland Waters; one gillnet entanglement in Hood Canal; and five gillnet entanglements in Southern Puget Sound. Harbor seal deaths caused by interactions with recreational hook and line fishing gear were also reported in 2007-2011: two seals had hook injuries and one ingested a hook in Washington Northern Inland Waters and two seals ingested hooks in Southern Puget Sound, resulting in mean annual mortalities of 0.6 and 0.4, respectively, from these two stocks. Estimates from stranding data are considered minimum estimates because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel). Two additional harbor seals that stranded with serious hook injuries from recreational hook and line gear in Washington Northern Inland Waters in 2007-2011 were treated and released with non-serious injuries (Carretta et al. 2013); therefore, they were not included in the mean annual mortality in this report.

**Other Mortality**

According to Northwest Marine Mammal Stranding Network records, maintained by the NMFS Northwest Region, a total of 32 human-caused harbor seal deaths or serious injuries were reported from non-fisheries sources in 2007-2011 for the Washington Northern Inland Waters stock. Eight animals were shot, 13 nine were struck by boats, two died in oil spills, three two were killed by dogs, and 13 were entangled in marine debris, resulting in a mean annual mortality of 6.4 harbor seals from this stock. During the same time period, 10 human-caused deaths or serious injuries were reported for the Southern Puget Sound stock: one animal entangled in marine debris, six were shot, one was killed by a dog, one entangled in a buoy line, and one entangled in a scientific research net, resulting in a mean annual mortality of 2.0 harbor seals. These are considered minimum estimates because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel). An additional seriously injured harbor seal was disentangled from marine debris and released with non-serious injuries in Washington Northern Inland Waters in 2007 (Carretta et al. 2013); therefore, it was not included in the mean annual mortality in this report.

**Subsistence Harvests by Northwest Treaty Indian Tribes**

Tribal subsistence takes of this stock may occur, but no data on recent takes are available.

## STATUS OF STOCK

Harbor seals are not considered to be "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Based on currently available data, the minimum level of human-caused mortality and serious injury is 9.8 harbor seals per year for the Washington Northern Inland Waters stock (2.8 from fishery sources in Table 1 + 0.6 from recreational hook and line fisheries + 6.4 from non-fishery sources). Annual human-caused serious injury and mortality for the Hood Canal stock is 0.2 from unknown fishery sources. Annual human-caused serious injury and mortality for the Southern Puget Sound stock is 3.4, including 1.0 from fishery sources listed in Table 1, 0.4 from recreational hook and line fisheries, and 2.0 from non-fishery sources. PBRs cannot be calculated for these stocks because there are no current abundance estimates. Human-caused mortality relative to PBR is unknown for these stocks, but is considered to be small relative to stock size. Therefore, the Washington Northern Inland Waters, Hood Canal, and Southern Puget Sound stocks of harbor seals are not classified as "strategic" stocks. At present, the minimum annual fishery mortality and serious injury for these stocks (based on stranding data) are 1.2 for the Washington Northern Inland Waters stock, 0.2 for the Hood Canal stock, and 1.0 for the Southern Puget Sound stock. Since a PBR cannot be calculated for these stocks, fishery mortality relative to PBR is unknown. The stock was previously reported to be within its Optimum Sustainable Population (OSP) range (Jeffries et al. 2003), but in the absence of recent abundance estimates, this stock's status relative to OSP is unknown.

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Revised 03/14/2022

# NORTHERN ELEPHANT SEAL (*Mirounga angustirostris*): California Breeding Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Northern elephant seals breed and give birth in California (U.S.) and Baja California (Mexico), primarily on offshore islands (Stewart *et al*. 1994), from December to March (Stewart and Huber 1993). Spatial segregation in foraging areas between males and females is evident from satellite tag data (Le Beouf *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 pelagic areas in the Gulf of Alaska and the central North Pacific to feed on pelagic prey (Le Beouf *et al*. 2000). Adults return to land between March and August to molt, with males returning later than females. Adults return to their feeding areas again between their spring/summer molting and their winter breeding seasons.

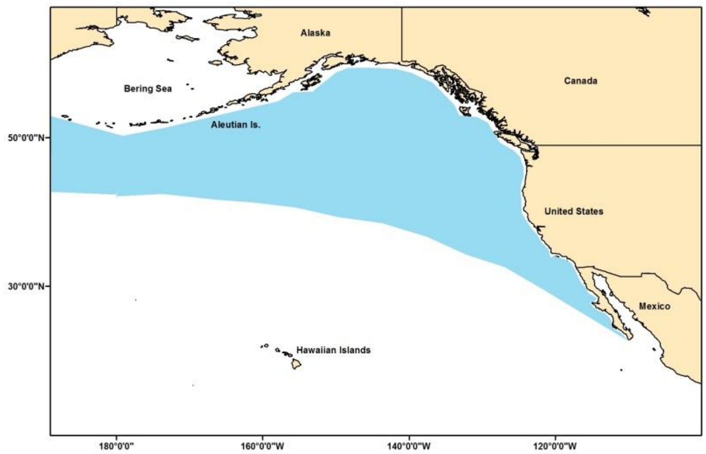


**Figure 1.** Pelagic range of northern elephant seals in the eastern North Pacific. Major breeding rookeries occur along the west coast of Baja California and the California coast, as described in Lowry *et al*. (2014).

Populations of northern elephant seals in the U.S. and Mexico have recovered after being nearly hunted to extinction (Stewart *et al*. 1994). Northern elephant seals underwent a severe population bottleneck and loss of genetic diversity when the population was reduced to an estimated 10-30 individuals (Hoelzel *et al*. 2002). Although movement and genetic exchange continues between rookeries, most elephant seals return to natal rookeries when they start breeding (Huber *et al*. 1991). The California breeding population is now demographically isolated from the Baja California population. No international agreements exist for the joint management of this species by the U.S. and Mexico. The California breeding population is considered here to be a separate stock.

## POPULATION SIZE

A complete population count of elephant seals is not possible because all age classes are not ashore simultaneously. Elephant seal population size is estimated by counting the number of pups produced and multiplying by the inverse of the expected ratio of pups to total animals (McCann 1985). Based on counts of elephant seals at U.S. Channel Islands rookeries in 2013, Lowry *et al*. (2020) reported 34,788 pups were born. This value represents the sum of live pups (33,454) and estimated pre-census pup mortality (1,334), but it excludes un-surveyed areas in central and northern California (Lowry *et al*. 2020). Lowry *et al*. (2014) reported that 81.5% of the U.S. population resided at the Channel Islands and uses the inverse of this percentage to estimate statewide births, which is 42,685 pups. Lowry *et al.* (2020) extrapolated from total births to a statewide population estimate of 187,386 (95% CI 161,876 – 214,418). This correction factor is based on life table data on elephant seal fecundity and survival rates, where approximately 23% of the population represents pups (Cooper and Stewart, 1983; Le Boeuf and Reiter, 1988; Hindell, 1991; Huber *et al*., 1991; Reiter and Le Boeuf, 1991; Clinton and Le Boeuf, 1993; Le Boeuf *et al*., 1994; Pistorius and Bester, 2002; McMahon *et al*., 2003; Pistorius *et al*., 2004; Condit *et al*., 2014).

### Minimum Population Estimate

The minimum population size for northern elephant seals in 2013 can be estimated conservatively as 85,369 seals, which is equal to twice the estimated statewide pup count (to account for the pups and their mothers).

### Current Population Trend

The population is reported to have grown at 3.1% annually since 1988 (Lowry *et al*. 2020).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATE

An annual growth rate of 17% for elephant seals in the U.S. from 1958 to 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 to 2013, the population is estimated to have grown 3.1% annually (Lowry *et al*. 2020). For this stock assessment report, we use the default maximum theoretical net productivity rate for pinnipeds, or 12% (Wade and Angliss 1997).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size ( 85,369) times one half the observed maximum net growth rate for this stock (½ of 12%) times a recovery factor of 1.0 (for a stock of unknown status that is increasing, Wade and Angliss 1997) resulting in a PBR of 5,122 animals per year.

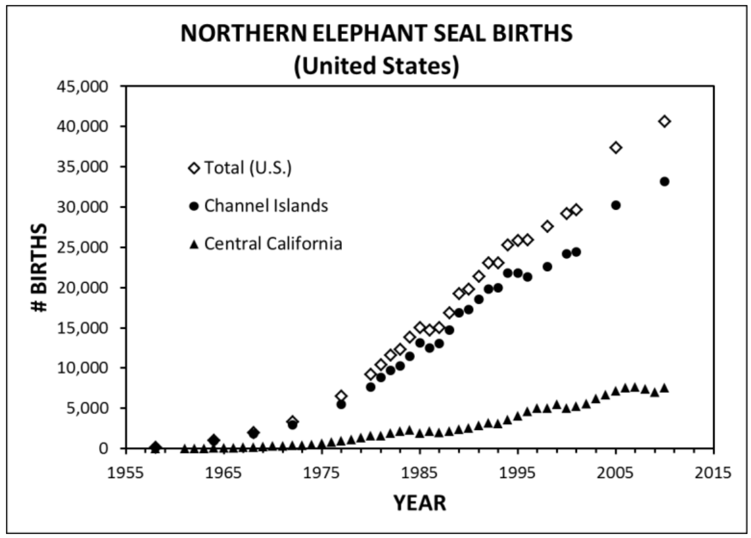


**Figure 2.** Estimated number of northern elephant seal births in California 1958-2010. Multiple independent estimates are presented for the Channel Islands 1988-91. Estimates are from Stewart *et al*. (1994), Lowry *et al*. (1996), Lowry (2002), Lowry *et al*. (2014), and unpublished data from Sarah Allen, Dan Crocker, Brian Hatfield, Ron Jameson, Bernie Le Boeuf, Mark Lowry, Pat Morris, Guy Oliver, Derek Lee, and William Sydeman.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

A summary of known commercial fishery mortality and serious injury for this stock of northern elephant seals is given in Table 1. Total estimated commercial fishery mortality is ≥ 5.3 elephant seals annually (Table 1). Although all of the mortality in Table 1 occurred in U.S. waters, some may be of seals from Mexico's breeding population that are migrating through U.S. waters.

### Other Mortality

For the period 2015-2019, deaths and serious injuries from the following non-commercial fishery sources were documented: shootings (2); marine debris entanglement (4); hook and line fisheries (2); dog attack (1); unidentified human interaction (2); harassment (7); vehicle collision (1); tar/oil (22); and vessel strike (1) (Carretta *et al*. 2021). These non-commerical fishery sources of mortality and serious injury total 42 animals, or an average of 8.4 elephant seals annually (Carretta *et al*. 2014b).

**Table 1.** 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 et al. 2020a, 2020b, Jannot *et al.* 2018). n/a indicates information is not available. Mean annual takes are based on 2015-2019 data unless noted otherwise. The California halibut and white seabass set gillnet fishery has been observed only sporadically in recent years and no elephant seal entanglements have been recorded in this fishery since 2000 when the fishery operated north of Point Conception.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA thresher shark/swordfish drift gillnet fishery** | 2015-2019 | observer | 21% | 3 | 10.8 (0.41) | 2.2 (0.41) |
| **CA halibut and white seabass set gillnet fishery** | 2017 | observer | ~10% | 0 | 0 | 0 (n/a) |
| **California halibut trawl fishery open access** | 2012<br>2013<br>2014<br>2015<br>2016 | observer | 0.06<br>0.06<br>0.22<br>0.33<br>0.30 | 0<br>0<br>0<br>0<br>1 | 0.63 (n/a)<br>0.76 (n/a)<br>0.63 (n/a)<br>0.60 (n/a)<br>1.61 (n/a) | 0.85 (n/a) |
| **Limited Entry Sablefish Hook and Line** | 2012<br>2013<br>2014<br>2015<br>2016 | observer | 0.22<br>0.22<br>0.27<br>0.42<br>0.33 | 1<br>0<br>0<br>3<br>0 | 2.33 (n/a)<br>0.95 (n/a)<br>0.87 (n/a)<br>3.86 (n/a)<br>1.08 (n/a) | 1.82 (n/a) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)** | 2012-2016 | observer data | 98% to 100% of tows in at-sea hake fishery | 2 | 2 | 0.4 (n/a) |
| **Total annual takes** | | | | | | ≥ 5.3 (n/a) |

## STATUS OF STOCK

Northern elephant seals are not listed as "endangered" or "threatened" under the Endangered Species Act nor designated as "depleted" under the MMPA. Total annual human-caused mortality ( commercial fishery (5.3) + other sources (8.4) = 13.7) is less than the calculated PBR for this stock ( 5,122), thus northern elephant seals are not considered a "strategic" stock under the MMPA. The average rate of incidental fishery mortality for this stock over the last five years ≥ 5.3) is less than 10% of the calculated PBR (5,122); therefore, the total fishery serious injury and mortality appears to be insignificant and approaching a zero mortality and serious injury rate. The population growth rate between 1958 and 1987 was 17% annually (Lowry *et al*. 2014). From 1988 to 2013, the population grew at an annual rate of 3.1% (Lowry *et al*. 2020). The population continues to grow, with ~80% of births occurring at southern California rookeries (Lowry *et al*. 2014, 2020). No estimate of carrying capacity is available for this population and the population status relative to 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*. 2021).

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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, as 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 and Lowry 1999) and a pup was born at San Miguel Island, California (Melin and 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 almost annually in California waters and animals are increasingly observed in Oregon and Washington waters. In 2015-2016, Guadalupe fur seal strandings totaled approximately 175 animals along the coast of California (compared with approximately 10 animals annually in prior years), and NMFS declared an Unusual Mortality Event. Most strandings involved animals less than 2 years old with evidence of malnutrition. Individuals have stranded or been sighted inside the Gulf of California and as far south as Zihuatanejo, Mexico (Hanni et al. 1997 and Aurioles-Gamboa and Hernadez-Camacho 1999) and another in 2012, at Cerro Hermoso, Oaxaca, Mexico (Esperon-Rodriguez and Gallo-Reynoso 2012). Recent video records of pinnipeds hooked in the mouth from international waters west of the California Current involving the shallow set Hawaii longline fishery were independently reviewed by pinniped experts and at least one animal in early 2016 was identified as a Guadalupe fur seal. 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. 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.

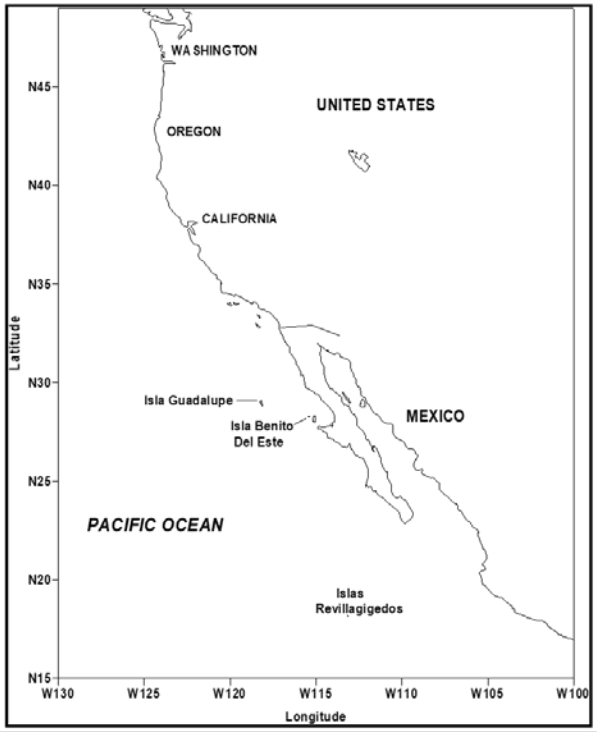


**Figure 1.** Geographic range of the Guadalupe fur seal, showing location of two rookeries at Isla Guadalupe and Isla Benito Del Este.

## POPULATION SIZE

Population size prior to commercial harvests in the 19th century is unknown, but estimates range from 20,000 to 100,000 animals (Fleischer 1987). García-Aguilar *et al*. (2018) estimate current population size is approximately one-fifth of its historical pre-exploitation size. The most recent estimate of population size is based on pup count data collected in 2013 and a range of correction factors applied to pup counts to account for uncounted age classes and pre-census pup mortality (García-Aguilar *et al.* 2018). The 2013 estimates are based on 4,924 pups counted from a boat survey of Isla Guadalupe and corrected for pre-census mortality, resulting in an estimated 9,768 pups (range 8,863 – 10,869) born. García-Aguilar *et al.* (2018) estimated total population size by scaling up pup counts assuming two different total population size to pup count ratios (3.5:1 and 4.5:1) that have been used as defaults for other pinniped populations (Harwood and Prime 1978). Resulting estimates were 34,187 individuals (range 31,019–38,043), and 43,954 individuals (range 39,882–48,912). These estimates do not include animals at San Benito Island, for which Elorriaga-Verplancken *et al.* (2016) counted a maximum of 3,710 animals (including 28 pups) and 1,494 animals (16 pups) in July of 2014 and 2015, respectively. García-Aguilar *et al*. (2018) and Elorriaga-Verplancken *et al.* (2016) note that the San Benito Island rookery is represented almost exclusively by immature animals migrating from Guadalupe Island, and that negligible numbers of pups are produced at San Benito.

**Minimum Population Estimate**

The minimum population size is taken as the lower bound of the estimate provided by García-Aguilar *et al*. (2018) using a population size:pup count ratio of 3.5, or 31,019 animals.

**Current Population Trend**

Counts of Guadalupe fur seals have been made sporadically since 1954 and are compiled by Seagars (1984), Fleischer (1987), Gallo (1994), Torres et al. (1990), and García-Capitanachi (2011). Historic counts vary in reliability in that some census efforts represent partial counts, either of age classes or lack complete spatial coverage of Guadalupe Island. A more recent study, based on only pup counts between 1984 and 2013 at Guadalupe Island, resulted in an estimated annual rate of increase of 5.9% (range 4.1–7.7%) (García-Aguilar *et al*. 2018) (Figure 2). This estimate of annual rate of increase does not include years prior to 1984 when the population was considerably smaller and higher population growth rates would be expected as the population recovered from historic anthropogenic removals.

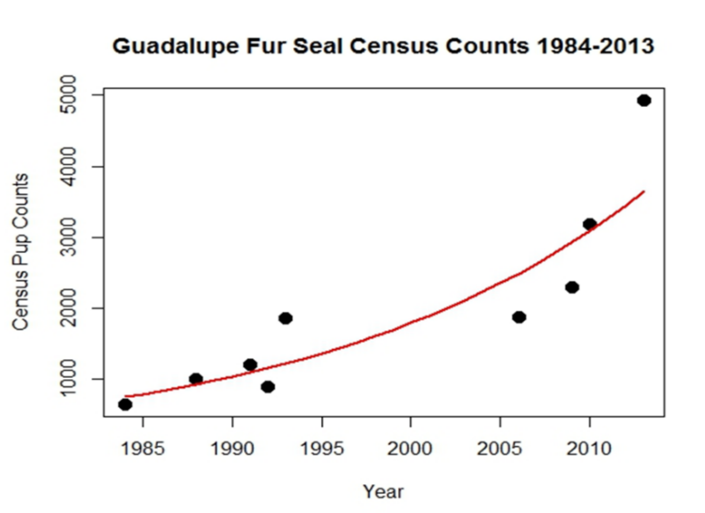


**Figure 1.** Guadalupe fur seal census counts through time.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Reported annual growth rates of 21% at Isla San Benito over an 11-year period are too high and likely result from immigration from Isla Guadalupe (Esperón-Rodríguez and 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).

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) for this stock is calculated as the minimum population size (31,019) <u>times</u> one half the maximum net growth rate observed for this species (½ of 13.7%) <u>times</u> a recovery factor of 0.5 (for a threatened species, Wade and Angliss 1997), resulting in a PBR of 1,062 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.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

Table 1. Summary of available information on the incidental mortality and serious injury of Guadalupe fur seals in commercial fisheries and other unidentified fisheries that might take this species.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality and Serious Injury (and non-serious injuries) | Estimated Mortality and Serious Injury (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA driftnet fishery for sharks and swordfish** | 2013-2017 | observer | 12%-37% | 0 | 0 | 0 |
| **CA set gillnet fishery for halibut/white seabass and other species** | 2013-2017 | observer | <10% | 0 | 0 | 0 |
| **Hawaii Shallow Set Longline Fishery** | 2013-2017 | observer | 100% | 2 (2) | 2 (0) | 0.4 (0) |
| Unidentified fishery interactions, including generic gillnets of unknown origin | 2013-2017 | strandings | n/a | 4 (1) | ≥ 4 | ≥ 0.8 |
| **Minimum total annual takes** | | | | | | ≥ 1.2 |

No Guadalupe fur seals have been observed entangled in California gillnet fisheries between 1990 and 2017 (Julian and Beeson 1998, Carretta *et al*. 2004, Carretta *et al*. 2016b, Carretta *et al*. 2019a, 2019b), although stranded animals have been found entangled in gillnet of unknown origin (see 'Other mortality' below). 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 Hawaii shallow set longline fishery (100% observer coverage, Table 1). Between 2013 and 2017 there were 2 serious and 2 non-serious injuries involving this species (Carretta *et al*. 2019a). These interactions occurred outside of the U.S. EEZ, west of the California Current.

### Other mortality and serious injury

There were 13 records of human-related deaths and/or serious injuries to Guadalupe fur seals from stranding data for the most recent 5-year period of 2013-2017 (Carretta *et al.* 2016a, Carretta *et al*. 2019a). These strandings included entanglement in marine debris and shootings. The average annual observed human-caused mortality and serious injury of Guadalupe fur seals for 2013-2017 from non-fishery sources is 2.6 animals annually (13 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 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. There is insufficient information to determine whether fishery mortality in Mexico exceeds the PBR for this stock, but given the observed growth of the population over time, this is unlikely. The total U.S. commercial fishery mortality and serious injury for this stock (≥1.2 animals per year) is less than 10% of the calculated PBR 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 (≥1.2) and other sources (≥2.6) is 3.8 animals per year, which is less than the range-wide PBR of 1,062 animals for this stock. The population was estimated to grow at 5.9% annually for the period 1984 to 2013 (García-Aguilar *et al*. 2018).

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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 (Fig. 1). As of 2014, the worldwide population size is approximately 1.1 million animals (Gelatt *et al*. 2015). During the breeding season, approximately 45% of the worldwide population is found on the Pribilof Islands in the southern Bering Sea, with the remaining animals spread throughout the North Pacific Ocean (Gelatt *et al*. 2015). Of the seals in U.S. waters outside of the Pribilofs, approximately 9% of the population is found on Bogoslof Island in the southern Bering Sea, 1% on San Miguel Island off southern California, and 0.3% on the Farallon Islands off central California (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).

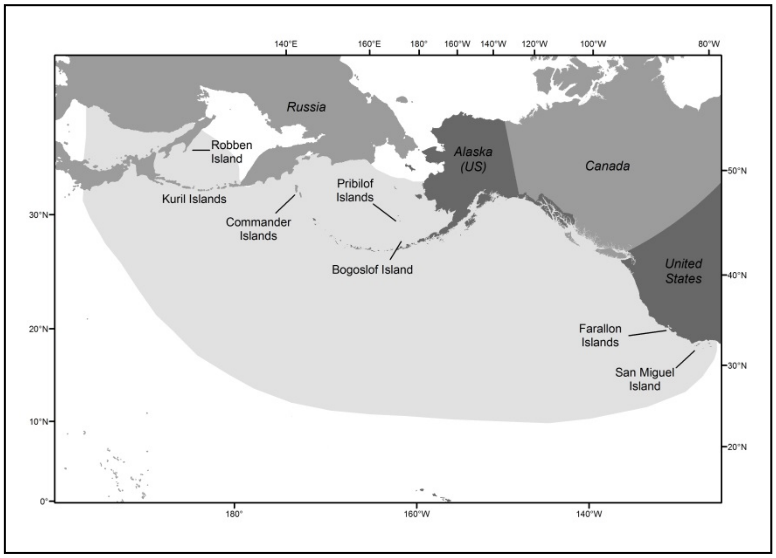


**Figure 1.** Approximate distribution of northern fur seals in the North Pacific (shaded area).

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 through 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 to 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 Pribilofs and San Miguel Island (DeLong 1982, DeLong and 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 and a California stock (including San Miguel Island and the Farallon Islands). The Eastern Pacific stock is reported separately in the Stock Assessment Reports for the Alaska Region.

## POPULATION SIZE

The population estimate for northern fur seals on San Miguel 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-olds, three-year-olds, and animals at least four years old. The resulting population estimate was equal to the pup count multiplied by 4.475. The expansion factors are 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 2010 the highest total pup count of 3,408 was recorded (Orr *et al*. in review). A possible cause for the decline in total pup counts from 2010 to 2011 was a combination of oceanographic events that occurred in the California Current in 2009, a coastal upwelling relaxation event in May and June and an El Niño event from Fall 2009 to Spring 2010. The oceanographic events caused fewer reproductive males and females to return to San Miguel Island to breed in 2010. During 2012, the population increased 9.4% from 2011 and this level was maintained during 2013. No counts were conducted at Castle Rock in 2014; however, a record number of pups (2,289) were counted at Adam's Cove that year. Additionally, the second highest number of territorial bulls (224) was observed in 2014 (Orr *et al*. in review). Based on these factors, and assuming the trends were similar at Castle Rock, the population size during 2014 would have been the highest recorded. However, based on the 2013 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 13,384 (3,346 x 4.0) northern fur seals (Orr *et al*. in review). 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.

The population estimate for northern fur seals on the Farallon Islands 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 in order to reduce human disturbance to other breeding pinnipeds and nesting seabirds; 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 to 2013 (Tietz 2012, Berger *et al*. 2013). Based solely on the count, the population estimate of northern fur seals at the Farallon Islands was 666 in 2013 and increased to 1,019 in 2014 (Orr *et al*. in review).

The most recent population estimate for the entire stock of California northern fur seals, which incorporates estimates from San Miguel Island and the Farallon Islands in 2013, is 14,050 (13,384 + 666).

**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 2013 (Tietz 2012, Berger *et al*. 2013, Orr *et al*. in review). The minimum number of animals at San Miguel Island is twice the pup count (3,346 x 2 = 6,692), to account for pups and mothers, plus the number of territorial males (166) counted the same year (i.e., 2013), or 6,858 fur seals. The minimum number at the Farallon Islands is the total number of individuals (666) counted during the survey in 2013. It should be noted that 1,019 individuals were counted in 2014, but this number is not used here to be consistent with data collected at San Miguel Island. The total minimum population size is the sum of the minimum population sizes at San Miguel Island (6,858) and the Farallon Islands (666) in 2013, or 7,524 northern fur seals.

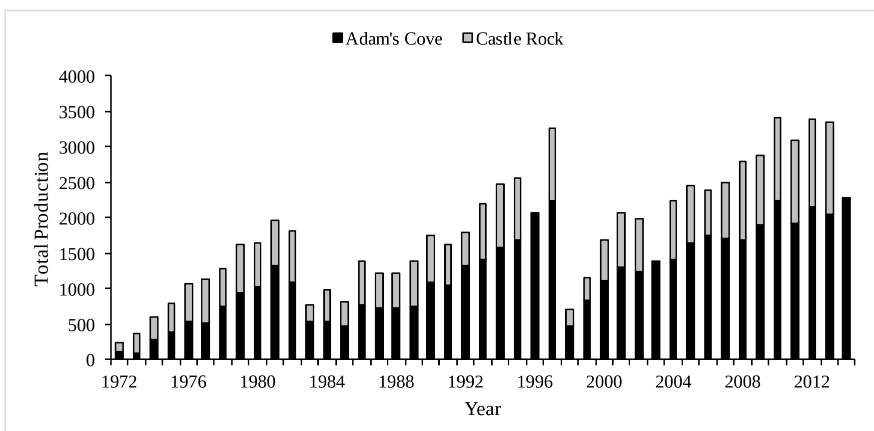


**Figure 2.** Total production of northern fur seal pups counted on San Miguel Island, including the mainland (Adam's Cove) and the offshore islet (Castle Rock), 1972-2014.

**Current Population Trend**

Northern fur seals were extirpated on San Miguel Island 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 and 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 and Antonelis 1991; Melin and DeLong 1994, 2000; Melin *et al*. 1996, 2005, 2008; Orr *et al*. 2012, in review).

Live pup counts increased about 24% annually from 1972 through 1982 (Fig. 2), 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 and 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 and 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).

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 3,408 in 2010 and, except for a slight decrease during 2011, levels have remained around 3,350 individuals in subsequent years (Orr *et al*. in review). The total production of northern fur seals has exceeded the 1997 levels during three of the last four years with complete counts; therefore, the San Miguel Island population has recovered from the 1997-1998 El Niño event.

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 San Miguel Island. 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 potential biological removal (PBR) level for this stock is calculated as the minimum population estimate (7,524) 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 and Angliss 1997), resulting in a PBR of 451 northern fur seals from the California stock per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fisheries Information**

Northern fur seals taken by commercial fisheries during the winter/spring along the west coast of the continental U.S. 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 through May will be assigned to both the Eastern Pacific and California stocks of northern fur seals. There were no observer reports of northern fur seal deaths or serious injuries in any observed fishery along the west coast of the continental U.S. in 2009-2013 (Carretta and Enriquez 2010, 2012a, 2012b; Jannot *et al*. 2011; Carretta *et al*. 2014a, 2015).

**Table 1.** 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 2009-2013 data unless noted otherwise.

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Unknown West Coast fisheries | 2009-2013 | stranding data | n/a | 1, 0, 2, 1, 0 | n/a | ≥0.8 (n/a) |
| Minimum total annual takes | | | | | | ≥0.8 (n/a) |

Strandings of northern fur seals entangled in fishing gear or with serious injuries caused by interactions with gear are another source of fishery-related mortality information. According to stranding records for California, Oregon, and Washington (Carretta *et al*. 2014b, 2015), four fishery-related deaths (in unidentified net and unknown trawl fisheries) were reported between 2009 and 2013 (Table 1), resulting in a mean annual mortality and serious injury rate of 0.8 California northern fur seals. 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). Two of the fishery-related deaths (one in an unidentified fishing net in February 2009 and one in trawl gear in April 2011) were also assigned to the Eastern Pacific stock of northern fur seals. Two additional northern fur seal strandings in 2012 (one in May and one in July) with serious injuries due to fishery interactions were treated and released with non-serious injuries (Carretta *et al*. 2014b). Both of these animals were assigned to the California stock of northern fur seals and the animal that stranded in May 2012 was also assigned to the Eastern Pacific stock.

**Other Mortality**

Since the Eastern Pacific and California stocks of northern fur seals overlap off the west coast of the continental U.S. during December through May, non-fishery mortality and serious injury reported off the coasts of California, Oregon, or Washington during that time will be assigned to both stocks. Mortality and serious injury of northern fur seals may occur incidental to research fishery activities. In 2007 and 2008, four northern fur seals were incidentally killed in California waters during scientific sardine trawling operations conducted by NMFS (Carretta *et al*. 2013): one death in 2007 and one in 2008 occurred before NMFS scientists implemented a mitigation plan to avoid future mortality. The initial mitigation plan included use of 162 dB acoustic pingers, a marine mammal watch, and scheduling trawls to occur when the ship first arrived on station to avoid attracting animals to a stationary vessel. Two additional northern fur seals were killed in subsequent 2008 trawls, so a marine mammal excluder device was added to the trawls in 2009 and no northern fur seal deaths or serious injuries were observed in this research fishery in 2009-2013. However, one northern fur seal was killed in a scientific rockfish trawling operation conducted by NMFS (Carretta *et al*. 2014b) in California waters in May 2009. This death was assigned to both the California and Eastern Pacific stocks of northern fur seals. The mean annual research-related mortality and serious injury rate of California northern fur seals from 2009 to 2013 is 0.2 northern fur seals.

According to stranding records for California, Oregon, and Washington (Carretta *et al*. 2014b, 2015), four human-caused northern fur seal deaths were reported from non-fisheries sources in 2009-2013. Three northern fur seals were entangled in marine debris in Oregon waters in April 2009 and one was entrained in the cooling water system of a California power plant in May 2012. All four of these deaths were assigned to both the California and Eastern Pacific stocks of northern fur seals. The mean annual mortality and serious injury rate from non-fishery sources in 2009-2013 is 0.8 California northern fur seals. 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 Marine Mammal Protection Act (MMPA) or listed as "threatened" or "endangered" under the Endangered Species Act. Based on currently available data, the minimum annual level of total human-caused mortality and serious injury (1.8) does not exceed the PBR (451). 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 (0.8) is not known to exceed 10% of the calculated PBR (45) 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, began to recover in 1999, and currently has surpassed the 1997 level by 2%. 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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# 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 (MHI). 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 1,564 (95% confidence interval 1,475 – 1,719; CV = 0.05), (Table 1, Johanos 2023a, b, c). In 2016, new approaches were developed to estimate Hawaiian monk seal abundance, both range-wide and at individual subpopulations (Baker *et al.* 2016, Harting *et al.* 2017). Obtaining abundance estimates for all NWHI subpopulations requires sea-going vessel support for approximately 56 days. In brief, 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 requires sufficient field presence to convincingly identify all the seals present, which is typically not achieved at most sites (Baker *et al*. 2006). When total enumeration is not possible, capture-recapture estimates (using Program CAPTURE) are conducted (Baker 2004; Otis *et al*. 1978, Rexstad & Burnham 1991, White *et al*. 1982). When no reliable estimator is obtainable in Program 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, Ni'ihau 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 MHI other than Niihau 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). Note that because 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 2021, total enumeration was not achieved at any subpopulation. Consequently, capture-recapture estimates were obtained at French Frigate Shoals, Laysan and Lisianski Islands, and at Pearl and Hermes Reef. Discovery curve analysis was used to generate abundance estimates at Midway and Kure Atolls (Table 1). 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 MHI, NMFS collects information on seal sightings reported throughout the year by a variety of sources, including a volunteer network, the public, and directed NMFS observation effort. A small number of surveys of Ni'ihau and nearby Lehua Islands are conducted through a collaboration between NMFS, Ni'ihau residents and the US Navy. Total MHI monk seal abundance is estimated by adding the number of individually identifiable seals documented during a calendar year on all MHI other than Ni'ihau 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 MHI 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 Ni'ihau and Lehua Islands was the total beach count at those sites (less individual seals already counted at other MHI) divided by the mean proportion of time hauled out in the MHI (Wilson *et al*., 2017). The total pups observed at Ni'ihau and Lehua Islands were added to obtain the total (Table 1).

Table 1. Total and minimum estimated abundance ($N_{min}$) of Hawaiian monk seals by location in 2021. 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. Note that 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, Ni'ihau/Lehua, and range-wide).

| | **Total** | | | **Nmin** | | | |
|---|---|---|---|---|---|---|---|
| **Location** | **Non-pups** | **Pups** | **Total** | **Non-pups** | **Pups** | **Total** | **Method** |
| French Frigate Shoals | 196 | 45 | 241 | 178 | 45 | 223 | CR |
| Laysan | 195 | 37 | 232 | 190 | 37 | 227 | CR |
| Lisianski | 140 | 19 | 159 | 130 | 19 | 149 | CR |
| Pearl & Hermes Reef | 130 | 18 | 148 | 124 | 18 | 142 | CR |
| Midway | 78 | 17 | 95 | 75 | 17 | 92 | DC |
| Kure | 85 | 19 | 104 | 82 | 19 | 101 | DC |
| Necker | 93 | 11 | 104 | 77 | 11 | 88 | CC |
| Nihoa | 82 | 3 | 85 | 68 | 3 | 71 | CC |
| MHI Kauai to Hawaii | 184 | 23 | 207 | 184 | 23 | 207 | Min |
| Ni'hau/Lehua | 148 | 20 | 168 | 124 | 20 | 144 | CC |
| Range-wide total | 1352 | 212 | 1564 | 1232 | 212 | 1444 | --- |

**Minimum Population Estimate**

The total numbers of seals identified at the NWHI subpopulations other than Necker and Nihoa, and in the MHI other than Ni'ihau and Lehua, are the best estimates of minimum population size at those sites. Minimum population sizes for Necker, Nihoa, Ni'ihau, 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 (1,444) are presented in Table 1.

**Current Population Trend**

Range-wide abundance estimates are available from 2013 to 2021 (Table 1, Figure 1). While these estimates remain somewhat negatively-biased for reasons explained in Baker *et al*. (2016), they provided a much more comprehensive assessment of status and trends than has been previously available. A Monte Carlo approximation of the annual multiplicative rate of realized population growth during 2013-2021 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 (1.01, 1.03). Thus, the best estimate is that the population grew at an average rate of about 2% per year from 2013 to 2021.

**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 2023a). Consistent with this value, a life table analysis representing a time when the MHI monk seal population was apparently expanding, yielded an estimated intrinsic population growth rate of 1.07 (Baker *et al*. 2011).

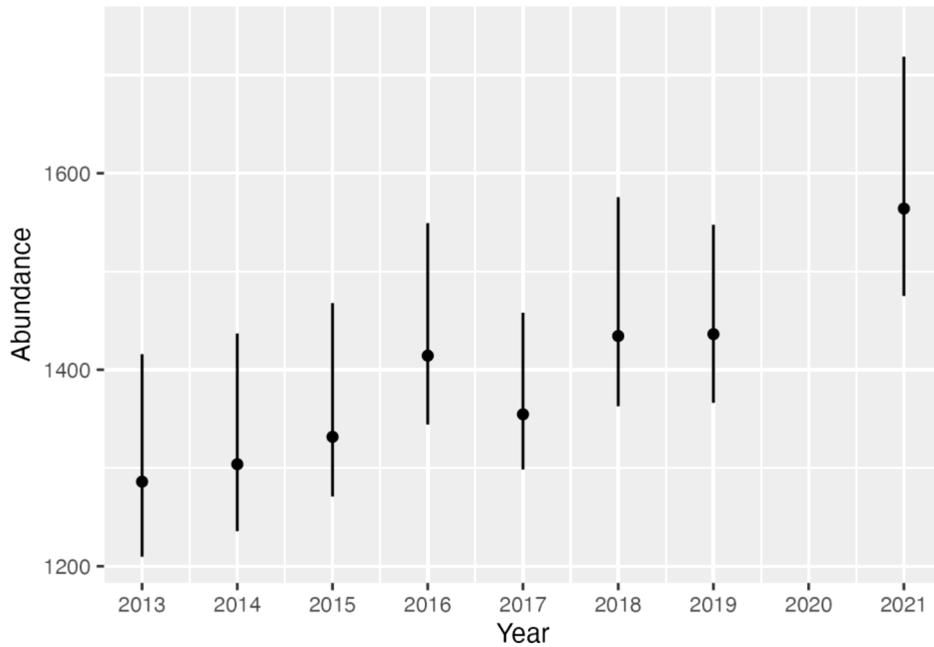


**Figure 1.** Range-wide abundance of Hawaiian monk seals, 2013-2021. Medians and 95% confidence limits are shown. Estimates prior to 2021 are re-estimated based on new data and represent negligible changes compared with values reported in the previous final stock assessments. (Table 1).

**POTENTIAL BIOLOGICAL REMOVAL**

Using current minimum population size (1,444), $R_{max}$ (0.07) and a recovery factor ($F_r$) for ESA endangered stocks (0.1), yields a Potential Biological Removal (PBR) of 5.1.

**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 and Bryan 1912; Wetmore 1925; Bailey 1952; Clapp and Woodward 1972). Following a period of at least partial recovery in the first half of the 20$^{th}$ 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 and Gilmartin 1990). Currently, human activities in the NWHI are limited and human disturbance is relatively rare, but human-seal interactions, have become an important issue in the MHI. Intentional killing of seals in the MHI is an ongoing and serious concern (Table 2). In 2021, three seals were bludgeoned or shot to death, all on Molokai.

**Table 2**. Intentional and potentially intentional killings of MHI monk seals, and anthropogenic mortalities not associated with fishing gear during 2017-2021 (Johanos 2022d, Mercer 2022). There were no confirmed cases in 2016, 2019, nor 2020.

| Year | Age/sex | Island | Cause of Death | Comments |
|---|---|---|---|---|
| 2017 | Adult female | Kauai | Trauma | Suspect intentional |
| 2017 | Juvenile female | Molokai | Blunt force trauma | Suspect intentional |
| 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 the 46% of carcasses of monk seals which died in the MHI during 2004-2019 were detected. Consequently, the cases in Table 2 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 MHI, especially involving nearshore fisheries managed by the State of Hawaii (Gobush et al. 2016). There are no fisheries operating in or near the NWHI. In 2021, 29 seal hookings were documented, two of which were classified as serious, and 27 as non-serious, injuries. Of the non-serious injuries, two would have been deemed serious had they not been mitigated (Henderson 2019a, Mercer 2023). Monk seals also interact with nearshore gillnets, and several confirmed deaths have resulted. In 2021, two seals became entangled in gillnets and were released alive, and were consequently classified as non-serious injuries. One adult seal was discovered swimming inside a mariculture pen and was displaced outside the pen through an existing hole. No mortality or injuries have been attributed to the MHI bottomfish handline fishery, and no interactions with longline fisheries have occurred since 1991. Consequently, these fisheries are no longer included in Table 3. 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 MHI bottomfish handline fishery.

**Table 3.** 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** | 2017<br>2018<br>2019<br>2020<br>2021 | Incidental observations of seals | None | 3<br>0<br>3<br>4<br>2 | n/a | 19(6)<br>11(3)<br>17(5)<br>29(4)<br>30(4) | ≥ 2.4 |
| **Mariculture** | 2017<br>2018<br>2019<br>2020<br>2021 | Incidental Observation | None | 1<br>0<br>0<br>0<br>0 | n/a | 0<br>0<br>0<br>0<br>1 | 0.2 (2.2) |
| **Minimum total annual takes** | | | | | | | ≥ 2.6 |

**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 MHI 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 Hawaii), and NMFS along with partner agencies is actively 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; Henderson 2019b, Mercer 2023). The number of marine debris entanglements documented in the past five years (Table 4) 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. 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 Hawaii 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 Hawaii since the 1980s.

**Table 4.** 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) |
|---|---|---|
| 2017 | 0 | 15(8) |
| 2018 | 1 | 15(6) |
| 2019 | 0 | 16(10) |
| 2020 | 0 | 5(1) |
| 2021 | 0 | 11(6) |
| **Minimum total annual takes** | ≥ 0.2 | |

The NMFS and partner agencies 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, Donohue *et al*. 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 Hawaii, but several hundred thousand feral and domestic cats occur throughout the MHI. 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 (2017-2021) seven monk seal deaths (representing a minimum average of 1.4 deaths per year) have been directly attributed to toxoplasmosis (Mercer 2021). 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 MHI.

**Other Mortality**

Sources of mortality that impede recovery include food limitation (see Habitat Issues), 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 NMFS (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 naïve monk seal population. Key disease threats include West Nile virus, morbillivirus and influenza.

**STATUS OF STOCK**

In 1976, the Hawaiian monk seal was designated depleted under the Marine Mammal Protection Act of 1972 and as endangered under the Endangered Species Act of 1973 (NMFS 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 (2017-2021) was at least 5.4 animals, including fishery-related mortality in nearshore gillnets, hook-and-line gear, and mariculture (≥ 2.6/yr, Table 3), intentional killings and other human-caused mortalities (≥ 1.2/yr, Table 2), entanglement in marine debris (≥ 0.2/yr, Table 4), and deaths due to toxoplasmosis (≥ 1.4/yr). The minimum rate of annual human-caused mortality exceeds PBR (5.1).

**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 and Thompson 2007, Baker *et al*. 2007, Baker 2008). Multiple strategies for improving juvenile survival, including translocation and captive care are being implemented (Baker and 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, NMFS 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 increasing entrapment hazard for monk seals and other fauna. The situation has worsened since 2012, when the USFWS ceased operations on Tern Island, thus 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 NMFS staff are on site. Furthermore, sea wall 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 MHI monk seals, and found no striking difference in prey selection between the NWHI and MHI.

Monk seal juvenile survival rates are favorable in the MHI (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 and Johanos 2004). Yet, there are many challenges that may limit the potential for growth in this region. The human population in the MHI is approximately 1.4 million compared to fewer than 100 in the NWHI, such that anthropogenic threats in the MHI are considerable. Intentional killing of seals is a very serious concern. Also, 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 MHI.

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# HARBOR PORPOISE (*Phocoena phocoena*): Morro Bay Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). That study also showed regional differences within California (although the sample size was small). This pattern stands as a sharp contrast to the eastern coast of the U.S. and Canada where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrate that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved. Subsequent genetic analyses of samples ranging from Morro Bay, California to British Columbia indicate that there is small-scale subdivision within the U.S. portion of this range (Chivers *et al.*, 2002, 2007; Morin *et al.*, 2021). Six harbor porpoise stocks have been designated off California/Oregon/Washington, based on genetic analyses and density discontinuities identified from aerial surveys. The stock boundaries in waters off California and southern Oregon are shown in Figure 1. For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) the Morro Bay stock (this report), 2) the Monterey Bay stock, 3) the San Francisco-Russian River stock, 4) the northern California/southern Oregon stock, 5) the northern Oregon/Washington coast stock, and 6) the Inland Washington stock. Three additional Alaskan harbor porpoise stocks are reported separately in the Alaska Stock Assessment Reports.

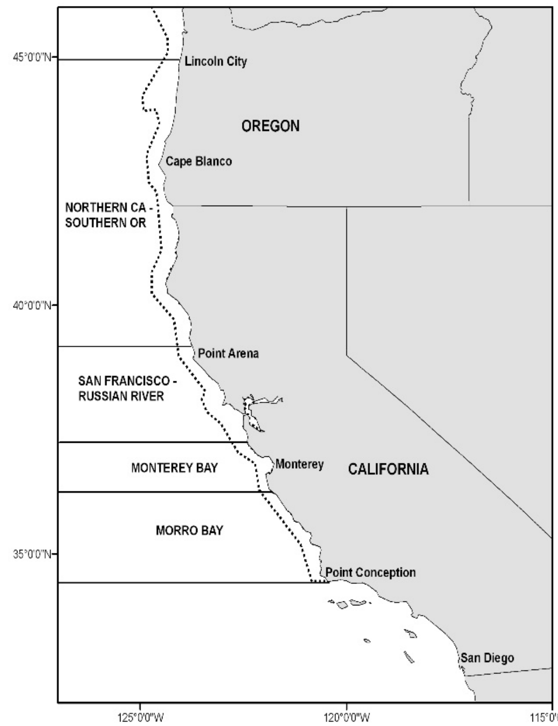


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the California and southern Oregon coasts. Dashed line represents harbor porpoise habitat (0-200 m) in this region.

## POPULATION SIZE

Previous estimates of abundance for California harbor porpoise were based on aerial surveys conducted between the coast and the 50-fm isobath during 1988-95 (Barlow and Forney 1994, Forney 1999). These estimates did not include an unknown number of animals found in deeper waters. Barlow (1988) found that the vast majority of harbor porpoise in California were within the 0-50-fm depth range; however, Green *et al.* (1992) found that 24% of harbor porpoise seen during aerial surveys of Oregon and Washington were between the 100m and 200m isobaths (55 to 109 fathoms). A systematic ship survey of depth strata out to 90 m in northern California showed that porpoise abundance declined significantly in waters deeper than 60 m (Carretta *et al.* 2001). Since 1999, aerial surveys have extended farther offshore (to the 200 m depth contour or a minimum of 10 nmi from shore in the region of the Morro Bay stock) to provide a more complete abundance estimate (Forney *et al.* 2014). A recent analysis of long-term trends in the Morro Bay harbor porpoise population (Forney *et al.* 2020) between 1986 and 2012 estimated a population size of 4,191

(CV=0.561) porpoises during 2012. This estimate includes a correction factor of 3.42 (1/*g*(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers, and it is the most recent estimate available for this stock.

**Minimum Population Estimate**

The minimum population estimate for the Morro Bay harbor porpoise stock is taken as the lower 20th percentile of the log-normal distribution of the abundance estimated from the 2012 aerial surveys, or 2,698 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2012 (Forney *et al.* 2020) showed a marked increase in population size after 1991, when gillnet bycatch was largely eliminated within the range of the Morro Bay stock (Figure 2). This study also concludes that unmonitored harbor porpoise bycatch extending back as far as the 1950s likely decimated this population to a greater extent than previously understood.

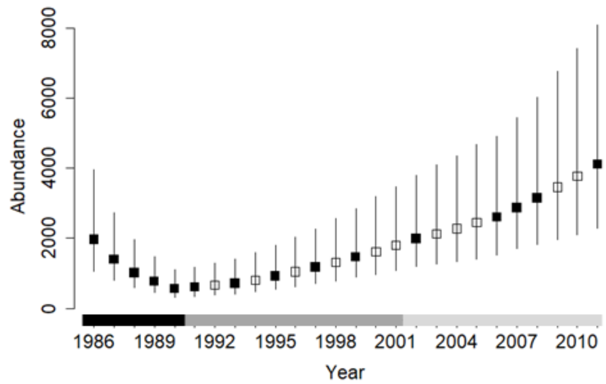


**Figure 2.** Population trends for the Morro Bay harbor porpoise stock, 1986-2012 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along the x-axis reflect the relative level of gillnet bycatch: high (black), low to moderate (dark gray), or none (light gray).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. This estimated growth rate can be considered a maximum net productivity rate, because this stock was estimated to include only 571 porpoises when gillnet bycatch was reduced to low levels in 1991, and by 2012 the population had increased to an estimated 4,191 individuals.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (2,698) times one half the estimated maximum net growth rate for this stock of harbor porpoise (½ of 9.6%) times a recovery factor of 0.5 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of 65.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Gillnet fisheries for halibut and white seabass that historically operated in the vicinity of Morro Bay were eliminated in this stock’s range in 2002 by a ban on gillnets inshore of 60 fathoms (~110 m) from Point Arguello to Point Reyes, California. The large-mesh drift gillnet fishery for swordfish and thresher shark operates too far offshore to interact with harbor porpoise in this region. In the most recent five-year period (2015-2019), no fishery-related strandings of harbor porpoise were documented within this stock’s range (Carretta *et al.* 2021).

**Table 1.** Summary of available data on incidental mortality and serious injury of Morro Bay Stock harbor porpoise in commercial fisheries that might take this species. Mean annual takes are based on 2015-2019 data, Carretta *et al.* (2021). n/a indicates that data are not available.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Kill/Day | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|---|
| Unknown fishery | 2015-2019 | Stranding | n/a | none | n/a | n/a | 0 (n/a) |
| **Minimum total annual takes** | | | | | | | 0 (n/a) |

## STATUS OF STOCK

Harbor porpoise in California are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. Barlow and Hanan (1995) calculate the status of harbor porpoise relative to historic carrying capacity (K) using a technique called back-projection. They calculate that the central California population (including Morro Bay, Monterey Bay, and San Francisco-Russian River stocks) could have been reduced to between 30% and 97% of K by incidental fishing mortality, depending on the choice of input parameters. They conclude that there is no practical way to reduce the range of this estimate. Although Forney *et al*. 2020 documented a marked increase in the Morro Bay harbor porpoise stock, the carrying capacity of this stock is not known and the population status relative to Optimum Sustainable Population (OSP) levels must be treated as unknown.

Because the known human-caused mortality or serious injury (zero harbor porpoise per year) is less than the PBR (65), this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate. Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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# HARBOR PORPOISE (*Phocoena phocoena*): Monterey Bay Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). That study also showed regional differences within California (although the sample size was small). This pattern stands as a sharp contrast to the eastern coast of the U.S. and Canada where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et a*l. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrate that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved. Subsequent genetic analyses of samples ranging from Morro Bay, California to British Columbia indicate that there is small-scale subdivision within the U.S. portion of this range (Chivers *et al*., 2002, 2007; Morin *et al*., 2021). Six harbor porpoise stocks have been designated off California/Oregon/Washington, based on genetic analyses and density discontinuities identified from aerial surveys. The stock boundaries in waters off California and southern Oregon are shown in Figure 1. For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) the Morro Bay stock, 2) the Monterey Bay stock (this report), 3) the San Francisco-Russian River stock, 4) the northern California/southern Oregon stock, 5) the northern Oregon/Washington coast stock, and 6) the Inland Washington stock. Three additional Alaskan harbor porpoise stocks are reported separately in the Alaska Stock Assessment Reports.

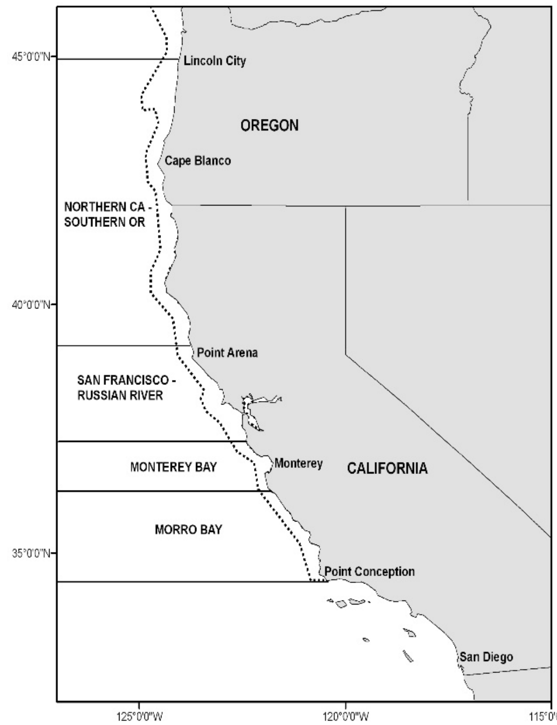


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the California/southern Oregon coast. Dashed line represents harbor porpoise habitat (0-200 m) along the U.S. west coast.

## POPULATION SIZE

Previous estimates of abundance for California harbor porpoise were based on aerial surveys conducted between the coast and the 50-fm isobath during 1988-95 (Barlow and Forney 1994, Forney 1999). These estimates did not include an unknown number of animals found in deeper waters. Barlow (1988) found that the vast majority of harbor porpoise in California were within the 0-50-fm depth range; however, Green *et al*. (1992) found that 24% of harbor porpoise seen during aerial surveys of Oregon and Washington were between the 100m and 200m isobaths (55 to 109 fathoms). A systematic ship survey of depth strata out to 90 m in northern California showed that porpoise abundance declined significantly in waters deeper than 60 m (Carretta *et al*. 2001). Starting in 1999, aerial surveys extended farther offshore (to the 200m depth contour or a minimum of 15 nmi from shore in the region of the Monterey Bay stock) to provide a more complete abundance estimate (Forney *et al.* 2014). A recent analysis of long-term trends in the Monterey Bay harbor porpoise population (Forney *et al.* 2020) between 1986 and 2013 estimated a population size of 3,760 (CV=0.561) porpoises during 2013. This estimate includes a correction factor of 3.42 (1/*g*(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers, and it is the most recent estimate available for this stock.

**Minimum Population Estimate**

The minimum population estimate for the Monterey Bay harbor porpoise stock is taken as the lower 20th percentile of the log-normal distribution of the abundance estimated from the 2013 aerial surveys, or 2,421 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2013 (Forney *et al.* 2020) showed an increase in population size from a low of about 1,500 porpoises in 1987 to more than 3,700 porpoises in 2013 (Forney *et al.* 2020). Most of this increase took place after gillnet fisheries were eliminated within the range of the Monterey Bay stock in 2002 (Figure 2).

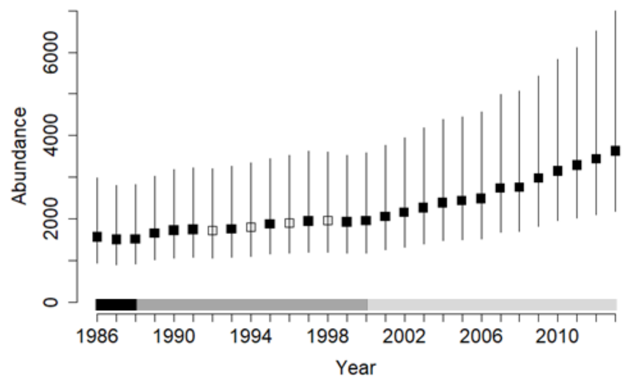


**Figure 2.** Population trends for the Monterey Bay harbor porpoise stock, 1986-2013 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along the x-axis reflect the relative level of gillnet bycatch: high (black), low to moderate (dark gray), or none (light gray).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This maximum theoretical rate represents maximum survival in a protected environment and may not be achievable for any wild population (Barlow and Boveng 1991). Woodley and Read (1991) calculate a maximum growth rate of approximately 5% per year, but their argument for this being a maximum (i.e. that porpoise survival rates cannot exceed those of Himalayan thar) is not well justified. Forney *et al.* (2020) estimated a growth rate of 5.8% per year (95% credible interval: 0% - 12.4%) for the Monterey Bay harbor porpoise stock after gillnet fisheries were eliminated in 2003. Although this growth rate cannot be considered a true maximum net productivity rate, because this stock's status relative to OSP in 2003 was unknown, it is greater than the default maximum net productivity rate ($R_{MAX}$) of 4% for cetaceans (Wade and Angliss 1997) and, therefore, can be considered a minimum estimate of $R_{MAX}$ for this stock.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (2,421) times one half the maximum net growth rate estimated for this stock (½ of 5.8%) times a recovery factor of 0.50 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of 35.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Gillnet fisheries for halibut and white seabass that historically operated in the vicinity of Monterey Bay were eliminated in this stock's range in 2002 by a ban on gillnets inshore of 60 fathoms (~110 m) from Point Arguello to Point Reyes, California. The large-mesh drift gillnet fishery for swordfish and thresher shark operates too far offshore to interact with harbor porpoise in this region. In the most recent five-year period (2015-2019), one fishery-related stranding of harbor porpoise was documented within the range of the Monterey Bay stock (in 2015, Carretta *et al.* 2021). The responsible fishery has not been identified.

**Table 1.** Summary of available on incidental mortality and injury of harbor porpoise in commercial fisheries that might take this species. Mean annual takes are based on 2015-2019 data, Carretta *et al.* (2021). n/a indicates that data are not available.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Kill/Day | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|---|
| Unidentified hook and line fishery | 2013-2017 | Stranding | n/a | 1 | n/a | ≥1 (n/a) | ≥ 0.2 (n/a) |
| Minimum total annual takes | | | | | | | ≥ 0.2 (n/a) |

## STATUS OF STOCK

Harbor porpoise in California are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. Barlow and Hanan (1995) calculate the status of harbor porpoise relative to historic carrying capacity (K) using a technique called back-projection. They calculate that the central California population could have been reduced to between 30% and 97% of K by incidental fishing mortality, depending on the choice of input parameters. They conclude that there is no practical way to reduce the range of this estimate. Although Forney *et al.* (2020) documented a population increase in the Monterey Bay harbor porpoise stock, the carrying capacity of this stock is not known and the population status relative to Optimum Sustainable Population (OSP) levels must be treated as unknown. Because the known human-caused mortality or serious injury (≥ 0.2 harbor porpoise per year) is less than the PBR (35), this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate. Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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# HARBOR PORPOISE (*Phocoena phocoena*): San Francisco-Russian River Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). That study also showed regional differences within California (although the sample size was small). This pattern stands as a sharp contrast to the eastern coast of the U.S. and Canada where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrate that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved. Subsequent genetic analyses of samples ranging from Morro Bay, California to British Columbia indicate that there is small-scale subdivision within the U.S. portion of this range (Chivers *et al.,* 2002, 2007; Morin *et al*., 2021). Six harbor porpoise stocks have been designated off California/Oregon/Washington, based on genetic analyses and density discontinuities identified from aerial surveys. The stock boundaries in waters off California and southern Oregon are shown in Figure 1. For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, Pacific coast harbor porpoise stocks include: 1) the Morro Bay stock (this report) 2) the Monterey Bay stock, 3) the San Francisco-Russian River stock, 4) the northern California/southern Oregon stock, 5) the northern Oregon/Washington coast stock, and 6) the Inland Washington stock. Three additional Alaskan harbor porpoise stocks are reported separately in the Alaska Stock Assessment Reports.

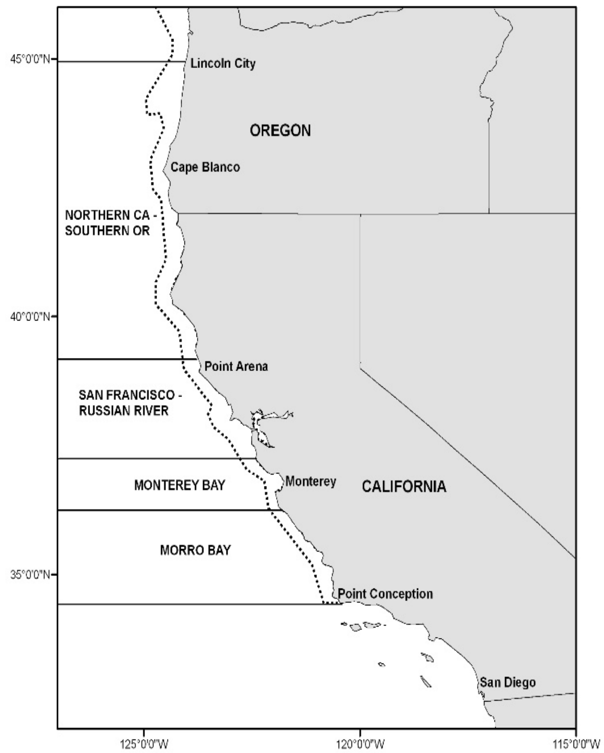


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the California and southern Oregon coasts. Dashed line represents harbor porpoise habitat (0-200 m) along the U.S. west coast.

## POPULATION SIZE

Previous estimates of abundance for California harbor porpoise were based on aerial surveys conducted between the coast and the 50-fm isobath during 1988-95 (Barlow and Forney 1994, Forney 1999). These estimates did not include an unknown number of animals found in deeper waters. Barlow (1988) found that the vast majority of harbor porpoise in California were within the 0-50-fm depth range; however, Green *et al*. (1992) found that 24% of harbor porpoise seen during aerial surveys of Oregon and Washington were between the 100m and 200m isobaths (55 to 109 fathoms). A systematic ship survey of depth strata out to 90 m in northern California showed that porpoise abundance declined significantly in waters deeper than 60 m (Carretta *et al.* 2001). Since 1999, aerial surveys extended farther offshore (to the 200m depth contour or a minimum of 15 nmi from shore in the region of the San Francisco-Russian River stock) to provide a more complete abundance estimate (Forney *et al.* 2014). A recent analysis of long-term trends in the San Francisco-Russian River harbor porpoise stock (Forney *et al.* 2020) between 1986 and 2017 estimated a population size of 7,777 (CV=0.620) porpoises during 2017. This estimate includes a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers, and it is the most recent estimate available for this stock.

**Minimum Population Estimate**

The minimum population estimate for the San Francisco-Russian River harbor porpoise stock is taken as the lower 20th percentile of the log-normal distribution of the abundance estimated from 1986 to 2017 aerial surveys, or 4,811 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends between 1986 and 2017 (Forney *et al.* 2020) showed an increase in population size following the elimination of gillnets from the range of the San Francisco – Russian River stock in 1987 (Forney *et al.* 2019). The population size peaked in 2005 at about 13,500 porpoises, and subsequently appeared to drop, leveling off at about 7,000-8,000 porpoises during 2010-2017 (Figure 2). Forney *et al.* (2020) noted that the apparent decrease after 2005 could be artefact of the large uncertainty in the abundance estimates during 2002-2007. Alternately, a shift in the distribution of harbor porpoise in this region, including a re-colonization of waters inside San Francisco Bay documented in 2009 (Stern *et al.* 2017), might have reduced their detectability during aerial surveys along the outer coast.

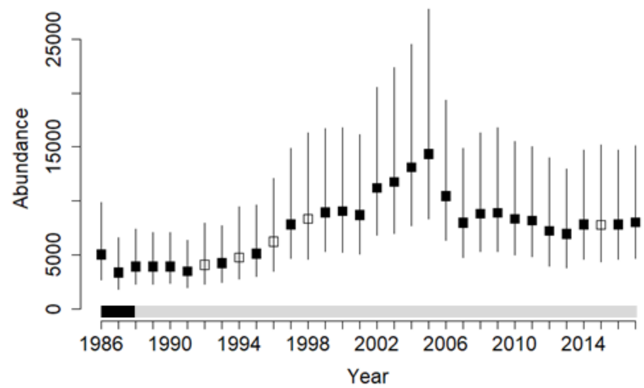


**Figure 2.** Population trends for the San Francisco-Russian River harbor porpoise stock, 1986-2017 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols). Shaded bars along the X-axis reflect the relative level of gillnet bycatch: high (black), or none (light gray).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This maximum theoretical rate represents maximum survival in a protected environment and may not be achievable for any wild population (Barlow and Boveng 1991). Woodley and Read (1991) calculate a maximum growth rate of approximately 5% per year, but their argument for this being a maximum (i.e. that porpoise survival rates cannot exceed those of Himalayan thar) is not well justified. Forney *et al.* (2020) estimated a growth rate of 6.1% per year (SE = 2.1%) for the San Francisco – Russian River harbor porpoise stock after gillnet fisheries were eliminated in 1987 until the population peaked and then leveled off in 2005. Although this growth rate cannot be considered a true maximum net productivity rate, because this stock's status relative to OSP in 1987 was unknown, it is greater than the default maximum net productivity rate ($R_{MAX}$) of 4% for cetaceans (Wade and Angliss 1997) and, therefore, can be considered a minimum estimate of $R_{MAX}$ for this stock.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (4,811) times one half the maximum net growth rate estimated for this stock (½ of 6.1%) times a recovery factor of 0.5 (for a stock of unknown status; Wade and Angliss 1997), resulting in a PBR of 73.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Although coastal gillnets are prohibited throughout this stock's range, there have been fishery-related strandings in past years. In the most recent five-year period (2015-2019), three fishery-related strandings of harbor porpoise were documented within the range of the San Francisco-Russian River stock (in 2015 and 2018; Carretta *et al.* 2021). Unidentified net fisheries were considered responsible for both porpoise deaths.

**Table 1.** Summary of available information on incidental mortality and injury of harbor porpoise (San Francisco-Russian River stock) in commercial fisheries that might take this species. Mean annual takes are based on 2015-2019 data, (Carretta *et al.* 2021). n/a indicates that data are not available.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Kill/Day | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|---|
| **Unknown net fishery** | 2015-2019 | stranding | n/a | 2 | n/a | ≥ 2 (n/a) | ≥ 0.4(n/a) |
| **Minimum total annual takes** | | | | | | | ≥ 0.4 (n/a) |

## STATUS OF STOCK

Harbor porpoise in California are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. Barlow and Hanan (1995) calculate the status of harbor porpoise relative to historic carrying capacity (K) using a technique called back-projection. They calculate that the central California population (including Morro Bay, Monterey Bay, and San Francisco-Russian River stocks) could have been reduced to between 30% and 97% of K by incidental fishing mortality, depending on the choice of input parameters. They conclude that there is no practical way to reduce the range of this estimate. Although Forney *et al.* (2020) documented a population increase in the San Francisco – Russian River harbor porpoise stock, the carrying capacity of this stock is not known, and the population status relative to Optimum Sustainable Population (OSP) levels must be treated as unknown. Because the known human-caused mortality or serious injury (≥ 0.4 harbor porpoise per year) is less than the PBR (73), this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate. Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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Revised 05/31/2024

# HARBOR PORPOISE (*Phocoena phocoena*): Northern California/Southern Oregon Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). That study also showed regional differences within California (although the sample size was small). This pattern is a sharp contrast with the eastern coast of the U.S. and Canada where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrated that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved.

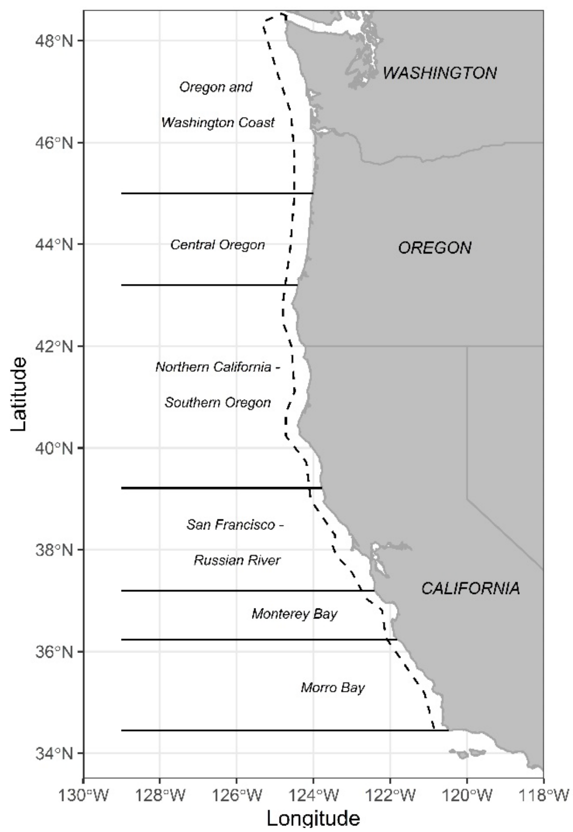


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth).

Significant genetic differences have been identified for harbor porpoises along the outer U.S. West Coast and in inland waters of Washington (Chivers *et al*. 2002, 2007; Morin *et al*. 2021), leading to the designation of multiple stocks in this region. The most recent study (Morin *et al.* 2021) identified additional genetic differences between porpoises found off central and southern Oregon, and suggested that a new stock boundary was warranted at approximately 43.2°N latitude. Based on these findings, the northern boundary of the Northern California – Southern Oregon stock has been moved south to 43.2°N, and a new central Oregon stock has been designated between 43.2°N and 45°N (Figure 1).

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, the following harbor porpoise stocks are designated in the Pacific Ocean: 1) Morro Bay, 2) Monterey Bay, 3) San Francisco-Russian River, 4) Northern California/Southern Oregon, 5) Central Oregon, 6) Northern Oregon/Washington Coast, 7) Washington Inland Waters, 8) Northern Southeast Alaska Inland Waters, 9) Southern Southeast Alaska Inland Waters, 10) Yakutat/Southeast Alaska Offshore Waters, 11) Gulf of Alaska, and 12) Bering Sea. Reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

## POPULATION SIZE

Previous estimates of abundance for California harbor porpoise were based on aerial surveys conducted between the coast and the 50-fm isobath during 1988-95 (Barlow and Forney 1994, Forney 1999). These estimates did not include an unknown number of animals found in deeper waters. Barlow (1988) found that the vast majority of harbor porpoise in California were within the 0-50 fm depth range; however, Green et al. (1992) found that 24% of harbor porpoise seen during aerial surveys of Oregon and Washington were between the 100m and 200m isobaths (55 to 109 fathoms). A systematic ship survey of depth strata out to 90 m in northern California showed that porpoise abundance declined significantly in waters deeper than 60 m (Carretta *et al*. 2001). Since 1999, aerial surveys extended farther offshore (to the 200m depth contour or 15 nmi distance, whichever is farther) to provide a more complete abundance estimate (Forney *et al.* 2014). An analysis of long-term trends in the northern California portion of this harbor porpoise stock between 1989 and 2016 (Forney *et al.* 2020) estimated a northern California population size of 12,160 (CV=0.663) porpoises during 2016. More recently, Forney *et al.* (2023) estimated the abundance of harbor porpoise within the Oregon range of this stock (south of 43.2°N) to be 3,143 (CV = 0.464) based on a habitat-based density model developed from 2021-2022 aerial surveys off Oregon and Washington. Both of these estimates include a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers. Combining these two abundance estimates yields an overall abundance estimate of 15,303 (CV = 0.575) for the entire northern California/southern Oregon stock (Forney *et al.* 2023).

**Minimum Population Estimate**

The minimum population estimate for harbor porpoise in the northern California/southern Oregon stock is calculated as the lower 20th percentile of the log-normal distribution of the abundance estimate given above, or 9,759 animals.

**Current Population Trend**

A hierarchical Bayesian analysis of harbor porpoise trends for the northern California portion of this stock between 1989 and 2016 (Forney *et al.* 2020) suggests largely stable population during this period, although there is considerable uncertainty in the estimates because of limited survey coverage (Figure 2). No trend estimates are available for the entire northern California/southern Oregon range of this stock.

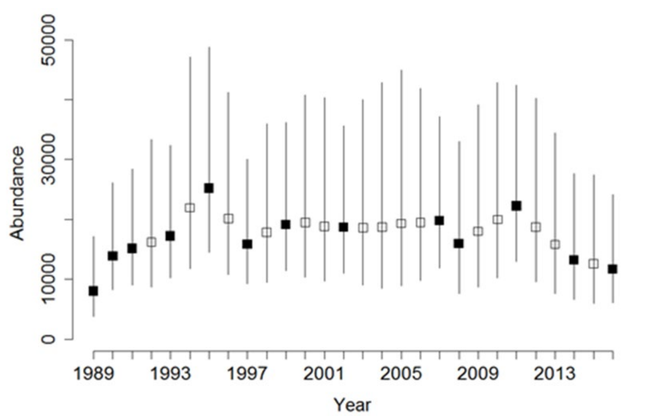


**Figure 2.** Population trends for the northern California portion of the Northern California / Southern Oregon harbor porpoise stock, 1989-2016 (from Forney *et al.* 2020). Estimates represent median abundance (with 95% credible intervals) for years with survey effort (solid symbols) and without survey effort (open symbols).

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. Because a reliable estimate of the maximum net productivity rate is not available for the Northern California / Southern Oregon harbor porpoise stock, we use the default maximum net productivity rate ($R_{MAX}$) of 4% for cetaceans (Wade and Angliss 1997).

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (9,759) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 1.0 (for a species within its Optimal Sustainable Population; see Status of Stock section; Wade and Angliss 1997), resulting in a PBR of 195.

## HUMAN-CAUSED MORTALITY

### Fishery Information

There were no harbor porpoise strandings in this stock's range with evidence of fishery interactions during 2017-2021 (Carretta *et al.* 2023).

**Table 1.** Summary of available information on incidental mortality and injury of harbor porpoise (northern California/southern Oregon stock) in commercial fisheries that might take this species during 2017-2021 (Carretta *et al.* 2023). n/a indicates that data are not available.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **Unknown fishery** | 2017-2021 | Stranding | - | none | n/a | 0 (n/a) |
| **Minimum total annual takes** | | | | | | 0 (n/a) |

## STATUS OF STOCK

Harbor porpoise in northern California/southern Oregon are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. The northern California portion of this harbor porpoise stock was determined to be within their Optimum Sustainable Population (OSP) level in the mid-1990s (Barlow and Forney 1994), based on a lack of significant anthropogenic mortality. Because there is no known human-caused mortality or serious injury, this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate.

## OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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Revised 05/31/2024

# HARBOR PORPOISE (*Phocoena phocoena*): Central Oregon Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). This pattern is a sharp contrast with the eastern coast of the U.S. and Canada, where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrated that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved.

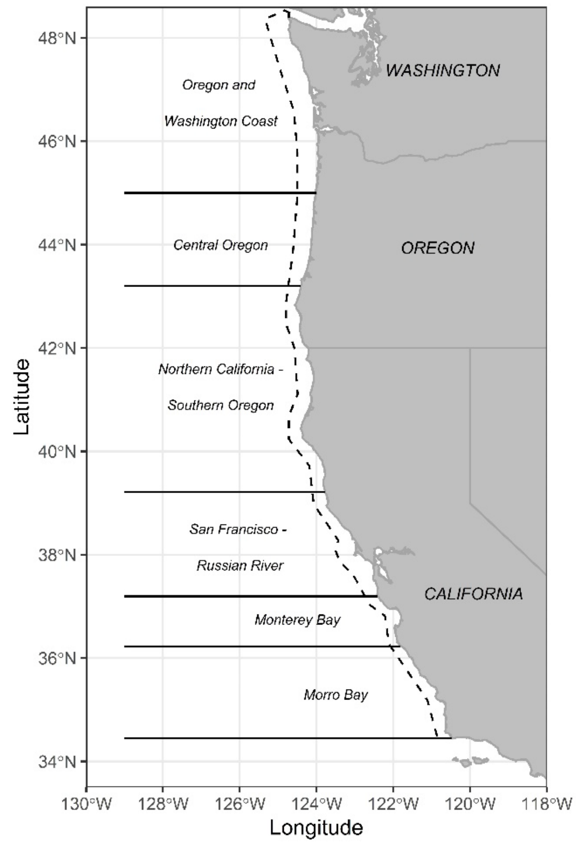


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the outer U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth).

Significant genetic differences have been identified for harbor porpoises along the outer U.S. West Coast and in inland waters of Washington (Chivers *et al*. 2002, 2007; Morin *et al*. 2021), leading to the designation of multiple stocks in this region. The most recent study (Morin *et al.* 2021) identified additional genetic differences between porpoises found off central and southern Oregon, and suggested that a new stock boundary was warranted at approximately 43.2°N latitude. Based on these findings, the northern boundary of the Northern California – Southern Oregon stock has been moved south to 43.2°N, and a new central Oregon stock has been designated between 43.2°N and 45°N (Figure 1).

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, the following harbor porpoise stocks are designated in the Pacific Ocean: 1) Morro Bay, 2) Monterey Bay, 3) San Francisco-Russian River, 4) Northern California/Southern Oregon, 5) Central Oregon, 6) Northern Oregon/Washington Coast, 7) Washington Inland Waters, 8) Northern Southeast Alaska Inland Waters, 9) Southern Southeast Alaska Inland Waters, 10) Yakutat/Southeast Alaska Offshore Waters, 11) Gulf of Alaska, and 12) Bering Sea. Reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

## POPULATION SIZE

Aerial surveys off Oregon were previously conducted during 2010-2011 (Forney *et al.* 2014); however, the abundance estimate presented in that study was for a larger area than the central Oregon stock range. More recently, Forney *et al.* (2023) estimated the abundance of harbor porpoise within shelf waters (0-200m water depth) of the central Oregon stock range to be 7,492 (CV = 0.421) based on a habitat-based density model developed from 2021-2022 aerial surveys off Oregon and Washington. This estimate includes a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers.

### Minimum Population Estimate

The minimum population estimate for harbor porpoise in the central Oregon stock is calculated as the lower 20th percentile of the log-normal distribution of the above abundance estimate, or 5,332 animals.

### Current Population Trend

There are no reliable data on population trends of harbor porpoise for coastal Oregon; however, the sum of the abundance estimates reported in Forney *et al.* (2023) for southern Oregon (3,143; CV=0.464) and central Oregon (7,492; CV=0.421), equal to 10,635 individuals, falls within the confidence limit of the previous abundance estimate of 12,525 (CV=0.48) reported for that region in Forney *et al.* (2014) based on 2010-2011 aerial surveys.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. Because a reliable estimate of the maximum net productivity rate is not available for the central Oregon harbor porpoise stock, we use the default maximum net productivity rate ($R_{MAX}$) of 4% for cetaceans (Wade and Angliss 1997).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (5,332) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status, Wade and Angliss 1997), resulting in a PBR of 53.

## HUMAN-CAUSED MORTALITY

### Fishery Information

There were no harbor porpoise strandings in this stock's range with evidence of fishery interactions during 2017-2021 (Carretta *et al.* 2023).

**Table 1.** Summary of available information on incidental mortality and injury of harbor porpoise (central Oregon stock) in commercial fisheries that might take this species during 2017-2021 (Carretta *et al.* 2023). n/a indicates that data are not available.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed Mortality | Estimated Mortality (CV in parentheses) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **Unknown fishery** | 2017-2021 | Stranding | - | none | n/a | 0 (n/a) |
| **Minimum total annual takes** | | | | | | 0 (n/a) |

## STATUS OF STOCK

Harbor porpoise in central Oregon are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. The status of this stock relative to its Optimum Sustainable Population (OSP) level and population trends is unknown. Because there is no known human-caused mortality or serious injury, this stock is not considered a "strategic" stock under the MMPA, and fishery mortality can be considered insignificant and approaching zero mortality and serious injury rate.

## OTHER FACTORS THAT MAY BE AFFECTING THE STOCK

Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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Revised 05/31/2024

# HARBOR PORPOISE (*Phocoena phocoena*): Northern Oregon/Washington Coast Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the Pacific, harbor porpoise are found in coastal and inland waters from Point Conception, California to Alaska and across to Kamchatka and Japan (Gaskin 1984). Harbor porpoise appear to have more restricted movements along the western coast of the continental U.S. than along the eastern coast. Regional differences in pollutant residues in harbor porpoise indicate that they do not move extensively between California, Oregon, and Washington (Calambokidis and Barlow 1991). This pattern is a sharp contrast with the eastern coast of the U.S. and Canada, where harbor porpoise are believed to migrate seasonally from as far south as the Carolinas to the Gulf of Maine and Bay of Fundy (Polacheck *et al*. 1995). A phylogeographic analysis of genetic data from northeast Pacific harbor porpoise did not show complete concordance between DNA sequence types and geographic location (Rosel 1992). However, an analysis of molecular variance (AMOVA) of the same data with additional samples found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel *et al*. 1995). These results demonstrated that harbor porpoise along the west coast of North America are not panmictic or migratory, and movement is sufficiently restricted that genetic differences have evolved.

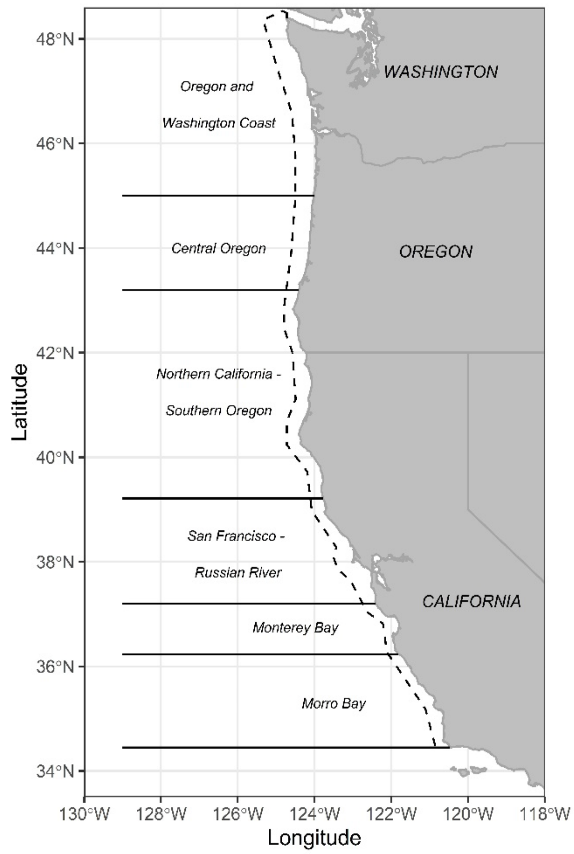


**Figure 1.** Stock boundaries and distributional range of harbor porpoise along the outer U.S. West Coast. Dashed line represents an approximate boundary for harbor porpoise habitat (0-200m water depth).

Significant genetic differences have been identified for harbor porpoises along the outer U.S. West Coast and in inland waters of Washington (Chivers *et al*. 2002, 2007; Morin *et al*. 2021), leading to the designation of multiple stocks in this region. The most recent study (Morin *et al.* 2021) identified additional genetic differences between porpoises found off central and southern Oregon, and suggested that a new stock boundary was warranted at approximately 43.2°N latitude. Based on these findings, the northern boundary of the Northern California – Southern Oregon stock has been moved south to 43.2°N, and a new central Oregon stock has been designated between 43.2°N and 45°N (Figure 1).

For the Marine Mammal Protection Act (MMPA) Stock Assessment Reports, the following harbor porpoise stocks are designated in the Pacific Ocean: 1) Morro Bay, 2) Monterey Bay, 3) San Francisco-Russian River, 4) Northern California/Southern Oregon, 5) Central Oregon, 6) Northern Oregon/Washington Coast, 7) Washington Inland Waters, 8) Northern Southeast Alaska Inland Waters, 9) Southern Southeast Alaska Inland Waters, 10) Yakutat/Southeast Alaska Offshore Waters, 11) Gulf of Alaska, and 12) Bering Sea. Reports for harbor porpoise stocks within waters of California, Oregon, and Washington appear in this volume. The five Alaska harbor porpoise stocks are reported separately in the Stock Assessment Reports for the Alaska Region.

## POPULATION SIZE

Aerial surveys were previously conducted off Oregon and Washington during 2010-2011 (Forney *et al.* 2014), yielding an abundance estimate for the northern Oregon/Washington coast stock of 21,487 (CV=0.44). More recently, Forney *et al.* (2023) estimated a similar abundance of 22,074 (CV = 0.391) harbor porpoise within shelf waters (0-200m water depth) of this stock's range based on a habitat-based density model developed from 2021-2022 aerial surveys off Oregon and Washington. Both estimates included a correction factor of 3.42 (1/g(0); g(0)=0.292, CV=0.366) (Laake *et al*. 1997) to adjust for groups missed by aerial observers.

### Minimum Population Estimate

The minimum population estimate for this stock is calculated as the lower 20th percentile of the log-normal distribution of the 2021-2022 abundance estimate, or 16,068 animals.

### Current Population Trend

There are no reliable data on population trends of harbor porpoise for coastal Oregon and Washington; however, the 2010-2011 (Forney *et al.* 2014) and 2021-22 (Forney *et al*. 2023) abundance estimates are very similar.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on what are argued to be biological limits of the species (i.e. females give birth first at age 4 and produce one calf per year until death), the theoretical, maximum-conceivable growth rate of a closed harbor porpoise population was estimated as 9.4% per year based on a human survivorship curve (Barlow and Boveng 1991). This is very similar to the growth rate of 9.6% per year (95% credible interval: 6.2% - 13.0%) estimated by Forney *et al.* (2020) for the Morro Bay harbor porpoise stock between 1991 and 2012, based on long-term aerial surveys. Because a reliable estimate of the maximum net productivity rate is not available for the Northern Oregon / Washington Coast harbor porpoise stock, we use the default maximum net productivity rate ($R_{MAX}$) of 4% for cetaceans (Wade and Angliss 1997).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (16,068) times one-half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status, Wade and Angliss 1997), resulting in a PBR of 161.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

There were 16 strandings or reports of fishery-related mortality and serious injury of harbor porpoise within the range of the northern Oregon/Washington coast stock during 2017-2021 (Carretta *et al*. 2023), resulting in a mean annual mortality of at least 3.2 harbor porpoise (Table 1). Stranding numbers are considered a minimum because not all stranded animals are found, reported, or examined for cause of death (via necropsy by trained personnel). Interactions with tribal fisheries are derived from annual reports submitted by the Northwest Indian Fisheries Commission (NWIFC) to NMFS.

**Table 1.** Summary of incidental mortality and serious injury of harbor porpoise (Northern Oregon/Washington Coast stock) in commercial and tribal fisheries that might take this species and calculation of the mean annual mortality rate; n/a indicates that data are not available. Mean annual takes are based on 2017-2021 data unless noted otherwise.

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Gillnet fishery (tribal)[1] | 2017-2021 | Fishery self-report | n/a | 0, 0, 2, 0, 0 | | >0.4 (n/a) |
| Unidentified gillnet fishery | 2017-2021 | stranding | n/a | 0, 3, 1, 0, 1 | | >1.0 (n/a) |
| Unidentified fishery | 2007-2011 | stranding | ~~n/a~~ | 8, 0, 0, 0, 1 | n/a | >1.8 (n/a) |

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| Minimum total annual takes | | | | | | >3.2 (n/a) |

[1]This is a tribal fishery; therefore, it is not listed in the NMFS list of commercial fisheries.

## STATUS OF STOCK

Harbor porpoise along the outer coast of northern Oregon and Washington are not listed as threatened or endangered under the Endangered Species Act nor as depleted under the Marine Mammal Protection Act. The status of this stock relative to its Optimum Sustainable Population (OSP) level and population trends is unknown. Based on currently available data, the minimum annual level of total human-caused mortality and serious injury (3.2 per year) does not exceed the PBR (161). Therefore, the Northern Oregon/Washington Coast stock of harbor porpoise is not classified as a "strategic"stock under the MMPA. The minimum annual fishery mortality and serious injury for this stock (3.2) is not known to exceed 10% of the calculated PBR (16.1) and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate.

## OTHER FACTORS THAT MAY AFFECTING THE STOCK

Harbor porpoises are sensitive to disturbance by a variety of anthropogenic sound sources, and the limited range of several U.S. West Coast harbor porpoise stocks makes them particularly vulnerable to potential impacts (see overview in Forney *et al*. 2017). A recent habitat concern along the U.S. West coast includes the use of acoustic deterrent devices ('seal bombs') that are used in commercial fishing activities off California (Simonis *et al*. 2020), especially in the Monterey Bay region.

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Revised 2/6/2017

# HARBOR PORPOISE (*Phocoena phocoena*): Washington Inland Waters Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

In the eastern North Pacific Ocean, harbor porpoise are found in coastal and inland waters from Point Barrow, along the Alaskan coast, and down the west coast of North America to Point Conception, California (Gaskin 1984). Harbor porpoise are known to occur year-round in the inland trans-boundary waters of Washington and British Columbia, Canada (Osborne et al. 1988), and along the Oregon/Washington coast (Barlow 1988, Barlow et al. 1988, Green et al. 1992). Aerial survey data from coastal Oregon and Washington, collected during all seasons, suggest that harbor porpoise distribution varies by depth (Green et al. 1992). Although distinct seasonal changes in abundance along the west coast have been noted, and attributed to possible shifts in distribution to deeper offshore waters during late winter (Dohl et al. 1983, Barlow 1988), seasonal movement patterns are not fully understood.

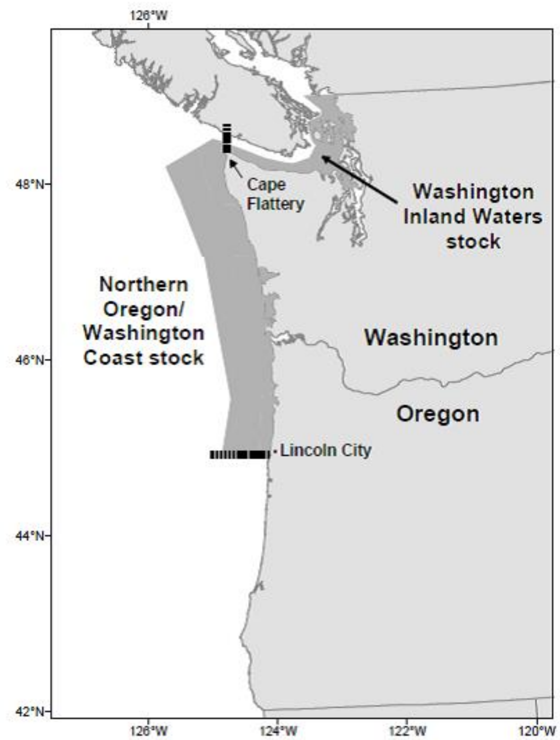


**Figure 1.** Stock boundaries (dashed lines) and approximate distribution (dark shaded areas) of harbor porpoise along the coasts of Washington and northern Oregon. The range of the Northern California/Southern Oregon stock of harbor porpoise (not shown), extends from Lincoln City. OR, south to Pt. Arena, CA.

Investigation of pollutant loads in harbor porpoise ranging from California to the Canadian border suggests restricted harbor porpoise movements (Calambokidis and Barlow 1991). Stock discreteness in the eastern North Pacific was analyzed using mitochondrial DNA from samples collected along the west coast (Rosel 1992) and is summarized in Osmek et al. (1994). Two distinct mtDNA groupings or clades exist. One clade is present in California, Washington, British Columbia, and Alaska (no samples were available from Oregon), while the other is found only in California and Washington. Although these two clades are not geographically distinct by latitude, the results may indicate a low mixing rate for harbor porpoise along the west coast of North America. Further genetic testing of the same data, along with additional samples, found significant genetic differences for four of the six pair-wise comparisons between the four areas investigated: California, Washington, British Columbia, and Alaska (Rosel et al. 1995). These results demonstrate that harbor porpoise along the west coast of North America are not panmictic or migratory and that movement is sufficiently restricted that genetic differences have evolved. Subsequent genetic analyses of samples ranging from Monterey Bay, California, to Vancouver Island, British Columbia, indicate that there is small-scale subdivision within the U.S. portion of this range (Chivers et al. 2002, 2007). This is consistent with low movement suggested by genetic analysis of harbor porpoise specimens from the North Atlantic, where numerous stocks have been delineated with clinal differences over areas as small as the waters surrounding the British Isles.

Using the 1990-1991 aerial survey data of Calambokidis et al. (1993) for water depths <50 fathoms, Osmek et al. (1996) found significant differences in harbor porpoise mean densities ($Z=6.9$, $P<0.001$) between the waters of coastal Oregon/Washington and inland Washington/southern British Columbia, Canada (i.e., Strait of Juan de Fuca/San Juan Islands). Following a risk averse management strategy, two stocks were recognized in the waters of Oregon and Washington, with a boundary at Cape Flattery, Washington. Based on more recent genetic evidence, which suggests that the population of eastern North Pacific harbor porpoise is more finely structured (Chivers et al. 2002, 2007), stock boundaries on the Oregon/Washington coast have been revised, resulting in three stocks in Oregon/Washington waters: a Northern California/Southern Oregon stock (Point Arena, CA, to Lincoln City, OR), a

Northern Oregon/Washington Coast stock (Lincoln City, OR, to Cape Flattery, WA), and the Washington Inland Waters stock (in waters east of Cape Flattery). Additional analyses are needed to determine whether to adjust the stock boundaries for harbor porpoise in Washington inland waters (Chivers et al. 2007).

Barlow and Hanan (1995) recommended two stocks of harbor porpoise be recognized in California, with the stock boundary at the Russian River. Based on more recent genetic findings (Chivers et al. 2002, 2007), California coast stocks were re-evaluated and significant genetic differences were found among four identified sampling sites. Revised stock boundaries, based on these genetic data and density discontinuities identified from aerial surveys, resulted in six California/Oregon/Washington stocks where previously there had been four (e.g., Carretta et al. 2001): 1) the Washington Inland Waters stock, 2) the Northern Oregon/Washington Coast stock, 3) the Northern California/Southern Oregon stock, 4) the San Francisco-Russian River stock, 5) the Monterey Bay stock, and 6) the Morro Bay stock. The stock boundaries for animals that occur in northern Oregon/Washington waters are shown in Figure 1. This report considers only the Washington Inland Waters stock. Stock assessment reports for Northern Oregon/Washington Coast, Northern California/Southern Oregon, San Francisco-Russian River, Monterey Bay, and Morro Bay harbor porpoise also appear in this volume. Stock assessment reports for the three harbor porpoise stocks in the inland and coastal waters of Alaska, including 1) the Southeast Alaska stock, 2) the Gulf of Alaska stock, and 3) the Bering Sea stock, are reported separately in the Stock Assessment Reports for the Alaska Region. The harbor porpoise occurring in British Columbia have not been included in any of the U.S. stock assessment reports.

## POPULATION SIZE

Aerial surveys of the inside waters of Washington and southern British Columbia were conducted from 2013 to 2015 (Smultea *et al.* 2015a, 2015b). These aerial surveys included the Strait of Juan de Fuca, San Juan Islands, Gulf Islands, Strait of Georgia, Puget Sound, and Hood Canal. These are the waters inhabited by the Washington Inland Waters stock of harbor porpoise as well as harbor porpoise from British Columbia. Harbor porpoise abundance estimates were corrected for trackline animals missed by aerial observers using *g(0)* from prior studies in the same area and using similar methods (Laake *et al*. 1997). For U.S. waters, the current estimate of abundance is 11,233 porpoise (CV=0.37) (Smultea *et al*. 2015a).

### Minimum Population Estimate

The minimum population estimate for the Washington Inland Waters stock of harbor porpoise is calculated as the lower 20$^{th}$ percentile of the log-normal distribution (Wade and Angliss 1997) of the 2015 population estimate of 11,233 harbor porpoise, or 8,308 animals.

### Current Population Trend

Estimates of population size for Washington Inland waters from 1990-1991 aerial surveys were 3,298 (CV=0.26) animals, corrected for diving animals not seen by observers (Calambokidis *et al*. 1993). Estimates of harbor porpoise abundance for the same region from 2013-2015 surveys (11,233; CV=0.37, Smultea *et al.* 2015a), are considerably higher, however a formal trend analysis has not been performed for this stock.

In southern Puget Sound, harbor porpoise were common in the 1940s (Scheffer and Slipp 1948), but marine mammal surveys (Everitt et al. 1980), stranding records since the early 1970s (Osmek et al. 1995), and harbor porpoise surveys in 1991 (Calambokidis et al. 1992) and 1994 (Osmek et al. 1995) indicated that harbor porpoise abundance had declined in southern Puget Sound. In 1994, a total of 769 km of vessel survey effort and 492 km of aerial survey effort conducted during favorable sighting conditions produced no sightings of harbor porpoise in southern Puget Sound. Reasons for the apparent decline are unknown, but it may have been related to fishery interactions, pollutants, vessel traffic, or other factors (Osmek et al. 1995). Annual winter aerial surveys conducted by the Washington Department of Fish and Wildlife from 1995 to 2015 revealed an increasing trend in harbor porpoise in Washington inland waters, including the return of harbor porpoise to Puget Sound. The data suggest that harbor porpoise were already present in Juan de Fuca, Georgia Straits, and the San Juan Islands from the mid-1990s to mid-2000s, and then expanded into Puget Sound and Hood Canal from the mid-2000s to 2015, areas they had used historically but abandoned. Changes in fishery-related entanglement was suspected as the cause of their previous decline and more recent recovery, including a return to Puget Sound (Evenson *et al*. 2016). Seasonal surveys conducted in spring, summer, and fall 2013-2015 in Puget Sound and Hood Canal documented substantial numbers of harbor porpoise in Puget Sound. Observed porpoise numbers were twice as high in spring as in fall or summer, indicating a seasonal shift in distribution of harbor porpoise (Smultea 2015b). The reasons for the seasonal shift and for the increase in sightings is unknown.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is not available for harbor porpoise. Therefore, until additional data become available, it is recommended that the cetacean maximum theoretical net productivity rate ($R_{MAX}$) of 4% (Wade and Angliss 1997) be employed for the Washington Inland Waters harbor porpoise stock.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) for this stock is calculated as the minimum population size (8,308) times one-half the default maximum net growth rate for cetaceans (1/2 of 4%) times a recovery factor of 0.4 (for a stock of unknown status and high uncertainty in the mortality and injury estimate), resulting in a PBR of 66 harbor porpoise per year. Although no CV is available for the mortality and serious injury estimate, there is large uncertainty because the available data are limited to stranding information, which is known to have a substantial downward bias (Carretta *et al*. 2016a, Williams *et al.* 2014). For this reason, the recovery factor was set equal to the value for a stock of unknown status with mortality and serious injury CV > 0.80 (Wade and Angliss 1997).

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

Fishing effort in the northern Washington marine gillnet tribal fishery is conducted within the range of both harbor porpoise stocks (Northern Oregon/Washington Coast and Washington Inland Waters) occurring in Washington State waters (Gearin et al. 1994). Some movement of harbor porpoise between Washington's coastal and inland waters is likely, but it is currently not possible to quantify the extent of such movements. For the purposes of this stock assessment report, animals taken in waters east of Cape Flattery, WA, are assumed to have belonged to the Washington Inland Waters stock. Between 2010 and 2014, no harbor porpoise deaths or serious injuries were reported in this fishery (Makah Fisheries Management, unpublished data).

**Table 1.** Summary of incidental mortality and serious injury of harbor porpoise (Washington Inland Waters stock) in commercial and tribal fisheries that might take this species and calculation of the mean annual mortality rate; n/a indicates that data are not available. Mean annual takes are based on 2010-2014 data unless noted otherwise.

| Fishery name | Years | Data type | Percent observer coverage | Observed mortality | Estimated mortality | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| WA Puget Sound Region salmon set/drift gillnet (observer programs listed below covered segments of this fishery): | | | | | | |
| Puget Sound non-treaty salmon gillnet (all areas and species) | 1993 | observer data | 1.3% | 0 | 0 | see text[1] |
| Puget Sound non-treaty chum salmon gillnet (areas 10/11 and 12/12B) | 1994 | observer data | 11% | 0 | 0 | see text[1] |
| Puget Sound treaty chum salmon gillnet (areas 12, 12B, and 12C) | 1994 | observer data | 2.2% | 0 | 0 | see text[1] |
| Puget Sound treaty chum and sockeye salmon gillnet (areas 4B, 5, and 6C) | 1994 | observer data | 7.5% | 0 | 0 | see text[1] |
| Puget Sound treaty and non-treaty sockeye salmon gillnet (areas 7 and 7A) | 1994 | observer data | 7% | 1 | 15 | see text[1] |
| Unknown Puget Sound Region fishery | 2010-2014 | stranding data | | 2, 0, 7, 1, 2 | n/a | ≥ 2.4 (n/a) |
| Minimum total annual takes | | | | | | ≥2.4 (n/a) |

[1]This fishery has not been observed since 1994 (see text); these data are not included in the calculation of recent minimum total annual takes.

Commercial salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994, with observer coverage levels typically <10% (Pierce et al. 1994, 1996; NWIFC 1995; Erstad et al. 1996). Drift gillnet fishing effort in the inland waters has declined considerably since 1994 because far fewer vessels participate today (NMFS WC Region, unpublished data), but entanglements of harbor porpoise likely continue to occur. The most recent data on harbor porpoise mortality from commercial gillnet fisheries is included in Table 1.

Strandings of dead or seriously injured harbor porpoise entangled in fishing gear are another source of fishery-related mortality. There were 12 fishery-related strandings of harbor porpoise from this stock in 2010-2014 (2 in 2010, 7 in 2012, 1 in 2013, and 2 in 2014), resulting in an average annual mortality and serious injury rate of 2.4 harbor porpoise per year (Carretta *et al*. 2016b). Evidence of fishery interactions included observed entanglements, net marks, and line marks. Since these deaths could not be attributed to a particular fishery, and were the only confirmed fishery-related deaths in this area in 2010-2014, they are listed in Table 1 as occurring in an unknown Puget Sound Region fishery. There are no observed fisheries in Washington inland waters, and the estimate of human-caused mortality of harbor porpoise (2.4/yr) is based solely on stranding data, which are uncorrected for negative biases in cetacean carcass recovery (Williams *et al.* 2014). The only published carcass recovery rate for harbor porpoise (<0.01) is from an oceanic-coast habitat in the NE United States (Moore and Read 2008), but due to the confined nature of inland waterways, recovery rates in Washington State inland waters are likely higher than that estimated by Moore and Read (2008). Wells *et al*. (2015) reported a carcass recovery rate (0.33) for bottlenose dolphins that inhabit the densely populated Sarasota Bay area. If this recovery rate of 0.33 is applied to Washington Inland Waters harbor porpoise fishery-related strandings for the period 2010-2014, annual mortality would be estimated at 7.2 (12 documented fishery-related strandings, times a correction factor of 3, divided by 5 years), which is less than the PBR of 66. In the absence of a carcass recovery correction factor for Washington inland waters harbor porpoise, a minimum correction factor of 3 from the Wells *et al*. (2015) coastal bottlenose dolphin study is applied to fishery-related strandings here, resulting in an estimate of 7.2 porpoise annually. Additional data are required to estimate a carcass recovery rate for harbor porpoise in Washington inland waters.

Although commercial gillnet fisheries in Canadian waters are known to have taken harbor porpoise in the past (Barlow et al. 1994, Stacey et al. 1997), few data are available because the fisheries were not monitored. In 2001, the Department of Fisheries and Oceans, Canada, conducted a federal fisheries observer program and a survey of license holders to estimate the incidental mortality of harbor porpoise in selected salmon fisheries in southern British Columbia (Hall et al. 2002). Based on the observed bycatch of porpoise (2 harbor porpoise deaths) in the 2001 fishing season, the estimated mortality for southern British Columbia in 2001 was 20 porpoise per 810 boat days fished or a total of 80 harbor porpoise. However, it is not known how many harbor porpoise from the Washington Inland Waters stock are currently taken in the waters of southern British Columbia.

**Other Mortality**

A significant increase in harbor porpoise strandings reported throughout Oregon and Washington in 2006 prompted the Working Group on Marine Mammal Unusual Mortality Events to declare an Unusual Mortality Event (UME) on 3 November 2006 (Huggins 2008). A total of 114 harbor porpoise strandings were reported and confirmed along the Oregon and Washington outer coasts and Washington inland waters in 2006 and 2007 (Huggins 2008). A more recent analysis of strandings before and after the suspected UME indicates that no UME occurred (Huggins *et al*. 2015). The perceived increase in mortality was the result of multiple factors: an increase in the population of harbor porpoise, a shift of the population into Washington inland waters, and a well-established stranding network with improved response and reporting (Huggins *et al*. 2015).

**STATUS OF STOCK**

Harbor porpoise are not listed as "depleted" under the MMPA or listed as "threatened" or "endangered" under the Endangered Species Act. Based on currently available data, the minimum annual level of total human-caused mortality and serious injury (7.2) harbor porpoise per year (corrected for undetected strandings) does not exceed the PBR of 66 animals. Therefore, the Washington Inland Waters harbor porpoise stock is not classified as "strategic." The minimum annual fishery mortality and serious injury for this stock (7.2 harbor porpoise per year) exceeds 10% of PBR (6.6) and, therefore, cannot be considered to be insignificant and approaching zero mortality and serious injury rate. The status of this stock relative to its Optimum Sustainable Population (OSP) and population trends is unknown. Although harbor porpoise sightings in southern Puget Sound declined from the 1940s through the 1990s, harbor porpoise sightings have increased seasonally in this area in the last 10 years.

This stock is not recognized as "strategic," however, the current mortality rate is based on stranding data, since the Washington Puget Sound Region salmon set/drift gillnet fishery has not been observed since 1994. Evaluation of the estimated take level is complicated by a lack of knowledge about the extent to which harbor porpoise from U.S. waters frequent the waters of British Columbia and are, therefore, subject to fishery-related mortality. It is appropriate to consider whether the current take level is different from the take level in 1994, when the fishery was last observed. No new information is available about mortality per set, but 1) fishing effort has decreased since 1994. Based on surveys conducted in between 1991/1992 and 2015 (Calambokidis *et al.* 1993, Smultea *et al.* 2015a, 2015b), the population appears to have increased, but a statistical trend analysis has not been performed with existing data. However, an increase in harbor porpoise use of southern Puget Sound in recent years is apparent (Evenson *et al*. 2016).

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# DALL'S PORPOISE (*Phocoenoides dalli dalli*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Dall's porpoises are endemic to temperate waters of the North Pacific. Off the U.S. west coast, they are seen in shelf, slope and offshore waters (Figure 1). Sighting patterns from aerial and shipboard surveys in California, Oregon and Washington waters (Green et al. 1992, 1993; Forney and Barlow 1998; Barlow 2016, Henry *et al.* 2020) suggest that movement between these states occurs as oceanographic conditions change, both on seasonal and inter-annual time scales (Boyd *et al.* 2018, Becker *et al.* 2020). The southern end of this stock's range is not well-documented, but they are seen off Southern California in winter, and during cold-water periods they probably range into Mexican waters off northern Baja California. The stock structure of eastern North Pacific Dall's porpoises is not known, but based on patterns of stock differentiation in the western North Pacific, where they have been more intensively studied, it is expected that separate stocks may exist (Perrin and Brownell 1994). Although Dall's porpoises occur outside U.S. waters, there are no cooperative management agreements with Mexico or Canada for fisheries which may take this species (e.g. gillnet fisheries). For the Marine Mammal Protection Act (MMPA) stock assessment reports, Dall's porpoises within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Alaskan waters.

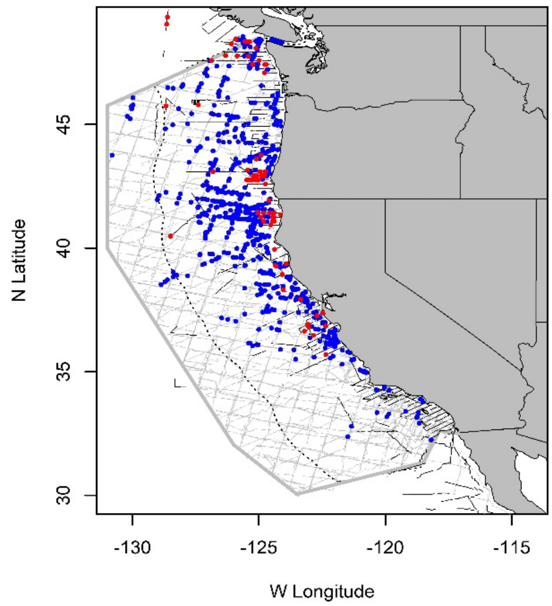


**Figure 1**. Dall's porpoise sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Population size has been estimated from a series of line-transect surveys using multiple-covariate line-transect approaches (Barlow 2016), Bayesian integrated population redistribution models (Boyd *et al.* 2018) and species distribution models (SDMs) (Becker *et al*. 2020) (Figure 2). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2012, 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 16,498 (CV=0.608) animals (Becker *et al.* 2020). Additional numbers of Dall's porpoises occur in the inland waters of Washington state, but the most recent abundance estimate obtained in 1996 (900 animals, CV=0.40) is over 8 years old (Calambokidis *et al*. 1997) and is not included in the overall estimate of abundance for this stock.

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate for the outer coasts of California, Oregon and Washington waters is 10,286 Dall's porpoises (Becker *et al.* 2020).

### Current Population Trend

The distribution and abundance of Dall's porpoise off California, Oregon and Washington varies at both seasonal and interannual time scales (Forney and Barlow 1998, Becker *et al*. 2012, Barlow 2016, Boyd *et al.* 2018), and the entire population does not reside within the California Current, thus, assessment of population trends isn't straightforward. Boyd *et al.* (2018) reported that the population size of Dall's porpoise within the California Current survey area was relatively stable over each summer/fall survey season from 1996 to 2008, and noted that the distribution of animals expanded and contracted with the extent of suitable habitat.

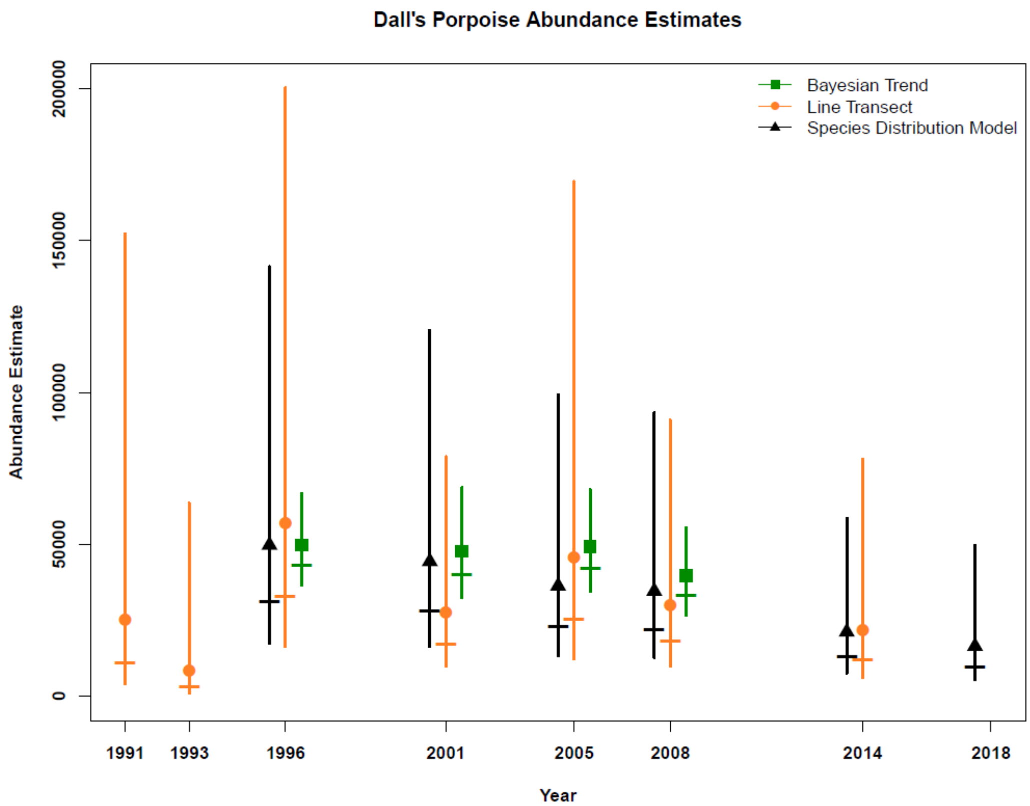


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016), Bayesian trend models (Boyd *et al.* 2018), and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical bars indicate approximate 95% confidence limits for line-transect, Bayesian trend, and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for Dall's porpoise off the U.S. west coast.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (10,286) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status and mortality rate CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 99 Dall's porpoises per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury information for this stock of Dall's porpoises is given in Table 1. More detailed information on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for Dall's porpoise in the California drift gillnet fishery for the five most recent years of monitoring, 2015-2019, averages 0.46 animals per year (Carretta 2021). Although Dall's porpoises have been incidentally killed in West Coast groundfish fisheries in the past, no takes of this species were observed during the five most recent years for which data are available, 2012-2016 (Jannot *et al*. 2018). One animal was killed in an unidentified gillnet fishery in Washington state inland waters in 2016 (Carretta *et al*. 2021).

**Table 1.** Summary of available information on the incidental mortality and serious injury of Dall's porpoises (California/ Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta 2021, Carretta *et al.* 2021; Jannot *et al*. 2018). All observed entanglements of Dall's porpoises resulted in the death of the animal. Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 0 | 2.3 (0.4) | 0.46 (0.4) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)** | observer | 2012-2016 | 98% - 100% | 0 | 0 | 0 |
| Unidentified gillnet fishery | Stranding | 2015-2019 | n/a | 1 | 1 | ≥ 0.2 |
| **Minimum total annual takes** | | | | | | 0.66 (0.4) |

## STATUS OF STOCK

The status of Dall's porpoises in California, Oregon and Washington relative to OSP is not known, and trends in abundance were described as stable by Boyd *et al.* (2018). No habitat issues are known to be of concern for this species. It is not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality of Dall's porpoise (0.66 animals) is estimated to be less than the PBR (99), and they are not classified as a "strategic" stock under the MMPA. The total fishery mortality and serious injury for this stock is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate.

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Revised 3/14/2022

# PACIFIC WHITE-SIDED DOLPHIN (*Lagenorhynchus obliquidens*): California/Oregon/Washington, Northern and Southern Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Pacific white-sided dolphins are endemic to temperate waters of the North Pacific Ocean, and common both on the high seas and along the continental margins (Brownell et al. 1999). Off the U.S. west coast, Pacific white-sided dolphins occur primarily in shelf and slope waters (Figure 1). Sighting patterns from aerial and shipboard surveys conducted in California, Oregon and Washington (Green et al. 1992; 1993; Forney and Barlow 1998; Barlow 2016) suggest seasonal north-south movements, with animals found primarily off California during the colder water months and shifting northward into Oregon and Washington as water temperatures increase in late spring and summer. Stock structure throughout the North Pacific is poorly understood, but based on morphological evidence, two forms are known off the California coast (Walker et al. 1986). Specimens belonging to the northern form were collected from north of about 33°N, (Southern California to Alaska), and southern specimens were obtained from about 36°N southward along the coasts of California and Baja California. Samples of both forms have been collected in the Southern California Bight, but it is unclear whether this indicates sympatry in this region or whether they may occur there at different times (seasonally or interannually). Genetic analyses have confirmed the distinctness of animals found off Baja California from animals occurring in U.S. waters north of Point Conception, California and the high seas of the North Pacific (Lux *et al*. 1997). Based on these genetic data, an area of mixing between the two forms appears to be located off Southern California (Lux *et al*. 1997). Two types of echolocation have been documented for Pacific white-sided dolphins off Southern California and these have been hypothesized to reflect acoustic differences between the two forms (Soldevilla et al. 2008, 2011; Henderson et al. 2011).

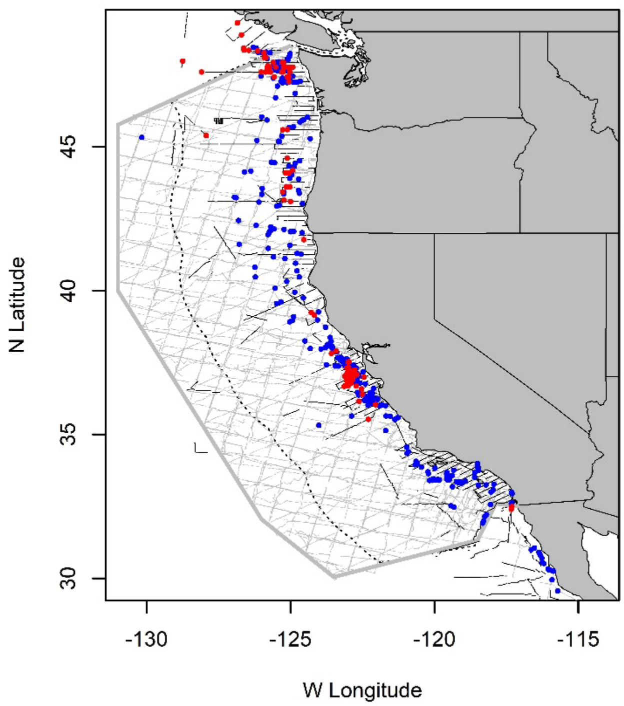


**Figure 1.** Pacific white-sided dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

Although there is clear evidence that two forms of Pacific white-sided dolphins occur along the U.S. west coast, there are no known differences in color pattern, and it is not currently possible to distinguish the two stocks reliably during surveys. Geographic stock boundaries appear dynamic and are poorly understood, and therefore cannot be used to differentiate the two forms. Until means of differentiating the two forms for abundance and mortality estimation are developed, these two stocks are managed as a single unit. Pacific white-sided dolphins are not restricted to U.S. territorial waters, but there are no cooperative management agreements with Mexico or Canada for fisheries which may take this species (e.g. gillnet fisheries). For the Marine Mammal Protection Act (MMPA) stock assessment reports, Pacific white-sided dolphins within the

Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Alaskan waters.

**POPULATION SIZE**

Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables in a generalized additive model framework using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2012, 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 34,999 (CV=0.222) animals (Becker *et al.* 2020).

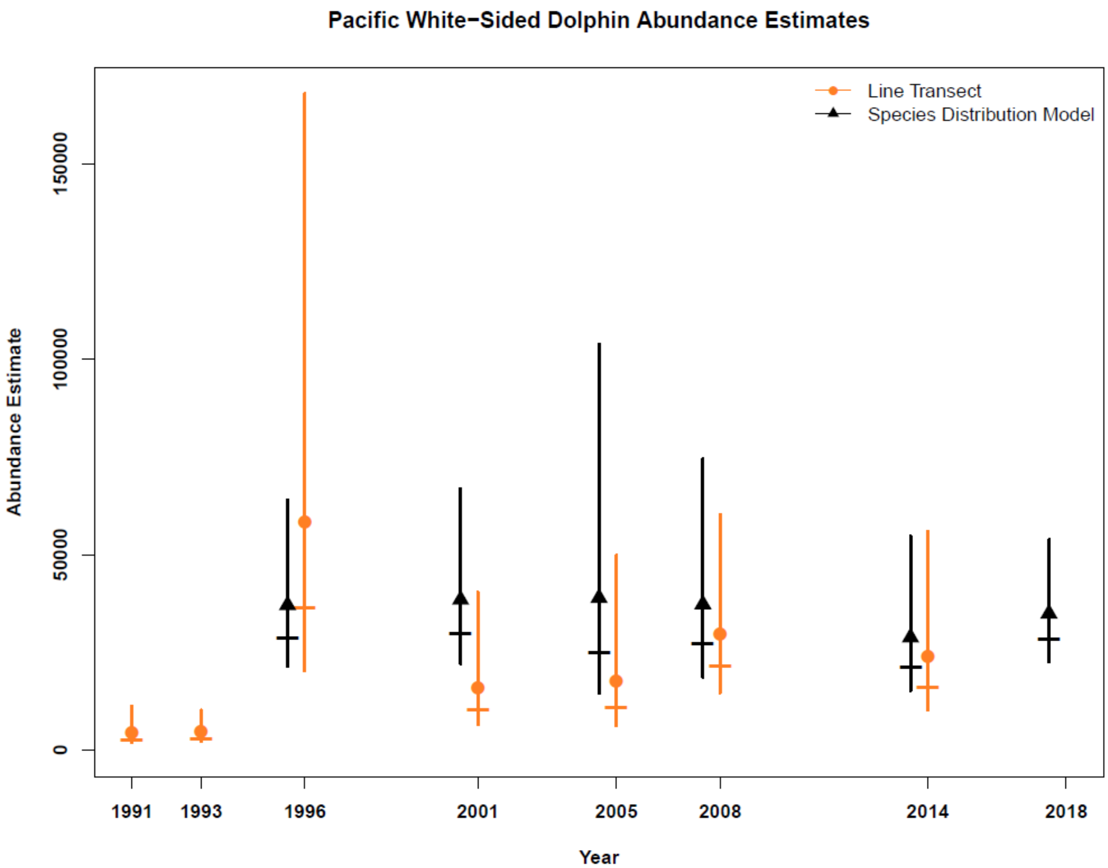


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical bars indicate approximate 95% log-normal confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20$^{th}$ percentiles of mean estimates.

**MINIMUM POPULATION ESTIMATE**

The log-normal 20th percentile of the 2018 abundance estimate is 29,090 Pacific white-sided dolphins.

**CURRENT POPULATION TREND**The distribution and abundance of Pacific white-sided dolphins off California, Oregon and Washington varies considerably at both seasonal and interannual time scales (Forney and Barlow 1998, Becker *et al*. 2012, 2020, Barlow 2016), but no long-term trends have been identified (Figure 2).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for Pacific white-sided dolphins off the U.S. west coast.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (29,090) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with a mortality rate CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 279 Pacific white-sided dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury information for this stock of Pacific white-sided dolphin is shown in Table 1. More detailed information on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for Pacific white-sided dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2015-2019 is 4.0 (CV=0.37) per year. Unidentified fishery deaths from strandings are multiplied by a correction factor of 4.0 to account for incomplete detection of carcasses (Carretta *et al.* 2016) (Table 1).

**Table 1.** Summary of available information on the incidental mortality and injury of Pacific white-sided dolphins (California/ Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta 2020; Jannot *et al.* 2018). All observed entanglements of Pacific white-sided dolphins resulted in the death of the animal. Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| CA/OR thresher shark/swordfish drift gillnet fishery | observer | 2015-2019 | 21% | 0 | 12.1 (0.76) | 2.4 (0.76) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)** | observer | 2012-2016 | 98% - 100% | 4 | 4 (n/a) | 0.8 (n/a) |
| **Unidentified gillnet fishery** | stranding | 2015-2019 | n/a | 1 | ≥ 4 (0.46) | ≥ 0.8 (0.46) |
| **Minimum total annual takes** | | | | | | 4.0 (0.37) |

### Other removals

Pacific white-sided dolphins have been seriously injured and killed in scientific research trawls for sardines and rockfish. From 2015 - 2019, there were 14 deaths and 1 serious injury of Pacific white-sided dolphins in scientific research trawls, or an average of 3.0 annually (Carretta *et al*. 2021).

## STATUS OF STOCK

The status of Pacific white-sided dolphins in California, Oregon and Washington relative to OSP is not known, and there is no indication of a trend in abundance for this stock. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality and serious injury from fisheries (4.0 animals), plus other anthropogenic sources (3.0) during 2015-2019 (7.0 annually) is estimated to be less than the PBR (279), and therefore this stock of Pacific white-sided dolphins is not classified as a "strategic" stock under the MMPA. The total commercial fishery mortality and serious injury for this stock (4.0/yr) is less than 10% of the calculated PBR and, therefore, is considered to be insignificant and approaching zero.

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Revised 02/9/2017

# RISSO'S DOLPHIN (*Grampus griseus*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso's dolphins are distributed world-wide in tropical and warm-temperate waters. Off the U.S. West coast, Risso's dolphins are commonly seen on the shelf in the Southern California Bight and in slope and offshore waters of California, Oregon and Washington. Based on sighting patterns from recent aerial and shipboard surveys conducted in these three states during different seasons (Figure 1), animals found off California during the colder water months are thought to shift northward into Oregon and Washington as water temperatures increase in late spring and summer (Green et al. 1992, 1993). The southern end of this population's range is not well-documented, but previous surveys have shown a conspicuous 500 nmi distributional gap between these animals and Risso's dolphins sighted south of Baja California and in the Gulf of California (Mangels and Gerrodette 1994). Thus this population appears distinct from animals found in the eastern tropical Pacific and the Gulf of California. Although Risso's dolphins are not restricted to U.S. waters, cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species (e.g. gillnet fisheries). For the Marine Mammal Protection Act (MMPA) stock assessment reports, Risso's dolphins within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Hawaiian waters.

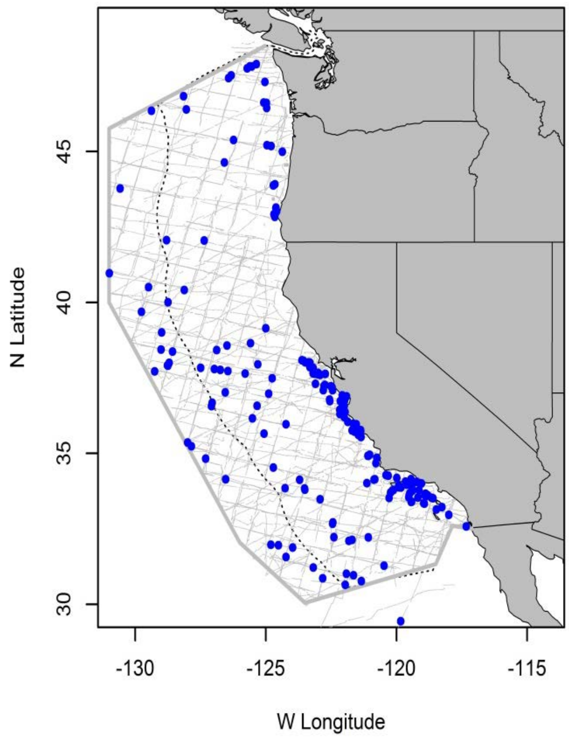


**Figure 1.** Risso’s dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2014 (Barlow 2016). Dashed line represents the U.S. EEZ, thin gray lines indicate completed transect effort of all surveys combined.

## POPULATION SIZE

The distribution of Risso’s dolphins throughout this region is highly variable, apparently in response to oceanographic changes on both seasonal and interannual time scales (Forney and Barlow 1998). As oceanographic conditions vary, Risso’s dolphins may spend time outside the U.S. Exclusive Economic Zone, and therefore a multi-year average abundance estimate is the most appropriate for management within U.S. waters. The most recent estimate of Risso’s dolphin abundance is the geometric mean of estimates from 2008 and 2014 summer/autumn vessel-based line-transect surveys of California, Oregon, and Washington waters, 6,336 (CV=0.32) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.

### Minimum Population Estimate

The log-normal 20th percentile of the 2008-2014 geometric mean abundance estimate is 4,817 Risso's dolphins.

### Current Population Trend

The distribution and abundance of Risso's dolphins off California, Oregon and Washington varies considerably at both seasonal and interannual time scales (Forney and Barlow 1998, Becker *et al*. 2012, Barlow 2016), but no long-term trends have been identified.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this stock.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (4,817) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with a mortality rate CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 46 Risso's dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury information for this stock of Risso's dolphin is shown in Table 1. More detailed information on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for Risso's dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2010-2014, is an average of 1.3 per year (Carretta *et al*. 2017, Table 1). Although some Risso's dolphins have been incidentally killed in West Coast groundfish fisheries in the past, no takes of this species were observed during 2009-2013 (Jannot *et al*. 2011, NWFSC unpublished data). 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.

Historically, Risso's dolphin mortality has been documented in the squid purse seine fishery off Southern California (Heyning et al. 1994). This mortality probably represented animals killed intentionally to protect catch or gear, rather than incidental mortality, and such intentional takes are now illegal under the 1994 Amendment to the MMPA. This fishery has expanded markedly since 1992 (California Department of Fish and Game, unpubl. data). An observer program in the squid purse seine fishery from 2004-2008 observed 377 sets (<10%) without an observed Risso's dolphin interaction.

Human-caused mortality and injury documentation is often based on stranding data, where raw counts are negatively-biased because only a fraction of carcasses are detected (Carretta *et al.* 2016a). Carretta *et al.* (2016b) estimated the mean recovery rate of California coastal bottlenose dolphin carcasses to be 25% (95% CI 20% - 33%) and stated that given the extremely coastal habits of coastal bottlenose dolphins, carcass recovery rates for this stock represented a maximum, compared with more pelagic dolphin species in the region. Therefore, in this stock assessment report and others involving dolphins along the U.S. West Coast, human-related deaths and injuries counted from beach strandings along the outer U.S. West Coast are multiplied by a factor of 4 to account for the non-detection of most carcasses (Carretta *et al*. 2016b). Three Risso's dolphins stranded during 2010-2014 with evidence of fishery interaction (Carretta *et al*. 2016a), yielding a minimum estimate of 12 fishery-related dolphin deaths.

**Table 1.** Summary of available information on the incidental mortality and serious injury of Risso's dolphin (California/ Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta *et al*. 2016b, 2017; Jannot *et al*. 2011; NWFSC, unpublished data). All observed entanglements of Risso's dolphins resulted in the death of the animal. Human-caused mortality values based on strandings recovered along the outer U.S. West Coast are multiplied by a correction factor of 4 to account for undetected mortality (Carretta *et al*. 2016a). Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated AnnualMortality (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| CA/OR thresher shark/swordfish drift gillnet fishery | observer | 2010<br>2011<br>2012<br>2013<br>2014 | 12%<br>20%<br>19%<br>37%<br>24% | 0<br>1<br>0<br>0<br>0 | 1.5 (2.5)<br>2.8 (1.3)<br>0.8 (2.8)<br>0.9 (1.9)<br>0.7 (2.8) | 1.3 (0.93) |
| CA deep set longline fishery | observer | 2005-2008 | 100% | 0 | 0 | 0 |
| Market squid purse seine | observer | 2004-2008 | <10% | 0 | 0 | 0 |
| Unknown fishery | Stranding | 2007-2013 | n/a | 3 | ≥ 12 | ≥2.4 (0.46) |
| WA/OR/CA groundfish (bottom trawl)[a] | observer | 2009-2013 | 23% (2009)<br>18% (2010)<br>100% (2011-2013) | 0 | 0 | 0 |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ 3.7 (0.44) |

## STATUS OF STOCK

The status of Risso's dolphins off California, Oregon and Washington relative to OSP is not known, and there are insufficient data to evaluate potential trends in abundance. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Over the last 5-year period (2010-2014), the average annual human-caused mortality (3.7 animals) is estimated to be less than the PBR (46), and therefore they are not classified as a "strategic" stock under the MMPA. The total fishery mortality and serious injury for this stock (3.7) is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate.

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# COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus*): California Coastal Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Bottlenose dolphins are distributed world-wide in tropical and warm-temperate waters. In many regions, including California, separate coastal and offshore populations are known (Walker 1981; Ross and Cockcroft 1990; Van Waerebeek *et al*. 1990). The California coastal stock of bottlenose dolphins is distinct from the offshore stock, based on significant differences in genetics and cranial morphology (Perrin *et al*. 2011, Lowther-Thielking *et al*. 2015). Of 56 haplotypes found among coastal and offshore bottlenose dolphins in the region, only one is shared by both populations (Perrin *et al*. 2011). California coastal bottlenose dolphins are found within about one kilometer of shore (Hansen, 1990; Carretta *et al*. 1998; Defran and Weller 1999) from central California south into Mexican waters, at least as far south as San Quintin, Mexico (Figure 1). In southern California, animals are found within 500 m of the shoreline 99% of the time and within 250 m 90% of the time (Hanson and Defran 1993). Oceanographic events appear to influence the distribution of animals along the coasts of California and Baja California, Mexico, as indicated by a change in residency patterns along Southern California and a northward range extension into central California after the 1982-83 El Niño (Hansen and Defran 1990; Wells *et al*. 1990). Since the 1982-83 El Niño, which increased water temperatures off California, they have been consistently sighted in central California as far north as San Francisco. Photo-identification studies have documented north-south movements of coastal bottlenose dolphins (Hansen 1990; Defran *et al*. 1999), and monthly counts based on surveys between the U.S./Mexican border and Point Conception are variable (Carretta *et al*. 1998), indicating that animals are moving into and out of this area. There is little site fidelity of coastal bottlenose dolphins along the California coast; over 80% of the dolphins identified in Santa Barbara, Monterey, and Ensenada have also been identified off San Diego (Defran *et al*. 1999, Feinholz 1996, Defran *et al.* 2015). The area between Ensenada and San Quintin, Mexico may represent a southern boundary for the California coastal population, as very low rates of photo-ID overlap of individuals (3%) have been found between the two areas, compared to higher overlap rates to the north (Defran *et al.* 2015, Figure 1). Although coastal bottlenose dolphins are not restricted to U.S. waters, cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species. Therefore, the management stock includes only animals found within U.S. waters. For the Marine Mammal Protection Act (MMPA) stock assessment reports, bottlenose dolphins within the Pacific U.S. Exclusive Economic Zone are divided into seven stocks: 1) California coastal stock (this report), 2) California, Oregon

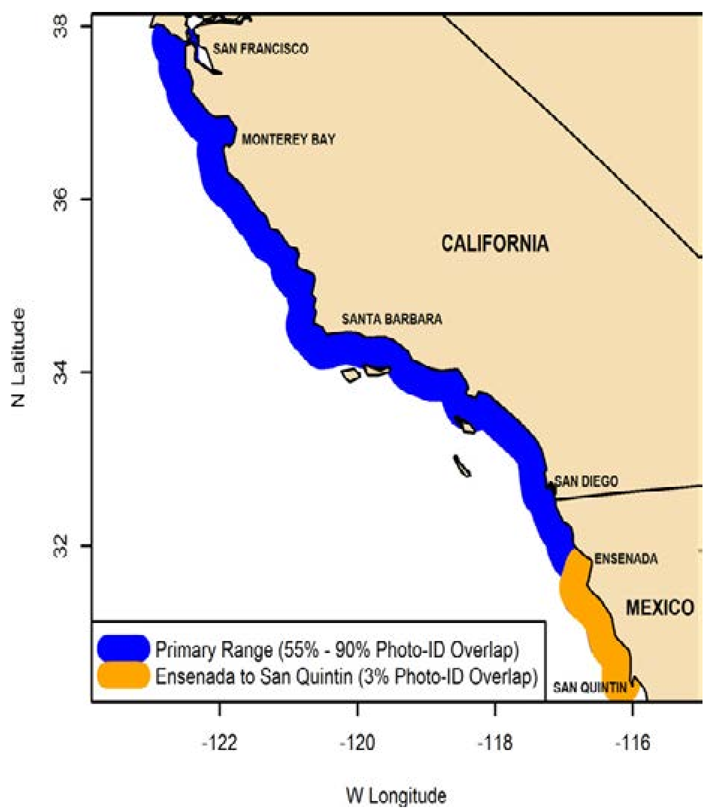


**Figure 1**. Approximate range of California coastal bottlenose dolphins, based on aerial and boat-based sighting surveys. This population of bottlenose dolphins is found within about 1 km of shore.

and Washington offshore stock, and five stocks in Hawaiian waters: 3) Kauai/Niihau, 4) Oahu, 5) 4-Islands (Molokai, Lanai, Maui, Kahoolawe), 6) Hawaii Island and 7) the Hawaiian Pelagic Stock.

## POPULATION SIZE

Based on photographic mark-recapture surveys conducted along the San Diego coast from 2009 to 2011 (Weller *et al*. 2016), two separate population size estimates were generated from open and closed mark-recapture models. The best open model generated an estimate of 515 (95% CI = 470–564, CV= 0.05) animals, while the best closed model produced an estimate of 453 (95% CI = 411–524, CV=0.06) animals. These estimates are for *marked animals only* and do not include an estimated ~ 40% of animals that are not individually recognizable (Weller *et al.* 2016). The estimated fraction of unmarked animals is highly uncertain because it is unknown how often unmarked animals are resighted. The new estimates are the largest obtained for this stock, dating back to the 1980s (Defran and Weller 1999, Dudzik 1999, Dudzik *et al.* 2006). For comparison with previous estimates of this stock, the closed population estimate of 453 (CV=0.06) animals is used as the best estimate of abundance.

### Minimum Population Estimate

The minimum population size is based on the minimum number of individually identifiable animals documented during surveys in 2009-2011, or 346 animals (Weller *et al*. 2016). This number of individually recognizable dolphins exceeds the number recorded in previous survey periods: 1984-1986 (160 dolphins); 1987-1989 (284); 1996-1998 (260); and 2004-2005 (164) (Weller *et al.* 2016).

### Current Population Trend

Based on a comparison of mark-recapture abundance estimates for the periods 1987-89 ($\hat{N}$= 354), 1996-98 ($\hat{N}$= 356), and 2004-05 ($\hat{N}$= 323), Dudzik *et al*. (2006) stated that the population size had remained stable over this period. New estimates of 450 – 515 animals based on 2009-2011 surveys are the highest to date and include a high proportion (~75%) of previously uncatalogued dolphins (Weller *et al.* 2016). The number of individually-identifiable animals from 2009-2011 surveys (346) is equal to or exceeds previous mark-recapture *abundance estimates* for this stock. This suggests that the population may be growing, although the movement of dolphins north from Mexican waters may also contribute to the observed increase in unique individuals.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for California coastal bottlenose dolphins.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (346) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with mortality rate CV ≥ 0.3 and ≤ 0.6; Wade and Angliss 1997), resulting in a PBR of 3.3 coastal bottlenose dolphins per year. Not all California coastal bottlenose dolphins are present in U.S. waters at any given moment and approximately 18% of the stock’s range occurs in Mexican waters. Thus, the PBR is prorated by a minimum factor of 0.82 to account for time that animals spend outside of U.S. waters. Without additional data on the residence times of dolphins in Mexican waters, this factor cannot be improved upon. Because this stock spends some of its time outside the U.S. EEZ, the PBR allocation for U.S. waters is 3.3 x 0.82 = 2.7 dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Due to its exclusive use of coastal habitats, this bottlenose dolphin population is susceptible to fishery-related mortality in coastal gillnet fisheries, such as the halibut and yellowtail set gillnet fishery, which was responsible for one documented coastal bottlenose dolphin death in 2003. Observer coverage in this fishery from 2010-2014 has been 9% (806 observed sets from an estimated 8,654 sets fished), with no observations of coastal bottlenose dolphin entanglements. Between 2010 and 2014, there were two fishery-related deaths of coastal bottlenose dolphins (stock ID confirmed via genetics, Lowther-Thielking *et al*. 2015). Both animals had evidence of entanglement with rope of unknown origin. A summary of information on fishery mortality and injury for this stock of bottlenose dolphin is shown in Table 1. Coastal gillnet fisheries exist in Mexico and may take animals from this population, but no details are available. Human-caused mortality and injury documentation is often based on stranding data, where raw counts are negatively-biased because only a fraction of carcasses are detected (Williams *et al.* 2011), even for extremely coastal species (Wells *et al.* 2015). Carretta *et al*. (2016b) estimated the mean recovery rate of carcasses of California coastal bottlenose dolphins to be 25% (95% CI 20% - 33%). Given the extremely coastal habits of California coastal bottlenose dolphins, Carretta *et al*. (2016b) argue that carcass recovery rates for this population represent a maximum rate, compared to more pelagic dolphin species in the region. Therefore, in this stock assessment report and others involving dolphins along the U.S. west coast, human-related deaths and injuries counted from beach strandings are multiplied by a factor of 4 to account for the non-detection of most carcasses (Carretta *et al*. 2016b).

**Table 1.** Summary of available information on the incidental mortality and serious injury of bottlenose dolphins (California Coastal Stock) in commercial fisheries that might take this species. Human-caused mortality values based on strandings recovered on the outer U.S. West Coast are multiplied by a correction factor of 4 to account for undetected mortality (Carretta *et al.* 2016b). The coefficient of variation (CV) for corrected carcass counts was derived from the results of Carretta *et al*. (2016b), who estimated that 25% (95% CI = 20% - 33%) of all available carcasses were recovered / documented.

| **Fishery Name** | **Data Type** | **Years** | **Percent Observer Coverage** | **Observed Mortality** | **Estimated Annual Mortality** | **Mean Annual Takes (CV)** |
|---|---|---|---|---|---|---|
| CA angel shark/ halibut and other species large mesh (>3.5 in) set gillnet fishery | Observer | 2010-2014 | 9% | 0 | 0 | 0 |
| Unknown fishery | Stranding | 2010-2014 | Two strandings with evidence of entanglement in rope or braided material | | | ≥0.4 x 4 (correction factor) = 1.6 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ 1.6 (0.46) |

**Other removals**

Seven coastal bottlenose dolphins were collected during the late 1950s in the vicinity of San Diego (Norris and Prescott 1961). Twenty-seven additional bottlenose dolphins were captured off California between 1966 and 1982 (Walker 1975; Reeves and Leatherwood 1984), but based on the locations of capture activities, these animals probably were offshore bottlenose dolphins (Walker 1975). No additional captures of coastal bottlenose dolphins have been documented since 1982, and no live-capture permits are currently active for this species.

In 2012, a coastal bottlenose dolphin (stock ID confirmed via genetics) was found floating under a U.S. Navy marine mammal program dolphin pen enclosure dock and was assumed to have become entangled in the net curtain (Carretta *et al.* 2016a). Another, presumed coastal bottlenose dolphin (based on proximity to shore) became entrapped and drowned in a sea otter research net in 2012. The average annual non-fishery related mortality and serious injury of coastal bottlenose dolphins from 2010-2014 is 0.4 animals (2 animals / 5 years).

**Habitat Issues**

Pollutant levels, especially DDT residues, found in Southern California coastal bottlenose dolphins have been found to be among the highest of any cetacean examined (O'Shea et al. 1980; Schafer et al. 1984). Although the effects of pollutants on cetaceans are not well understood, they may affect reproduction or make the animals more prone to other mortality factors (Britt and Howard 1983; O'Shea et al. 1999). This population of bottlenose dolphins may also be vulnerable to the effects of morbillivirus outbreaks, which were implicated in the 1987-88 mass mortality of bottlenose dolphins on the U.S. Atlantic coast (Lipscomb et al. 1994).

## STATUS OF STOCK

The status of coastal bottlenose dolphins in California relative to OSP is not known, and there is no evidence of a trend in abundance. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Coastal bottlenose dolphins are not classified as a "strategic" stock under the MMPA because total annual fishery (1.6) and other anthropogenic mortality (0.4) and serious injury for this stock ($\geq$ 2.0 per year) is less than the PBR (2.7). The total human-caused mortality and serious injury for this stock is not less than 10% of the calculated PBR and, therefore, cannot be considered to be insignificant and approaching zero. Recent population size estimates of 450 to 515 marked individuals are the highest recorded to date (Weller et al. 2016), but it is unknown how much of this increase is due to population growth versus immigration.

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Revised 3/15/2022

# COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): California/Oregon/Washington Offshore Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Bottlenose dolphins are distributed world-wide in tropical and warm-temperate waters. In many regions, including California, separate coastal and offshore populations are known (Walker 1981; Ross and Cockcroft 1990; Van Waerebeek et al. 1990; Lowther 2006; Perrin *et al.* 2011). On surveys conducted off California, offshore bottlenose dolphins have been found at distances greater than a few kilometers from the mainland and throughout the Southern California Bight. They have also been documented in offshore waters as far north as about 41ºN (Figure 1), and they may range into Oregon and Washington waters during warm-water periods. Sighting records off California and Baja California (Lee 1993; Mangels and Gerrodette 1994) suggest that offshore bottlenose dolphins have a continuous distribution in these two regions. There is no apparent seasonality in distribution (Forney and Barlow 1998). Offshore bottlenose dolphins are not restricted to U.S. waters, but cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species (e.g. gillnet fisheries). Therefore, the management stock includes only animals found within U.S. waters. For the Marine Mammal Protection Act (MMPA) stock assessment reports, bottlenose dolphins within the Pacific U.S. Exclusive Economic Zone are divided into seven stocks: 1) California coastal stock, 2) California, Oregon and Washington offshore stock (this report), and five stocks in Hawaiian waters: 3) Kauai/Niihau, 4) Oahu, 5) 4-Islands (Molokai, Lanai, Maui, Kahoolawe), 6) Hawaii Island and 7) the Hawaiian Pelagic Stock.

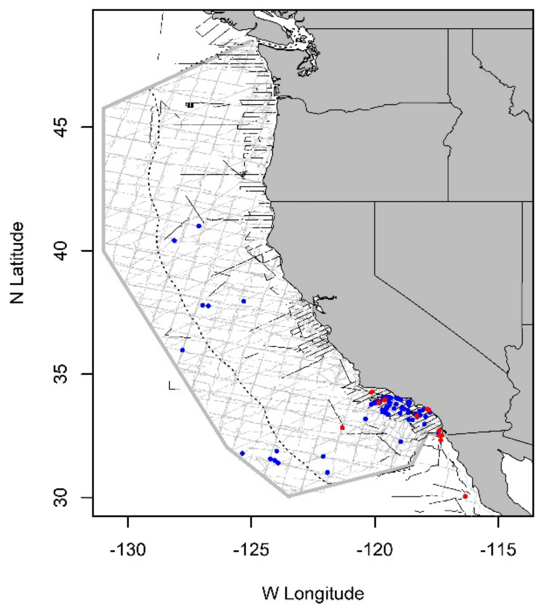


**Figure 1.** Bottlenose dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables in a generalized additive model framework using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2012, 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 3,477 (CV=0.696) animals (Becker *et al.* 2020).

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate is 2,048 offshore bottlenose dolphins (Becker *et al* 2020).

### Current Population Trend

Trend analyses for this stock have not been performed to date, while other stocks with more urgent conservation concerns are analyzed (e.g., Moore and Barlow 2011, 2013). Abundance estimates based on 1991-2014 line-transect surveys (Barlow 2016) and habitat model-based estimates from those same surveys (Becker *et al.* 2020) do not show an apparent trend (Figure 2).

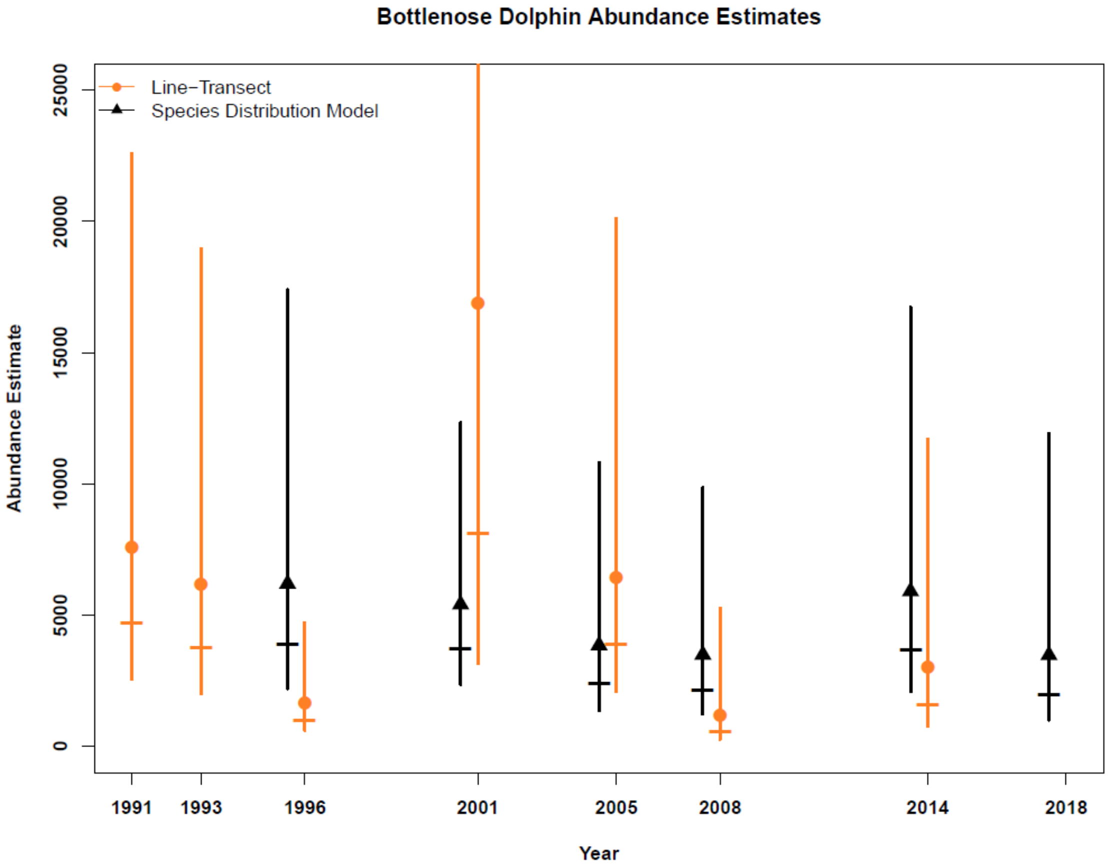


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical bars indicate approximate 95% confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates. The y-axis has been truncated to provide the best display in variability in mean estimates between line-transect and species distribution models. The upper 95% confidence limit for the 2001 line-transect survey is approximately 90,000 animals.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this population of offshore bottlenose dolphins.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (2,048) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with fishery mortality CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 19.7 offshore bottlenose dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

A summary of known fishery mortality and serious injury for this stock of bottlenose dolphin is shown in Table 1. The estimate of mortality and serious injury for bottlenose dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2015-2019, is 0.8 (CV=0.52) individuals, or an average of 0.16 annually (Carretta 2021). Estimated bycatch in fixed-gear groundfish fisheries averages 0.65 annually for the period 2012-2016 (Jannot *et al*. 2018). Estimated annual bycatch across commercial fisheries is 0.82 (CV=0.52) animals (Table 1).

**Table 1.** Summary of available information on the incidental mortality and serious injury of bottlenose dolphins (California/ Oregon/Washington Offshore Stock) in commercial fisheries that might take this species (Carretta *et al*. 2021, Carretta 2021; Jannot *et al*. 2018). Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality (and Serious Injury) | Estimated Mortality and Serious Injury (CV) | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 0 | 0.8 (0.52) | 0.16 (0.52) |
| **CA halibut / white seabass and other species set gillnet fishery** | observer | 2017 | ~ 10% | 0 | 0 | 0 |
| **Limited entry fixed gear** | observer | 2012-2016 | ~ 4% | 0 | 3.28 (n/a)* | 0.656 (n/a) |
| **Minimum total annual takes** | | | | | | ≥ 0.82 (0.52) |

*No coefficient of variation is given for the 5-year Bayesian mean bycatch estimate, but a 95% CI of zero to six animals is reported (Jannot *et al*. 2018).

## STATUS OF STOCK

The status of offshore bottlenose dolphins in California relative to OSP is not known, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Because average annual fishery takes (0.82/yr) are less than the calculated PBR (19.7), offshore bottlenose dolphins are not classified as a "strategic" stock under the MMPA. The total fishery mortality and serious injury for this stock is less than 10% of the PBR and, therefore, can be considered to be insignificant and approaching zero.

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Revised 3/14/2022

# STRIPED DOLPHIN (*Stenella coeruleoalba*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Striped dolphins are distributed world-wide in tropical and warm-temperate pelagic waters. Striped dolphins are commonly encountered in warm offshore waters of California, and a few sightings have been made off Oregon (Figure 1, Barlow 2016, Henry *et al.* 2020). Striped dolphins are also commonly found in the central North Pacific, but sampling between this region and California has been insufficient to determine whether the distribution is continuous. Based on sighting records off California and Mexico, striped dolphins appear to have a continuous distribution in offshore waters of these two regions (Perrin *et al*. 1985; Mangels and Gerrodette 1994). No information on possible seasonality in distribution is available, because the California surveys which extended 300 nmi offshore were conducted only during the summer/fall period. Although striped dolphins are not restricted to U.S. waters, cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species (e.g. gillnet fisheries). Therefore, the management stock includes only animals found within U.S. waters. For the Marine Mammal Protection Act (MMPA) stock assessment reports, striped dolphins within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) waters around Hawaii.

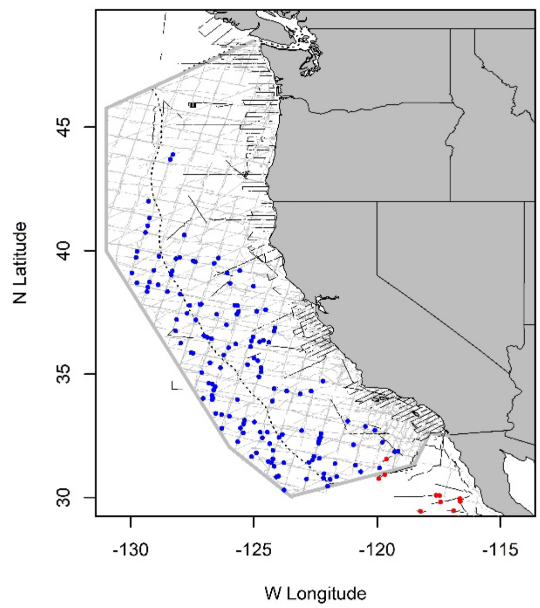


**Figure 1.** Striped dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

The abundance of striped dolphins in this region appears to be variable between years and may be affected by oceanographic conditions, as with other odontocete species (Forney 1997, Becker et al. 2012, Barlow 2016). Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2012, 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 29,988 (CV=0.299) animals (Becker *et al.* 2020).

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate is 23,448 striped dolphins.

### Current Population Trend

The distribution and abundance of striped dolphins off California, Oregon and Washington varies interannually (Barlow 2016, Becker *et al*. 2020), but no long-term trends have been identified (Figure 2). The highest estimates of abundance were obtained in 2014, an anomalously-warm year in the California Current (Bond *et al.* 2015).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for striped dolphins off California.

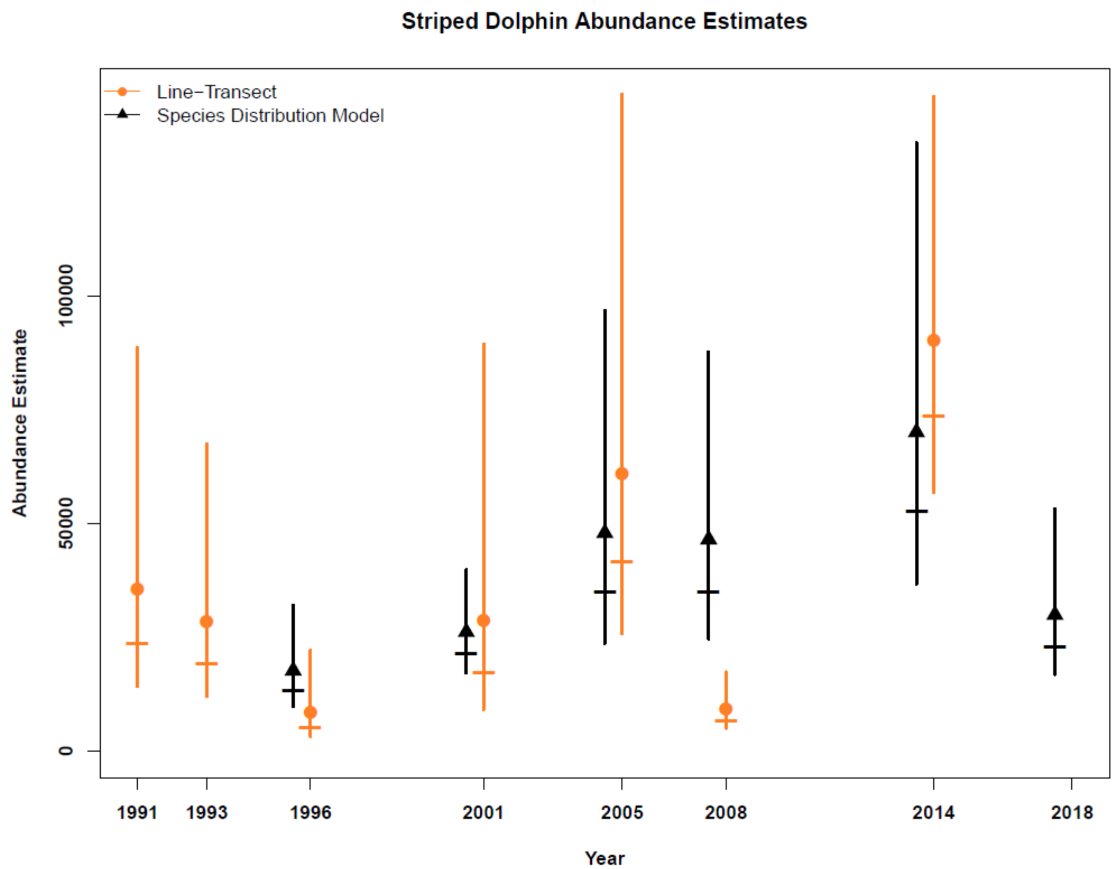


**Figure 2**. Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical bars indicate approximate 95% confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (23,448) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with fishery mortality CV > 0.3 and < 0.6; Wade and Angliss 1997), resulting in a PBR of 225 striped dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury for this stock of striped dolphin is shown in Table 1. More detailed information on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for striped dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2015-2019, is zero animals per year (Carretta 2020). Human-caused mortality and injury documentation is often based on stranding data, where raw counts are negatively-biased because only a fraction of carcasses are detected. In this stock assessment report and others involving dolphins along the U.S. West Coast, human-related deaths and injuries counted from beach strandings along the outer U.S. West Coast are multiplied by a factor of 4 (including a coefficient of variation = 0.46 derived from the results of Carretta *et al*. 2016) to account for the non-detection of most carcasses (Carretta *et al*. 2016). Five striped dolphin stranded during 2015-2019 with evidence of fishery interactions (Carretta *et al*. 2021), yielding a minimum estimate of 20 fishery-related dolphin deaths.

**Table 1.** Summary of available information on the incidental mortality and serious injury of striped dolphins (California/ Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta *et al*. 2021, Carretta 2021). Human-caused mortality values based on strandings recovered along the outer U.S. West Coast are multiplied by a correction factor of 4 to account for undetected mortality (Carretta *et al*. 2016a).

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality + Serious Injury | Estimated Mortality + Serious Injury | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 0 | 0 (n/a) | 0 (n/a) |
| **Unidentified fishery (includes unidentified gillnet)** | Stranding | 2015-2019 | n/a | 5 | ≥ 20 | ≥ 4.0 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ 4.0 (0.46) |

## STATUS OF STOCK

The status of striped dolphins in California relative to OSP is not known, and there are insufficient data to evaluate potential trends in abundance. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Because recent fishery and human-caused mortality (≥ 4.0) is less than 10% of the PBR (225), striped dolphins are not classified as a "strategic" stock under the MMPA, and the total fishery mortality and serious injury for this stock can be considered to be insignificant and approaching zero.

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Revised 3/15/2022

# SHORT-BEAKED COMMON DOLPHIN (*Delphinus delphis delphis*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Short-beaked common dolphins are the most abundant cetacean off California, and are widely distributed from the coast to at least 300 nmi distance from shore (Figure 1). The abundance of this species off California changes on both seasonal and inter-annual time scales (Dohl *et al.* 1986; Forney and Barlow 1998; Barlow 2016). Significant seasonal shifts in the abundance and distribution of common dolphins were identified based on winter/spring 1991-92 and summer/fall 1991 surveys (Forney and Barlow 1998). The distribution of short-beaked common dolphins is continuous southward into Mexican waters to about 13ºN (Perrin *et al.* 1985; Wade and Gerrodette 1993; Mangels and Gerrodette 1994), and short-beaked common dolphins off California may be an extension of the "northern common dolphin" stock defined for management of eastern tropical Pacific tuna fisheries (Perrin *et al.* 1985). Variation in dorsal fin color patterns by latitude suggest there may be multiple stocks in this region, including at least two possible stocks in California (Farley 1995). Although short-beaked common dolphins are not restricted to U.S. waters, cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species. Under the Marine Mammal Protection Act (MMPA), short-beaked common dolphins involved in tuna purse seine fisheries in international waters of the eastern tropical Pacific are managed separately, and they are not included in the assessment reports. For the MMPA stock assessment reports, there is a single Pacific management stock, including only animals found within the U.S. Exclusive Economic Zone of California, Oregon and Washington.

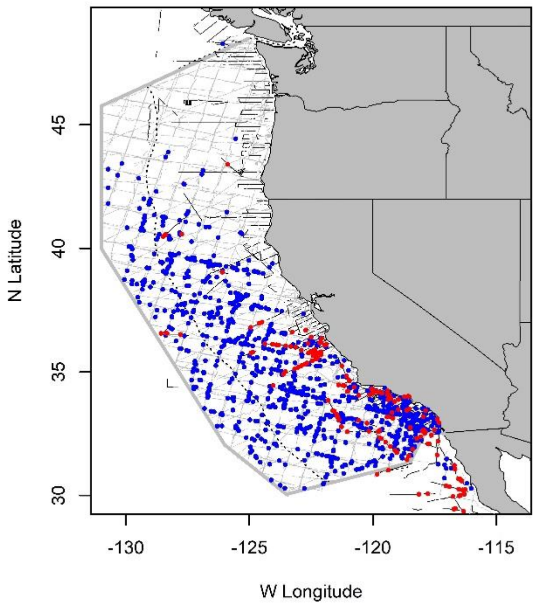


**Figure 1**. Short-beaked common dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry et al. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

The distribution of short-beaked common dolphins in this region is highly variable, in response to oceanographic changes on both seasonal and inter-annual time scales (Heyning and Perrin 1994; Forney 1997; Forney and Barlow 1998). Becker *et al*. (2020a) generated species distribution models (SDMs) from fixed and dynamic ocean variables using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2016, 2020b, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 1,056,308 (CV=0.207) animals (Becker *et al.* 2020a).

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate is 888,971 short-beaked common dolphins.

**Current Population Trend**

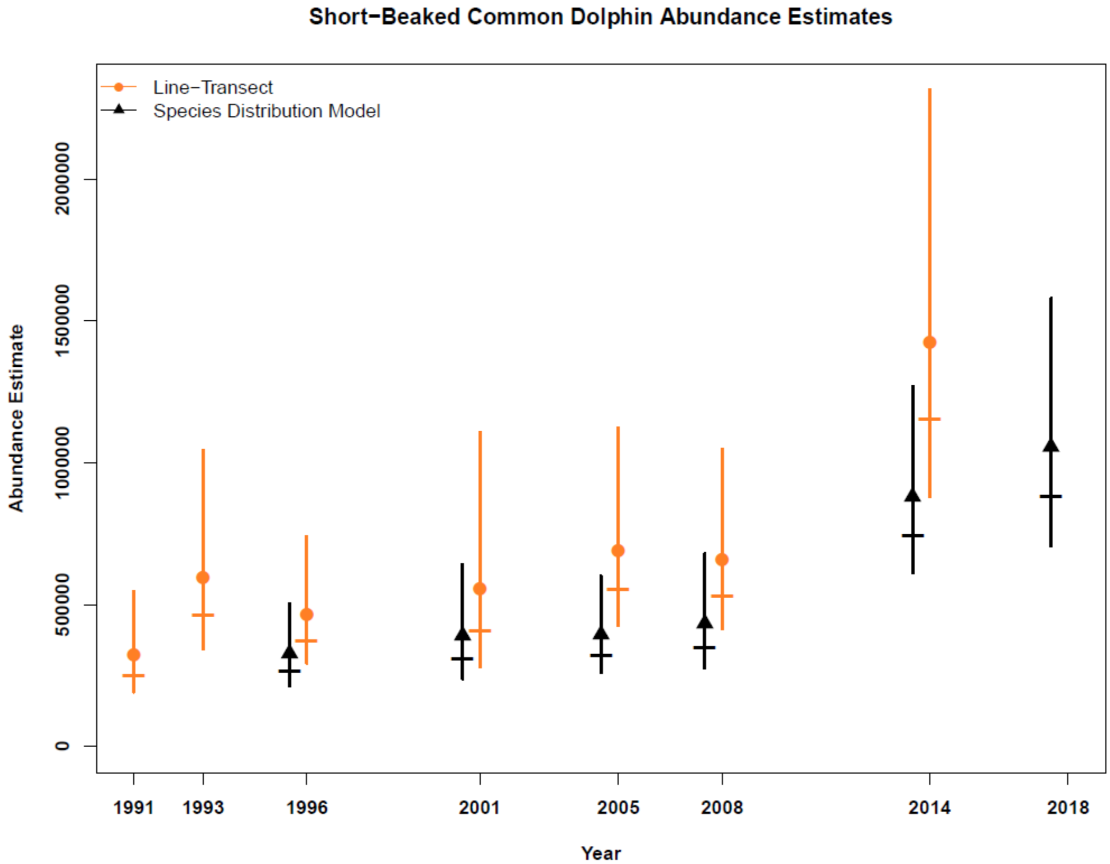


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al.* 2020a) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Vertical bars indicate approximate 95% confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20$^{th}$ percentiles of mean estimates.

Short-beaked common dolphin abundance off the U.S. West Coast increases during warm-water periods (Dohl *et al.* 1986, Forney and Barlow 1998, Barlow 2016). Estimated abundance increased significantly beginning in 2014 survey during extremely warm ocean conditions (Bond *et al*. 2015) and the 2018 estimate is also elevated compared with earlier surveys in the 1991-2018 time series. The increase in short-beaked common dolphin abundance is likely a result of northward movement of this transboundary stock from waters off Mexico (Barlow 2016).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of current or maximum net productivity rates for short-beaked common dolphins.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (888,971) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with a mortality rate CV< 0.30; Wade and Angliss 1997), resulting in a PBR of 8,889 short-beaked common dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury for short-beaked common dolphins is shown in Table 1. The estimate of mortality for short-beaked common dolphin in the California drift gillnet fishery for most recent 5 year period of 2015-2019 averages 26.7 (CV=0.25) per year (Carretta *et al*. 2021, Carretta 2021) (Table 1).

Short-beaked common dolphin have also been killed in the California halibut and white seabass set gillnet fishery, but the fishery has not been observed recently. There were 3 strandings attributed to this fishery during the most-recent 5-year period of 2015-2019. These 3 strandings are corrected for incomplete detection of stranded carcasses, following the methods of Carretta *et al*. 2016, by multiplying observed carcasses by 4 (Table 1). One stranding involved an unidentified gillnet fishery and this one carcass is also multiplied by 4 to account for undetected carcasses. The mean annual bycatch from strandings in the set gillnet fishery and unidentified gillnet fisheries is 16 animals during 2015-2019, or 3.8 animals per year (Table 1). The coefficient of variation for corrected strandings is derived from the results of Carretta et al. (2016). Most common dolphin strandings in the region where gillnet entanglement is identified as a cause of death involve small-mesh typically used in the set gillnet fishery and not large-mesh from the swordfish drift gillnet fishery that operates farther from shore.

**Table 1.** Summary of available information on the incidental mortality and injury of short-beaked common dolphins (California/Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta *et al*. 2021, Carretta 2021). All entanglements resulted in the death of the animal. Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available. Human-caused mortality values based on strandings recovered along the outer U.S. West Coast are multiplied by a correction factor of 4 to account for undetected mortality (Carretta *et al.* 2016a).

| Fishery Name | Data Type | Year | Percent Observer Coverage | Observed Mortality (and Serious Injury) | Estimated Mortality and Serious Injury (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 31 | 133.6 (0.25) | 26.7 (0.25) |
| **CA halibut / white seabass and other species set gillnet fishery** | strandings | 2015-2019 | n/a | 3 | ≥ 12 (0.46) | ≥ 3 ( 0.46) |
| **Hawaii Shallow Set Longline fishery** | observer | 2015-2019 | 100% | 0 | 0 | 0 |
| **Unidentified gillnet fishery** | Stranding | 2015-2019 | n/a | 1 | ≥ 4 (0.46) | ≥ 0.8 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ 30.5 (0.22) |

The California squid purse seine fishery has not been observed since 2008, but there have been past interactions with this fishery, including one mortality (Carretta and Enriquez 2006). No current estimates of bycatch exist for this fishery. There have also been short-beaked common dolphin interactions with the Hawaii shallow set longline fishery (one each in 2011 and 2014 with 100% observer coverage), but no recent interactions have been observed.

### Other Mortality

Short-beaked common dolphins may occasionally be injured or killed by recreational hook and line fisheries, similar to documented deaths and injuries for long-beaked common dolphins. Other risks may include exposure to underwater detonations in coastal waters, such as those documented for long-beaked common dolphins (Danil and St. Leger 2011).

## STATUS OF STOCK

The status of short-beaked common dolphins in Californian waters relative to OSP is not known. The observed increase in abundance of this species off California may reflect a distributional shift (Anganuzzi *et al.* 1993; Forney and Barlow 1998, Barlow 2016, Becker *et al*. 2020a), rather than an overall population increase due to growth. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality in 2015-2019 (≥ 30.5 animals) is estimated to be less than the PBR (8,889), and therefore they are not classified as a "strategic" stock under the MMPA. The total estimated fishery mortality and injury for short-beaked common dolphins is less than 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate.

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Revised 3/15/2022

# LONG-BEAKED COMMON DOLPHIN (*Delphinus delphis bairdii*): California Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Long-beaked common dolphins were recognized as a distinct species in the 1990s (Heyning and Perrin 1994; Rosel *et al*. 1994), but Cunha *et al.* (2015) suggests that *Delphinus capensis* is an invalid species and the Society of Marine Mammalogy now provisionally recognizes animals from this stock as the subspecies *Delphinus delphis bairdii*. In the future, it is possible that this stock will be recognized as a separate species (perhaps *D. bairdii*), as discussed by Dall (1873) and advocated by Banks and Brownell (1969), but further taxonomic analyses are required. Along the U.S. west coast, their distribution overlaps with that of the short-beaked common dolphin. Long-beaked common dolphins are commonly found within about 50 nmi of the coast, from Baja California (including the Gulf of California) northward to about central California (Figure 1). Along the west coast of Baja California, long-beaked common dolphins primarily occur inshore of the 250 m isobath, with very few sightings (<15%) in waters deeper than 500 meters (Gerrodette and Eguchi 2011). Stranding and sighting records indicate that the abundance of this species off California changes both seasonally and inter-annually (Heyning and Perrin 1994, Forney and Barlow 1998, Barlow 2016). Although long-beaked common dolphins are not restricted to U.S. waters, cooperative management agreements with Mexico exist only for the tuna purse seine fishery and not for other fisheries which may take this species (e.g. gillnet fisheries). For the MMPA stock assessment reports, there is a single Pacific management stock including only animals found within the U.S. Exclusive Economic Zone off California.

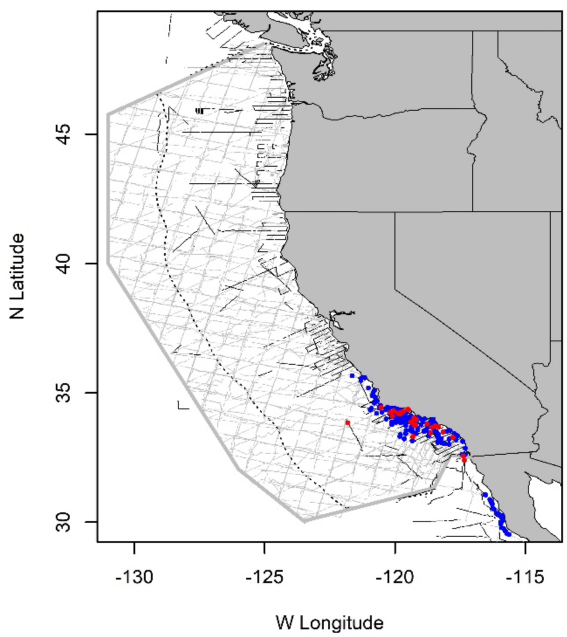


**Figure 1.** Long-beaked common dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 83,379 (CV=0.216) animals (Becker *et al.* 2020).

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate is 69,636 long-beaked common dolphins.

### Current Population Trend

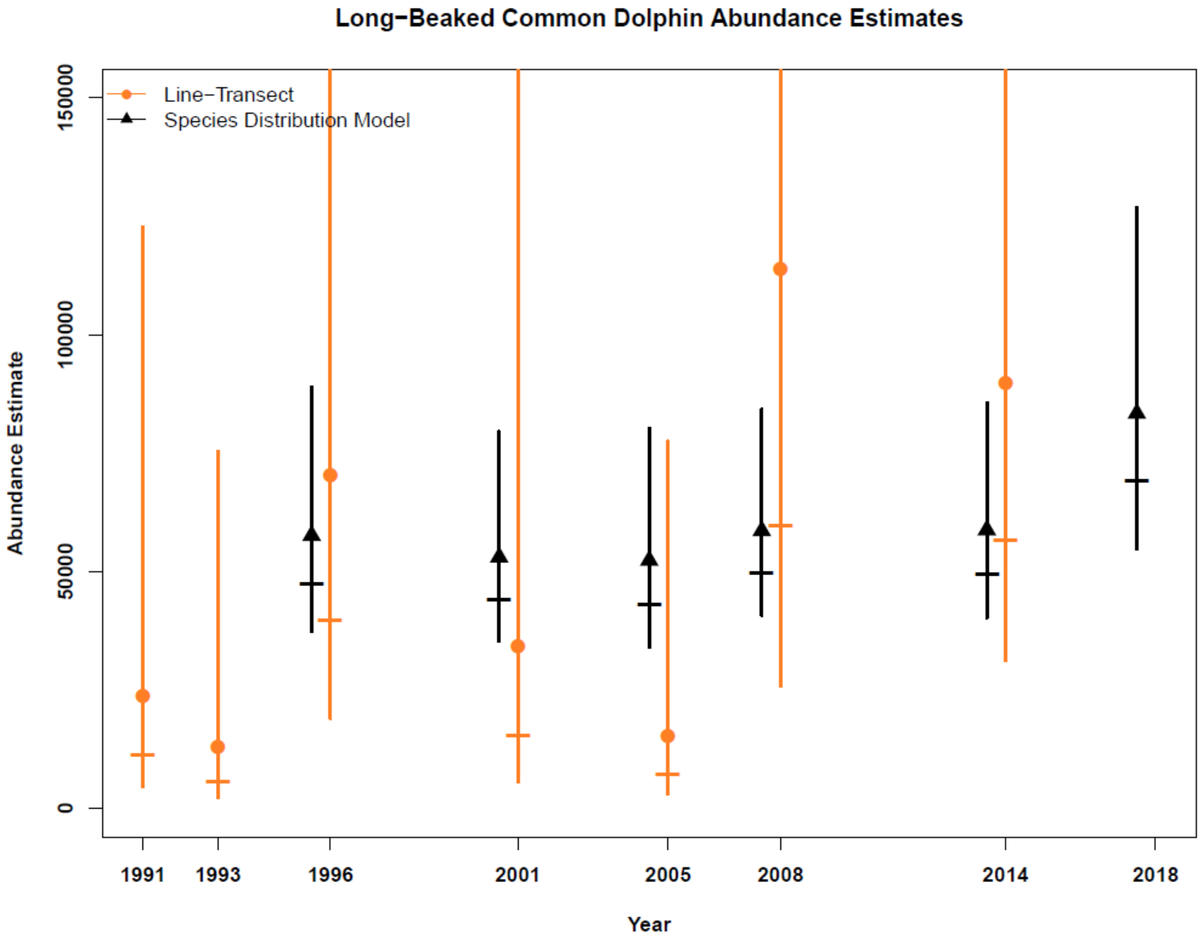


**Figure 2**. Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al.* 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington, but all research vessel sightings of this stock have occurred in California waters. Vertical bars indicate approximate 95% confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates. The y-axis has been truncated to provide the best display in variability in mean estimates between line-transect and species distribution models, due to relatively poor precision in the line-transect estimates. Upper 95% confidence limits for line-transect surveys in 1996, 2001, 2008, and 2014 not visible in the plot ranged between 205,000 and 503,000 animals.

California waters represent the northern limit for this stock and animals likely move between U.S. and Mexican waters. . The ratio of strandings of long-beaked to short-beaked common dolphin in southern California has varied, suggesting that the proportions of each species varies with ocean conditions (Heyning and Perrin 1994, Danil *et al*. 2010). There appears to be an increasing trend of long-beaked common dolphins in California waters over the last 30 years coincident with warming ocean conditions (Fig. 2), but a trend analysis for this stock has not been performed to date, while other stocks with more urgent conservation concerns are analyzed (e.g., Moore and Barlow 2011, 2013).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of current or maximum net productivity rates for long-beaked common dolphins.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (69,636) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with a mortality rate CV of 0.3 to 0.6; Wade and Angliss 1997), resulting in a PBR of 668 long-beaked common dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury for long-beaked common dolphins is shown in Table 1. More detailed information on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for long-beaked common dolphin in the California drift gillnet fishery for the five most recent years of monitoring, 2015-2019, averages 1.7 (CV= 0.60) per year (Carretta 2020). Stranding data are the primary source of documenting human-caused mortality for this stock during the most-recent 5-year period of 2015-2019 (Table 1). Human-caused mortality totals based on observed counts from strandings are negatively-biased because only a fraction of carcasses are detected (Carretta *et al*. 2016). Therefore, in this stock assessment report and others involving dolphins along the U.S. West Coast, human-related deaths and injuries counted from beach strandings along the outer U.S. West Coast are multiplied by a factor of 4 (including a coefficient of variation = 0.46 derived from the results of Carretta *et al*. 2016) to account for the non-detection of most carcasses (Carretta *et al*. 2016a). Applying this correction factor to the 21 stranded long-beaked common dolphins yields a minimum estimate of 84 fishery-related dolphin deaths, or an average of 16.8 annually (Table 1).

**Table 1.** Summary of available information on the incidental mortality and serious injury of long-beaked common dolphins (California Stock) in commercial fisheries that might take this species (Carretta 2021, Carretta *et al*. 2021). All observed entanglements resulted in the death of the animal. Coefficients of variation for mortality estimates are provided in parentheses, when available. n/a = information not available. Human-caused mortality values based on strandings recovered along the outer U.S. West Coast are multiplied by a correction factor of 4 to account for undetected mortality (Carretta *et al*. 2016a).

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality (and Serious Injury) | Estimated Annual Mortality (CV) | Mean Annual Takes (CV |
|---|---|---|---|---|---|---|
| **CA thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 2 | 8.7 (0.60) | 1.7 (0.60) |
| **CA Spot Prawn Trap Fishery** | Strandings | 2015-2019 | n/a | 1 | ≥ 4 | ≥ 0.8 (0.46) |
| **CA halibut/white seabass and other species set gillnet fishery** | Strandings | 2015-2019 | n/a | 2 | ≥ 8 (n/a) | ≥ 1.6 (0.46) |
| **Unidentified gillnet fishery interaction** | Strandings | 2015-2019 | n/a | 21 | ≥ 84 | ≥ 16.8 (0.46) |
| **Unidentified fishery interaction** | Strandings | 2015-2019 | n/a | 7 | ≥ 28 | ≥ 5.6 (0.46) |
| **Minimum total annual takes** (includes correction for unobserved beach strandings) | | | | | | ≥ 26.5 (0.39) |

### Other Mortality

Stranding records from 2015-2019 include two deaths resulting from hook and line fishery entanglements (Carretta *et al*. 2021). Applying the minimum correction factor of 4 to account for undetected mortality (Carretta *et al*. 2016a), yields an estimated 16 human-caused long-beaked common dolphin deaths from hook and line fisheries or 3.2 annually for 2015-2019.

'Unusual mortality events' of long-beaked common dolphins off California due to domoic acid toxicity have been documented by NMFS as recently as 2007. One study suggests that increasing anthropogenic $CO_2$ levels and ocean acidification may increase the toxicity of the diatom responsible for these mortality events (Tatters *et al*. 2012).

## STATUS OF STOCK

The status of long-beaked common dolphins in California waters relative to OSP is not known, and there are insufficient data to evaluate potential trends in abundance. Exposure to blast trauma resulting from underwater detonations is a local concern for this stock (Danil and St. Leger 2011), but population level impacts from such activities are unclear. In response to the 2011 event, the U.S. Navy has implemented training protocols to reduce the probability of blast trauma events occurring (Danil and St. Leger 2011). Long-beaked common dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality from commercial fisheries (≥ 26.5/yr) and other sources (3.2 /yr) is 29.7 long-beaked common dolphins. This does not exceed the PBR (668), and therefore they are not classified as a "strategic" stock under the MMPA. The average total fishery mortality and injury for long-beaked common dolphins (29.7/yr) is less than 10% of the PBR and therefore, is considered to be insignificant and approaching zero mortality and serious injury rate.

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# NORTHERN RIGHT-WHALE DOLPHIN (*Lissodelphis borealis*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Northern right-whale dolphins are endemic to temperate waters of the North Pacific Ocean. Off the U.S. west coast, they have been seen primarily in shelf and slope waters (Figure 1), with seasonal movements into the Southern California Bight (Leatherwood and Walker 1979; Dohl *et al*. 1980; 1983). Sighting patterns from aerial and shipboard surveys conducted in California, Oregon and Washington during different seasons (Green et al. 1992; 1993; Forney and Barlow 1998; Barlow 2016) suggest seasonal north-south movements, with animals found primarily off California during the colder water months and shifting northward into Oregon and Washington as water temperatures increase in late spring and summer. The southern end of this population's range is not well-documented, but during cold-water periods, they probably range into Mexican waters off northern Baja California. Genetic analyses have not found statistically significant differences between northern right-whale dolphins from the U.S. West coast and other areas of the North Pacific (Dizon *et al*. 1994); however, power analyses indicate that the ability to detect stock differences for this species is poor, given traditional statistical error levels (Dizon *et al*. 1995). Although northern right-whale dolphins are not restricted to U.S. territorial waters, there are currently no international agreements for cooperative management. For the Marine Mammal Protection Act (MMPA) stock assessment reports, there is a single management stock including only animals found within the U.S. Exclusive Economic Zone of California, Oregon and Washington.

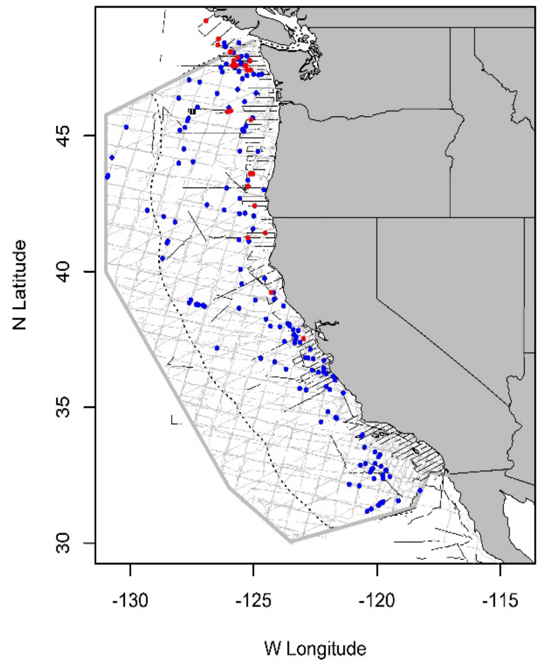


**Figure 1.** Northern right whale dolphin sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

The distribution of northern right-whale dolphins throughout this region is highly variable, apparently in response to oceanographic changes on both seasonal and inter-annual time scales (Forney and Barlow 1998, Barlow 2016). As oceanographic conditions vary, northern right-whale dolphins may spend time outside the U.S. Exclusive Economic Zone. Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2012, 2016, 2017, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 29,285 (CV=0.717) animals (Becker *et al.* 2020).

### Minimum Population Estimate

The log-normal 20th percentile of the 2018 abundance estimate is 17,024 northern right-whale dolphins (Becker *et al.* 2020).

**Current Population Trend**

The distribution and abundance of northern right whale dolphins off California, Oregon and Washington varies considerably at both seasonal and inter-annual time scales (Forney and Barlow 1998, Becker *et al*. 2012, 2020, Barlow 2016), but no long term trends have been identified (Figure 2).

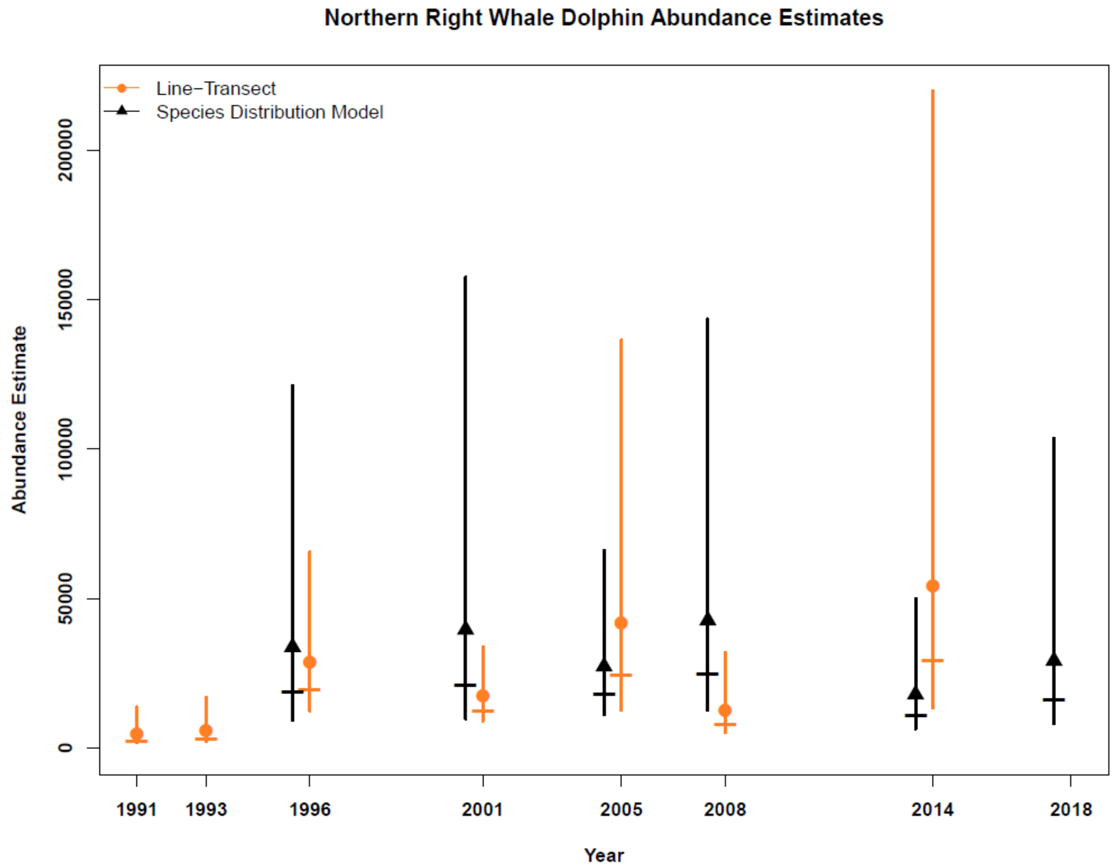


**Figure 2.** Abundance estimates and 95% confidence intervals from vessel-based line transect surveys (Barlow 2016) and species distribution models (Becker *et al*. 2020) within the California Current. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for northern right-whale dolphins off the U.S. west coast.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (17,024) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with a mortality rate CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 163 northern right-whale dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

A summary of recent fishery mortality and injury information for this stock of northern right-whale dolphin is shown in Table 1. More detailed information on these fisheries is provided in Appendix 1. Fishery-related deaths from strandings in Table 1 are multiplied by a correction factor of 4.0 to account for incomplete detection of carcasses (Carretta *et al.* 2016).

**Table 1.** Summary of available information on the incidental mortality and serious injury of northern right-whale dolphins (California/Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta 2021, Carretta *et al.* 2021, Jannot *et al.* 2018). All observed entanglements of northern right-whale dolphins resulted in the death of the animal. Coefficients of variation for mortality estimates are provided in parentheses.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2015-2019 | 21% | 7 | 28.1 (0.39) | 5.6 (0.39) |
| **WA, OR, CA domestic groundfish trawl fishery (includes at-sea hake and other limited-entry groundfish sectors)** | observer | 2012-2016 | 98% - 100% | 1 | 1 (n/a) | 0.2 (n/a) |
| **Unidentified fishery** | stranding | 2015-2019 | n/a | 1 | ≥ 4 (0.46) | ≥ 0.8 (0.46) |
| **Minimum total annual takes** | | | | | | ≥ 6.6 (0.33) |

## STATUS OF STOCK

The status of northern right-whale dolphins in California, Oregon and Washington relative to OSP is not known, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this species. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality from 2015-2019 (6.6 animals) is estimated to be less than the PBR (163), and therefore they are not classified as a "strategic" stock under the MMPA. The total fishery mortality and serious injury for northern right-whale dolphins does not exceed 10% of the calculated PBR and, therefore, can be considered to be insignificant and approaching zero mortality and serious injury rate.

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# Killer Whale (*Orcinus orca*): Eastern North Pacific Offshore Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales are observed worldwide from the tropics to polar regions (Leatherwood and Dahlheim 1978), although they prefer colder waters of both hemispheres, with greatest abundances found within 800 km of major continents (Mitchell 1975, Forney and Wade 2006). Near the west coast of North America, killer whales occur along the entire Alaskan coast (Braham and Dahlheim 1982, Hamilton *et al.* 2009), in British Columbia and Washington inland waterways (Bigg *et al.* 1990), and along the outer coasts of Washington, Oregon and California (Hamilton *et al.* 2009). Seasonal and year-round occurrence are noted for killer whales throughout Alaska (Braham and Dahlheim 1982) and in the intra-coastal waterways of British Columbia and Washington, where three ecotypes are recognized: 'resident', 'transient' and 'offshore' (Bigg *et al.* 1990, Ford *et al.* 1994), based on aspects of morphology, ecology, genetics and behavior (Ford and Fisher 1982; Baird and Stacey 1988; Baird *et al.* 1992, Hoelzel *et al.* 1998, Morin *et al.* 2010, Ford *et al.* 2014). Offshore killer whales are known from southern California waters north to the Aleutian Islands and are considered to represent a single network of socially-connected individuals (Dahlheim *et al.* 2008, Ford *et al.* 2014). Photographic matches of individuals between Dutch Harbor, Alaska and southern California waters near Dana Point are documented (Dahlheim *et al.* 2008). Offshore killer whales apparently do not mix with transient and resident killer whale stocks in these regions (Ford *et al.* 1994, Black *et al.* 1997). Studies indicate the 'offshore' type, although distinct from the other types ('resident' and 'transient'), appears to be more closely related genetically, morphologically, behaviorally, and vocally to 'resident' type killer whales (Black *et al.* 1997, Hoelzel *et al.* 1998, Morin *et al.* 2010). Global genetic studies suggest that residents and transient ecotypes warrant subspecies recognition (Morin *et al.* 2010) and each are currently listed as unnamed subspecies of *Orcinus orca* (Committee on Taxonomy 2018). Currently, the offshore killer whale ecotype is included under *Orcinus orca* (Committee on Taxonomy 2018).

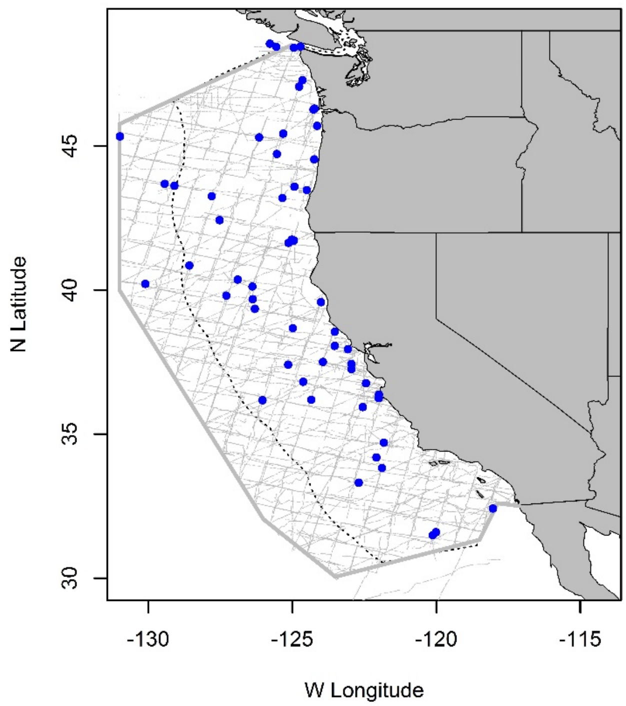


**Figure 1**. Sightings of killer whales (all ecotypes/stocks) encountered during Southwest Fisheries Science Center line-transect vessel surveys in the California Current ecosystem, 1991-2014.

Based on 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, (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 Alaska through California, and (8) the Hawaiian stock. The Stock Assessment

Reports for the Alaska Region contains data on Eastern North Pacific Alaska Resident, Eastern North Pacific Northern Resident and the Gulf of Alaska, Aleutian Islands, and Bering Sea, AT1, and West Coast Transient stocks.

## POPULATION SIZE

Population size of the eastern North Pacific stock of offshore killer whales is estimated with photo-ID mark-recapture methods at 300 whales (95% Highest Posterior Density Interval (HPDI) = 257–373, CV=0.10), including marked and unmarked individuals encountered from 1988-2012 (Ford *et al.* 2014). This study included 157 encounters of 355 distinct whales from the Aleutian Islands to southern California. The cumulative number of unique animals reported via a 'discovery curve' was not asymptotic, implying that additional individuals are undocumented. Most encounters (n=85) during the photo-ID study were from southeast Alaska and Vancouver Island, where survey effort was most intense. The fraction of this population utilizing U.S. waters is unknown and the number of animals using areas outside of the currently known geographic range (Aleutian Islands to southern California) is unknown.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20$^{th}$ percentile of the estimate ($N$=300, CV=0.1) reported by Ford *et al.* (2014), or 276 animals.

### Current Population Trend

The population trajectory for eastern North Pacific offshore killer whales is described as 'stable' by Ford *et al.* (2014). The stable designation includes considerations such as an estimated average annual survival rate of 0.98 (95% HPDI = 0.92–0.99) and annual recruitment rates of 0.02 (95% HPDI = 0–0.07) (Ford *et al.* 2014).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Annual recruitment rates of 2% (95% HPDI = 0 – 7%) were estimated by Ford *et al.* (2014) for offshore killer whales, based on a Bayesian mark-recapture model.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (276) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997), resulting in a PBR of 2.8 offshore killer whales.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Offshore killer whales have not been documented killed by anthropogenic sources in Alaska or U.S. west coast waters, but mortalities are likely to be undetected, given the offshore range of this ecotype. Ford *et al.* (2014) reports one offshore killer whale injury (severed dorsal fin) due to a vessel strike, but does not report a location or year. Offshore killer whales are likely vulnerable to the same anthropogenic threats (fishery interactions, vessel strikes, sonar) as other killer whale stocks.

**Table 1.** Data on incidental mortality and injury of Eastern North Pacific Offshore killer whales in commercial fisheries. No killer whale entanglements have been observed in the CA swordfish drift gillnet fishery since 1995, when a single whale was killed (Carretta *et al.* 2018a). The whale was genetically identified as a transient ecotype and is the only killer whale observed entangled in the fishery over a 27-year period (Carretta *et al.* 2017, 2018). Bycatch estimates for the fishery appear in Table 1 and are based on a bycatch model that pools all years of observer data, but does not include the observation of a transient killer whale.

| Fishery Name | Data Type | Years | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| CA thresher shark/swordfish drift gillnet | Observer | 2012<br>2013<br>2014<br>2015<br>2016 | 19%<br>37%<br>24%<br>20%<br>18% | 0<br>0<br>0<br>0<br>0 | 0<br>0<br>0<br>0<br>0 | 0 |
| **Minimum total annual takes** | | | | | | 0 |

## STATUS OF STOCK

The status of Eastern North Pacific offshore killer whales in relation to OSP is unknown. The estimated population size is described as 'stable' by Ford *et al.* (2014). No habitat issues are known to be of concern for this stock. The tendency for whales in this population to occur in large groups, sometimes between 50 -100 animals, combined with the small population size, raises concern that a relatively large fraction of the population faces exposure risk to such anthropogenic events as fishery interactions, vessel strikes, oil spills, or military sonar (Ford *et al.* 2014). Offshore killer whales are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. There has been no documented human-caused mortality of this stock but Ford *et al.* (2014) reported one injury due to a vessel strike. It is likely that undetected mortality and injury of killer whales from this stock occurs in gillnets and other fishing gear. Along the U.S. west coast, observations of the California swordfish drift gillnet fishery includes one *transient* killer whale entangled and killed during 8,845 fishing sets from 1990-2016 (Carretta *et al.* 2017a, Carretta *et al.* 2018). Documented injuries and mortalities of offshore killer whales due to anthropogenic sources are extremely rare, and the fishery most likely to interact with them along the U.S. west coast has not had a documented interaction in 27 years, therefore Eastern North Pacific offshore killer whales are not classified as a "strategic" stock under the MMPA. The total fishery mortality and serious injury for offshore killer whales is considered to be insignificant and approaching zero mortality and serious injury rate.

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# KILLER WHALE (*Orcinus orca*): Eastern North Pacific Southern Resident Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Killer whales occur in all oceans and seas (Leatherwood and 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 (Mitchell 1975, Forney and Wade 2006). Along the west coast of North America, killer whales occur along the entire Alaskan coast (Braham and Dahlheim 1982, Hamilton *et al.* 2009), in British Columbia and Washington inland waterways (Bigg *et al.* 1990), and along the outer coasts of Washington, Oregon and California (Hamilton *et al.* 2009). Seasonal and year-round occurrence is documented for killer whales throughout Alaska (Braham and 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 and Fisher 1982; Baird and Stacey 1988; Baird *et al.* 1992, Hoelzel *et al.* 1998, Morin *et al.* 2010, Ford *et al.* 2014). Genetic studies of killer whales globally suggest that residents and transient ecotypes warrant subspecies recognition (Morin *et al.* 2010) and each are currently listed as unnamed subspecies of *Orcinus orca* (Committee on Taxonomy 2018).

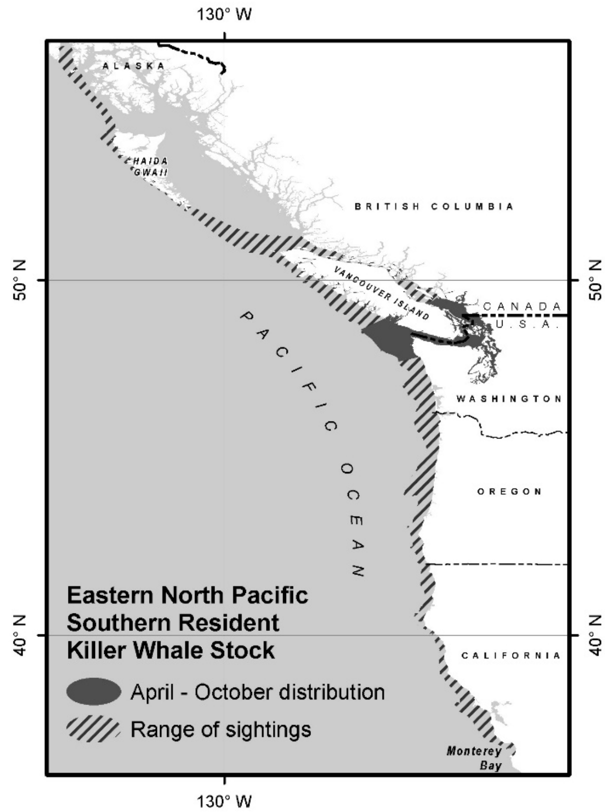


**Figure 1**. Approximate April - October distribution of the Eastern North Pacific Southern Resident killer whale stock (shaded area) and range of sightings (diagonal lines).

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 the 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 (J. Ford, pers. comm.). 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). 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 (see Fig. 1), 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 74 whales in 2021 (Fig. 2; Ford *et al.* 2000; Center for Whale Research 2021). The most recent census spanning 1 July 2020 through 1 July 2021 includes three new calves (J57, J58, L125), the death of a post-reproductive female, but does not include the death of an adult male in late summer of 2021, or two calves born in early 2022.

### 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 74 animals.

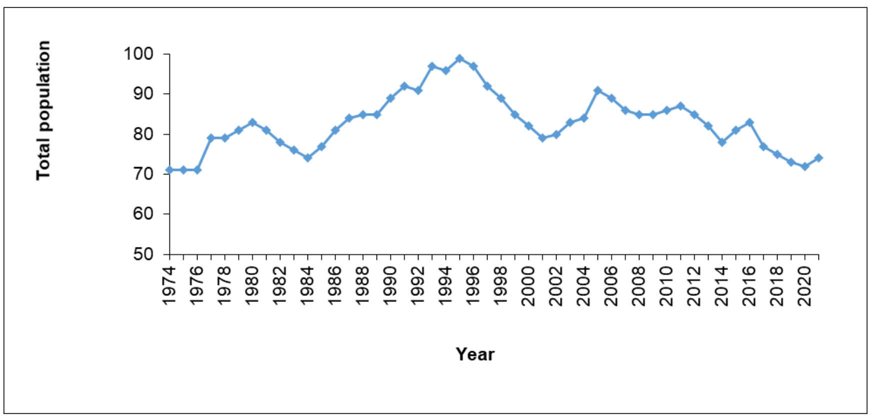


**Figure 2.** Population of Eastern North Pacific Southern Resident stock of killer whales, 1974-2021. Each year's count includes animals first seen and first missed; a whale is considered first missed the year after it was last seen alive (Ford *et al.* 2000; Center for Whale Research 2021).

### Current Population Trend

During the live-capture fishery that existed from 1967 to 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 74 animals as of the 2021 census (Ford *et al.* 2000; Center for Whale Research 2021).

## 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%, Olesiuk *et al.* 1990) and "probably represents a population at r-max (maximum rate of growth)." 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 potential biological removal (PBR).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (74) times one-half the maximum net growth rate for *Alaska* resident killer whales (½ of 3.5%) times 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.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### 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 2015-2020 (Carretta 2021, Carretta *et al.* 2021, Carretta *et al*. 2022).

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, WA, 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, 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, WA. 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 (NOAA 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 Hawaii noted the occurrence of human interactions across all age classes (Raverty et al. 2020).

**Habitat Issues**

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 DDT (Ebre 2002, Clark *et al.* 2009, Krahn *et al.* 2007, 2009, Lacy *et al.* 2017). 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, b). In 2011, vessel approach regulations were implemented to restrict vessels from approaching closer than 200m. 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 and 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 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*. 2018, 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 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 2019b), which spend more time in California waters where DDTs still persist in the marine ecosystem (Sericano *et al*. 2014).

**STATUS OF STOCK**

Total documented annual fishery mortality and serious injury for this stock from 2015-2020 (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 includes the death of L95 (fungal infection related to a satellite-tag) and J34 (vessel strike), or 0.4 whales annually, which exceeds the PBR (0.13). 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).

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# SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Two genetically and morphologically distinct short-finned pilot whale types are described in the Pacific ('Shiho' and 'Naisa') by Van Cise *et al*. (2016), which correspond to the northern and southern types (respectively) described off Japan (Kasuya *et al.* 1988; Wada 1988; Miyazaki and Amano 1994). Shiho type animals are largely confined to the California Current and eastern tropical Pacific, while Naisa type pilot whales occur in the central Pacific and Japan. Differences in body size, head shape, coloration, and number of teeth characterize Shiho and Naisa morphotypes, with the larger eastern Pacific Shiho type characterized by a rounder melon and distinct light saddle patch. Short-finned pilot whales were once common off Southern California, with an apparently resident population around Santa Catalina Island, as well as seasonal migrants (Dohl et al. 1980). After a strong El Niño event in 1982-83, short-finned pilot whales virtually disappeared from this region, and despite increased survey effort along the entire U.S. west coast, sightings and fishery takes are rare and have primarily occurred during warm-water years (Julian and Beeson 1998, Carretta *et al*. 2004, Barlow 2016). Figure 1 summarizes the sightings of short-finned pilot whales off the U.S. west coast from 1991-2014. Pilot whales in the California Current and eastern tropical Pacific likely represent a single population, based on a lack of differentiation in mtDNA (Van Cise *et al.* 2016), while animals in Hawaiian waters are characterized by unique haplotypes that are absent from eastern and southern Pacific samples, despite relatively large sample sizes from Hawaiian waters. For the Marine Mammal Protection Act (MMPA) stock assessment reports, short-finned pilot whales within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Hawaiian waters. Shiho-type short-finned pilot whales comprise the California, Oregon and Washington stock, and are covered in this report. Naisa-type short-finned pilot whales comprise the Hawaiian stock.

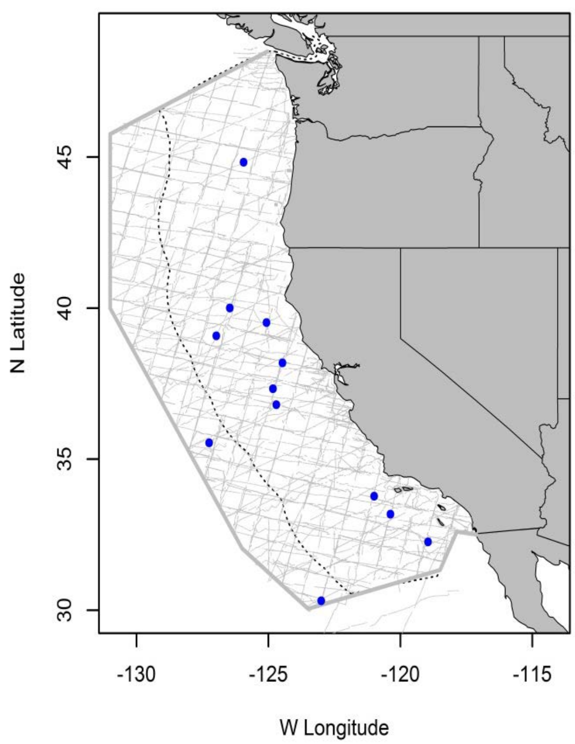


**Figure 1.** Short-finned pilot whale sightings made during shipboard surveys conducted off California, Oregon, and Washington, 1991-2014 (Barlow 2016). Dashed line represents the U.S. EEZ, thin gray lines indicate completed transect effort of all surveys combined.

## POPULATION SIZE

The abundance of short-finned pilot whales in this region is variable and may be influenced by prevailing oceanographic conditions (Forney 1997, Forney and Barlow 1998, Barlow 2016). Because animals may spend time outside the U.S. Exclusive Economic Zone as oceanographic conditions change, a multi-year average abundance estimate is the most appropriate for management within U.S. waters. The most recent estimate of short-finned pilot whale abundance is the geometric mean of estimates from 2008 and 2014 summer/autumn vessel-based line-transect surveys of California, Oregon, and Washington waters, or 836 (CV=0.79) animals (Barlow 2016). This estimate includes new correction factors for animals missed during the surveys.

### Minimum Population Estimate

The log-normal 20th percentile of the 2008-2014 geometric mean abundance estimate is 466 short-finned pilot whales.

### Current Population Trend

Following the virtual disappearance of short-finned pilot whales from California after the 1982-83 El Niño, they have been encountered infrequently and primarily during warm-water years, such as 1991, 1993, 1997, 2014, and 2015 (e.g., Carretta et al. 1995, Julian and Beeson 1998, Carretta et al. 2004, Barlow 2016). These patterns likely reflect large-scale, long-term movements of this species in response to changing oceanographic conditions. It is not known whether the animals sighted more recently are part of the same population that was documented off Southern California before the mid-1980s or a different wide-ranging pelagic population. Therefore, no inferences can be drawn regarding trends in abundance of short-finned pilot whales off California, Oregon and Washington.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for short-finned pilot whales off California, Oregon and Washington.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (466) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.48 (for a species of unknown status with bycatch mortality rate CV between 0.3 and 0.6; Wade and Angliss 1997), resulting in a PBR of 4.5 short-finned pilot whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of known fishery mortality and injury for this stock of short-finned pilot whales appears in Table 1. More data on these fisheries is provided in Appendix 1. The estimate of mortality and serious injury for short-finned pilot whale in the California drift gillnet fishery for the five most recent years of monitoring, 2010-2014, is 6 (CV= 0.39) whales, or an average of 1.2 per year (Carretta *et al*. 2017). Bycatch of short-finned pilot whales in the drift gillnet fishery is rarely-observed (14 animals in 8,711 observed sets), but high multivariate El Niño index values associated with warm-water years (Wolter and Timlin 2011) were identified as a significant predictor of bycatch (Carretta et al. 2017). Historically, short-finned pilot whales were also killed in squid purse seine operations off Southern California (Miller *et al*. 1983; Heyning *et al*. 1994), but these deaths occurred when pilot whales were still common in the region. An observer program in the squid purse seine fishery was initiated in 2004 and a total of 377 sets (<10% of effort) were observed through 2008 without a pilot whale interaction. 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.

**Table 1.** Summary of available information on the incidental mortality and serious injury of short-finned pilot whales (California/Oregon/Washington Stock) in commercial fisheries that might take this species (Carretta *et al*. 2017). Coefficients of variation for mortality estimates are provided in parentheses; n/a = not available.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Mortality | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2010<br>2011<br>2012<br>2013<br>2014 | 12%<br>20%<br>19%<br>37%<br>24% | 0<br>0<br>0<br>0<br>2 | 0<br>0<br>0<br>0<br>6 (0.39) | 1.2 (0.39) |
| **Market squid purse seine** | observer | 2004-2008 | <10% | 0 | 0 | 0 |
| **Minimum total annual takes** | | | | | | 1.2 ( 0.39) |

## STATUS OF STOCK

The status of short-finned pilot whales off California, Oregon and Washington in relation to OSP is unknown. They have declined in abundance in the Southern California Bight, since the 1982-83 El Niño, but the nature of these changes and potential habitat issues are not adequately understood. Short-finned pilot whales are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. The average annual human-caused mortality, 1.2 animals, is less than the PBR of 4.5, and therefore they are not classified as a "strategic" stock under the MMPA. Total annual human-caused mortality and serious injury for this stock is greater than 10 % of PBR; therefore, mortality and serious injury cannot be considered to be approaching a zero mortality and serious injury rate.

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Revised 3/14/2022

# BAIRD'S BEAKED WHALE (*Berardius bairdii*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Baird's beaked whales are distributed throughout deep waters and along the continental slopes of the North Pacific (Balcomb 1989, Macleod *et al.* 2006). They have been harvested and studied in Japanese waters, but little is known about this species elsewhere (Balcomb 1989). A second species of *Berardius*, *'minimus'*, has been described in the North Pacific, based on genetic (Morin *et al.* 2016) and morphological data (Yamada *et al.* 2019). The new species is darker and smaller than *B. bairdii*, with an apparently limited range between 40°N and 60°N, and 140°E and 160°W (Yamada *et al.* 2019). Sightings along the U.S. West Coast represent *B. bairdii*. Along the U.S. west coast, Baird's beaked whales have been seen primarily along the continental slope (Figure 1) from late spring to early fall. They are seen less frequently and are presumed to be farther offshore during the colder water months of November - April. For the Marine Mammal Protection Act (MMPA) stock assessment reports, Baird's beaked whales within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Alaskan waters.

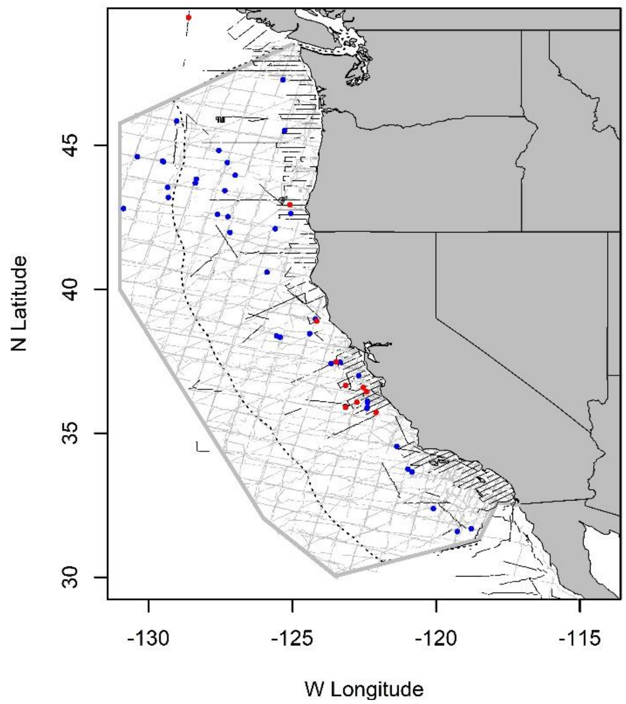


**Figure 1.** Baird's beaked whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Abundance of Baird's beaked whales has recently been estimated from Bayesian trend analyses (Moore and Barlow 2017) and species distribution models based on 1991-2014 and 1991-2018 line-transect data respectively (Becker *et al*. 2020, Figure 2). The differences in absolute abundance estimates for the trend-based estimates (Moore and Barlow 2017) and SDM estimates (Becker *et al*. 2020) are due to different values of *g(0)* that were used in the analyses. In both analyses, the overall *g(0)* is calculated as the average of sea-state specific *g(0)* values (Barlow 2015). Moore and Barlow (2017) assumed that in calm seas (Beaufort state = 0), *g(0)* = 0.47 (based on an estimate for *Mesoplodon*), with an average *g(0)* across all sea states of 0.30 - 0.37 across years. Becker *et al*. (2020) assumed *g(0)* = 1 in calm seas, with an average *g(0)* across effort segments > 0.5. The population size estimates from Becker *et al*. (2020) will be biased low, given the long synchronous dive times for *Berardius* groups, but an accurate correction for *Berardius* has not been estimated. The best estimate of abundance is taken as the most-recent estimate for 2018 from habitat-based species distribution models, or 1,363 (CV=0.533) whales.

### Minimum Population Estimate

The minimum population size estimate is taken as the lower 20$^{th}$ percentile of the 2018 abundance estimate, or 894 whales (Becker *et al.* 2018).

## Current Population Trend

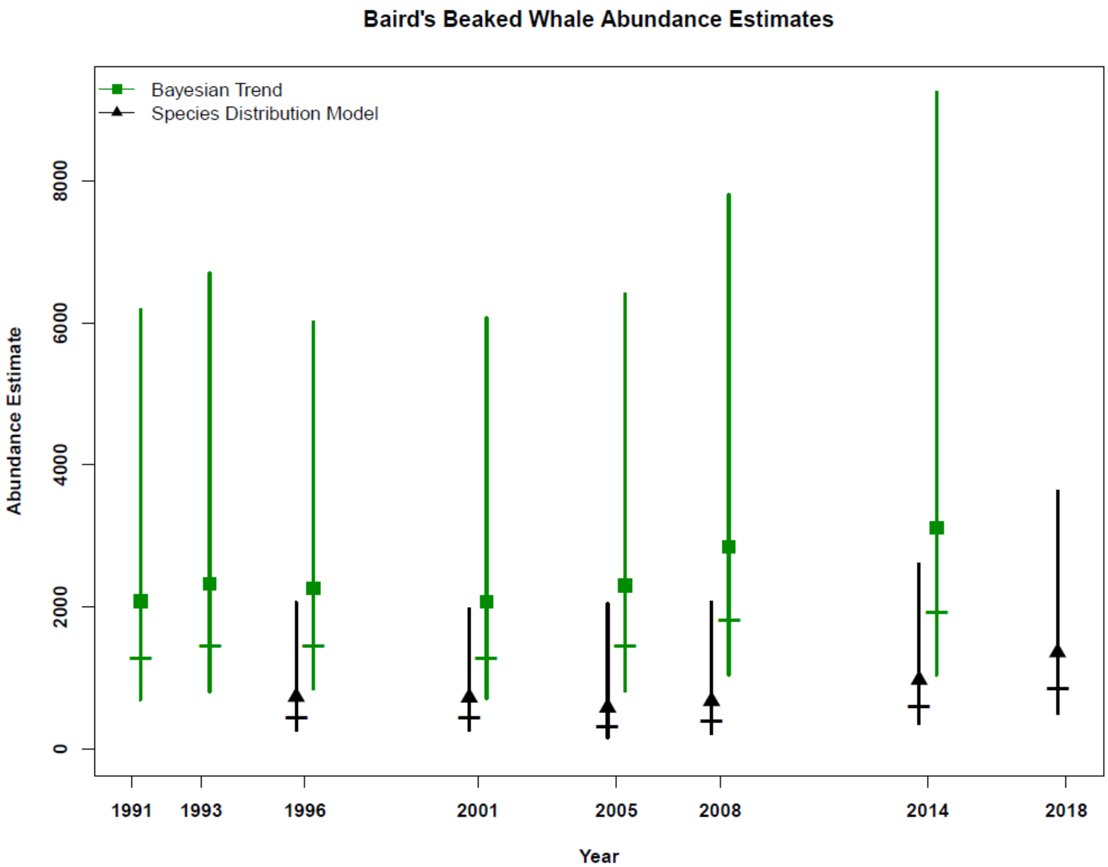


**Figure 2.** Baird's beaked whale abundance estimated from a Bayesian trend analysis (Moore and Barlow 2017) and habitat-based species distribution models based on 1991-2018 line-transect survey data (Becker *et al*. 2020). Vertical bars indicate approximate 95% log-normal confidence limits for Bayesian trend and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20th percentiles of mean estimates.

The population of Baird's beaked whales has remained stable or increased slightly, based on a Bayesian trend analysis by Moore and Barlow (2017, Figure 2). An annual growth rate geometric mean (λ) of 1.02 (SD = 0.03) was estimated based on the latest analysis, with 95% CRI ranging from 0.96 to 1.08 and a 72% chance of being positive (Moore and Barlow 2017). Estimates from species distribution models, while lower than the Bayesian estimates due to different g(0) values compared with Bayesian estimates, also show an apparent increase in abundance from 2008 to 2018 (Becker *et al*. 2020).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this species.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (894) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997), resulting in a PBR of 8.9 Baird's beaked whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The California large mesh drift gillnet fishery has been the only fishery known to interact with this stock. One Baird's beaked whale was incidentally killed in this fishery in 1994 (Julian and Beeson 1998), before acoustic pingers were first used in the fishery in 1996 (Barlow and Cameron 2003). Since 1996, no beaked whale of *any* species have been observed entangled or killed in this fishery (Carretta *et al.* 2008, Carretta 2021). Mean annual takes in Table 1 are based on 2015-2019 data. This results in an average estimated annual mortality of zero Baird's beaked whales (Carretta 2021).

**Table 1.** Summary of available information on the incidental mortality and injury of Baird's beaked whales (California/Oregon/Washington Stock) in commercial fisheries that might take this species. Coefficients of variation for mortality estimates are provided in parentheses. Mean annual takes are based on 2015-2019 data unless noted otherwise.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer data | 2015-2019 | 21% | 0 | 0 | 0 |
| **Minimum total annual takes** | | | | | | 0 |

### Other mortality

California coastal whaling operations killed 15 Baird's beaked whales between 1956 and 1970, and 29 additional Baird's beaked whales were taken by whalers in British Columbian waters (Rice 1974). One Baird's beaked whale stranded in California in 2016 and the cause of death was attributed to a vessel strike (Carretta *et al*. 2021). No other human-caused mortality has been reported for this stock for the period 2015-2019 (Carretta *et al.* 2021).

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to the remote nature of such activities and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. In Hawaiian waters, Faerber & Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt the foraging dives and echolocation activities of tagged beaked whales (Tyack *et al.* 2011). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al.* 2011). Fernández *et al.* (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al.* 2008, Carretta and Barlow 2011).

## STATUS OF STOCK

The status of Baird's beaked whales in California, Oregon and Washington waters relative to OSP is not known, and no abundance trend is evident (Moore and Barlow 2017). They are not listed as "threatened" or "endangered" under the Endangered Species Act nor designated as "depleted" under the MMPA. The average annual human-caused mortality during 2015-2019 is 0.2 animals/year (one vessel strike death). Because recent fishery and human-caused mortality is less than the PBR (8.9), Baird's beaked whales are not classified as a "strategic" stock under the MMPA. Moore and Barlow (2017) estimated that there was a 72% probability that this population had a positive growth rate over the period 1991-2014. Abundance estimates derived from species distribution models (Becker *et al.* 2020) also show an apparent increase between 2008 and 2018. The total fishery mortality and serious injury for this stock is zero and can be considered to be insignificant and approaching zero. The impacts of anthropogenic sound on beaked whales remains a concern (Barlow and Gisiner 2006, Cox *et al*. 2006, Hildebrand *et al*. 2005, Weilgart 2007).

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Revised 6/4/2018

# MESOPLODONT BEAKED WHALES (*Mesoplodon* spp.): California/Oregon/Washington Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Mesoplodont beaked whales are distributed throughout deep waters and along the continental slopes of the North Pacific Ocean. The six species known to occur in this region are: Blainville's beaked whale (*M. densirostris*), Perrin's beaked whale (*M. perrini*), Lesser beaked whale (*M. peruvianus*), Stejneger's beaked whale (*M. stejnegeri*), Gingko-toothed beaked whale (*M. gingkodens*), and Hubbs' beaked whale (*M. carlhubbsi*) (Mead 1989, Henshaw *et al.* 1997, Dalebout *et al.* 2002, MacLeod *et al.* 2006). Based on bycatch and stranding records in this region, it appears that Hubb's beaked whale is most commonly encountered (Carretta *et al.* 2008, Moore and Barlow 2013). Insufficient sighting records exist off the U.S. west coast (Figure 1) to determine any possible spatial or seasonal patterns in the distribution of mesoplodont beaked whales.

Until methods of distinguishing these six species at-sea are developed, the management unit must be defined to include all *Mesoplodon* stocks in this region. However, in the future, species-level management is desirable, and a high priority should be placed on finding means to obtain species-specific abundance information. For the Marine Mammal Protection Act (MMPA) stock assessment reports, three *Mesoplodon* stocks are defined: 1) all *Mesoplodon* species off California, Oregon and Washington (this report), 2) *M. stejnegeri* in Alaskan waters, and 3) *M. densirostris* in Hawaiian waters.

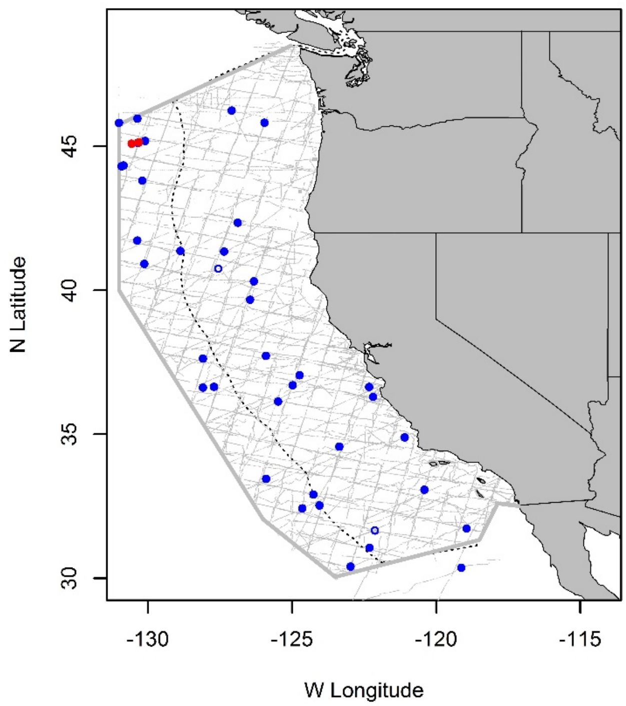


**Figure 1.** *Mesoplodon* beaked whale sightings based on shipboard surveys off California, Oregon and Washington, 1991-2014 . Key: ● = *Mesoplodon* spp.; ○ = identified *Mesoplodon densirostris*; ● = identified *Mesoplodon carlhubbsi*. Dashed line represents the U.S. EEZ, thin lines indicate completed transect effort of all surveys combined.

## POPULATION SIZE

A trend-based analysis of line-transect data from surveys conducted between 1991 and 2014 provides new estimates of Mesoplodon species abundance (Moore and Barlow 2017). The new estimate accounts for the proportion of unidentified beaked whale sightings likely to be Mesoplodon beaked whales and uses a correction factor for missed animals adjusted to account for the fact that the proportion of animals on the trackline missed by observers increases in rough seas. The trend-model analysis incorporates information from the entire 1991- 2014 time series for each annual estimate of abundance, and suggests evidence of an increasing abundance trend over that time (Moore and Barlow 2017), which is a reversal of the population decline reported by Moore and Barlow 2013. The authors note caveats to this observation: sea surface temperatures in 2014 were extremely warm in the California Current, with many previously undetected (and rarely detected) subtropical and tropical species occurring in the study area (Cavole et al.

2016). They hypothesize that an influx of warm-water Mesoplodon species into the California Current may have contributed to the higher estimate for 2014. They also reiterate that very few temperate species of Mesoplodon have stranded in recent years, a piece of supporting evidence for the previously observed population decline (Moore and Barlow 2013). The best estimate of Mesoplodon abundance is represented by the model-averaged estimate for 2014 (Moore and Barlow 2017). Based on this analysis, the best (50th percentile) estimate of abundance for all species of Mesoplodon species combined in 2014 in waters off California, Oregon and Washington is 3,044 (CV=0.54).

**Minimum Population Estimate**

The minimum population estimate (defined as the log-normal 20th percentile of the abundance estimate) for mesoplodont beaked whales in California, Oregon, and Washington is 1,967 animals.

**Current Population Trend**

Moore and Barlow (2013) provided strong evidence, based on line-transect survey data and the historical stranding record off the U.S. west coast, that the abundance of *Mesoplodon* beaked whales declined in waters off California, Oregon and Washington between 1991 and 2008 (Moore and Barlow 2013,). This apparent trend is reversed with the additional analysis of data collected in 2014, which includes the highest estimate of *Mesoplodon* abundance in the 1991-2014 time series (Moore and Barlow 2017, Figure 2). Statistical analysis of line-transect survey data from 1991 - 2014 indicates a 0.87 probability of an increase during this period, with the mean long-term growth rate estimate from a Markov model of $r = 0.03$ (SD = 0.07), with 95% CRI ranging from −0.10 to +0.18, indicating high uncertainty in long-term dynamics. Patterns in the historical stranding record alone provide limited information about beaked whale abundance trends, but the stranding record appears generally consistent rather than at-odds with results of the line-transect survey analysis. Regional stranding networks along the Pacific coast of the U.S. and Canada originated during the 1980s, and beach coverage and reporting rates are thought to have increased throughout the 1990s and in to the early 2000s. Therefore, for a stable or increasing population, an overall increasing trend in stranding reports between the 1980s and 2000s would be expected. In contrast, reported strandings for *M. carlhubbsi* and *M. stejnegeri* in the California Current region have declined monotonically since the 1980s.

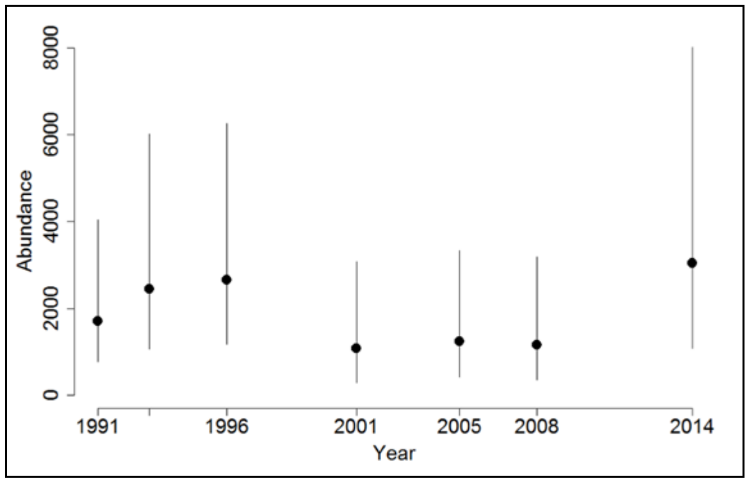


**Figure 2.** Abundance and trend estimates for mesoplodont beaked whales in the California Current, 1991-2014 (Moore and Barlow 2017). For each year, the Bayesian posterior median (●) is shown, along with 95% CRIs.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No information on current or maximum net productivity rates is available for mesoplodont beaked whales.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,967) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no known recent fishery mortality; Wade and Angliss 1997), resulting in a PBR of 20 mesoplodont beaked whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The California large mesh drift gillnet fishery has been the only fishery historically known to interact with *Mesoplodon* beaked whales in this region. Between 1990 and 1995, a total of eight *Mesoplodon* beaked whales (5 Hubb's beaked whales (*Mesoplodon carlhubbsi*), one Stejneger's beaked whale (*Mesoplodon stejnegeri*), and two unidentified whales of the genus *Mesoplodon* were observed entangled in approximately 3,300 sets (Julian and Beeson 1998, Carretta *et al.* 2008, Carretta *et al.* 2017). Following the introduction of acoustic pingers into this fishery (Barlow and Cameron 2003), no beaked whales of any species have been observed entangled in over 5,400 observed sets (Carretta *et al.* 2008, Carretta *et al.* 2017). New model-based estimates of bycatch based on regression trees result in a very small estimate of bycatch with high uncertainty for a single species (*M. carlhubbsi*), for the most recent 5-year period, 2011-2015 (0.5 whales total, CV=2.3), despite zero entanglements observed during that time period (Carretta *et al.* 2017). This is due to the bycatch model incorporating all 26 years of observer data in the estimation process (Carretta *et al.* 2017). Estimates for *M. stejnegeri* and unidentified *Mesoplodon* species are zero for the same time period.
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.

**Table 1.** Summary of available information on the incidental mortality and injury of *Mesoplodon* beaked whales (California/Oregon/Washington Stocks) in commercial fisheries that might take these species. Mean annual takes are based on 2011-2015 data unless noted otherwise.

| Fishery Name | Data Type | Year | Percent Observer Coverage | Observed Mortality | Estimated Annual Mortality | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer | 2011<br>2012<br>2013<br>2014<br>2015 | 20%<br>19%<br>37%<br>24%<br>20% | 0<br>0<br>0<br>0<br>0 | 0<br>0<br>0<br>0<br>0 | 0 (unidentified *Mesoplodon* and *M. Stejnegeri* only) |
| | | 2011-2015 | 24% | 0 | *M. carlshubbsi* only 0.5 (2.3) | *M. carlshubbsi* only 0.1 (2.3) |
| **Minimum total annual takes of all *Mesoplodon* beaked whales** | | | | | | 0.1 (2.3) |

### Other mortality

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to the remote nature of such activities and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. In Hawaiian waters, Faerber and Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt the foraging dives and echolocation activities of tagged beaked whales (Tyack *et al.* 2011, DeRuiter *et al.* 2013). Cuvier's beaked whales tagged and tracked during simulated mid-frequency sonar exposure showed avoidance reactions, including prolonged diving, cessation of echolocation click production associated with foraging, and directional travel away from the simulated sonar source (DeRuiter *et al.* 2013). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al.* 2011). Fernández *et* *al.* (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al.* 2008, Carretta and Barlow 2011).

## STATUS OF STOCKS

The status of mesoplodont beaked whales in California, Oregon and Washington waters relative to OSP is not known, and the population decline previously reported by Moore and Barlow (2013) is no longer apparent with the addition of 2014 survey data, which includes the highest estimate of *Mesoplodon* abundance in the 1991-2014 time series (Moore and Barlow 2017). The probability of a population increase over the time period 1991-2014 was estimated as 0.87 by Moore and Barlow (2017), but this is confounded by the fact that most *Mesoplodon* sightings are not identified to species, and thus, which species are driving the observed increase are not known. The previously-reported decline in abundance by Moore and Barlow (2013) (trend-fitted 2008 abundance at approximately 30% of 1991 levels) and current uncertainty in the long-term growth rate of this genus in the region warrants further investigation. If the relatively high 2014 abundance estimate was due to a temporary influx of subtropical and tropical species into the region, the remaining temperate species may be below their carrying capacity and may be depleted, based on the previous findings of Moore and Barlow (2013). Assessing changes in abundance for any species may also be confounded by distributional shifts within the California Current related to ocean-warming (Cavole *et al.* 2015). The average annual known human-caused fishery mortality between 2011 and 2015 is zero for *M. stejnegeri* and unidentified *Mesoplodon.* A negligible estimate of drift gillnet bycatch (0.1 whales annually) is predicted for *M. carlshubbsi* over the same time period, despite zero observations of entanglements in the fishery since 1994 (Carretta *et al.* 2017). None of the six species is listed as "threatened" or "endangered" under the Endangered Species Act and given the relative lack of bycatch in gillnet fisheries in this region, these stocks are considered non-strategic. It is likely that the difficulty in identifying these animals in the field will remain a critical obstacle to obtaining species-specific abundance estimates and stock assessments in the future. The impacts of anthropogenic sound on beaked whales remains a concern (Barlow and Gisiner 2006, Cox *et al.* 2006, Hildebrand *et al.* 2005, Weilgart 2007).

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Revised 05/30/2022

# CUVIER'S BEAKED WHALE (*Ziphius cavirostris*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Cuvier's beaked whales are distributed widely throughout deep waters of all oceans (MacLeod *et al.* 2006). Off the U.S. west coast, this species is the most commonly encountered beaked whale (Figure 1). No seasonal changes in distribution are apparent from stranding records, and morphological evidence is consistent with the existence of a single eastern North Pacific population from Alaska to Baja California, Mexico (Mitchell 1968). For the Marine Mammal Protection Act (MMPA) stock assessment reports, Cuvier's beaked whales within the Pacific U.S. Exclusive Economic Zone are divided into three discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), 2) Alaskan waters, and 3) Hawaiian waters.

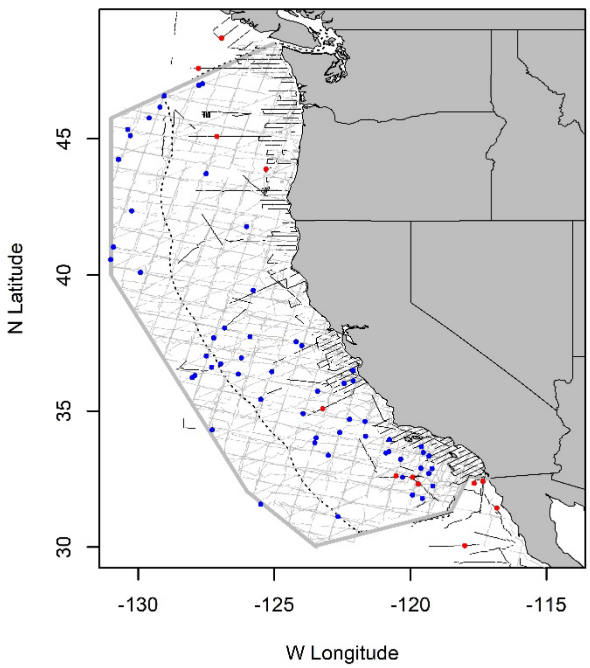


**Figure 1.** Cuvier's beaked whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Although Cuvier's beaked whales are sighted along the U.S. west coast on most vessel-based line transect surveys, the rarity of sightings results in imprecise abundance estimates (Barlow 2016, Moore and Barlow 2017). Furthermore, survey data includes a large number of unidentified beaked whale sightings that are probably either *Mesoplodon* sp. or Cuvier's beaked whales (*Ziphius cavirostris*). A trend-based analysis of line-transect data from surveys conducted between 1991 and 2014 provided a range of estimates from 2,242 to 4,860 Cuvier's beaked whales with coefficients of variation between 0.59 and 0.67 (Moore and Barlow 2017). Barlow *et al*. (2021) developed a new method for estimating Cuvier's beaked whale density and abundance, using a modified point-transect distance sampling framework applied to passive acoustic data collected on drifting hydrophone arrays. They estimated the abundance of Cuvier's beaked whales in 2016 to be 5,454 whales (CV=0.27, 95% CI = 3,151 – 8,907), which is higher than any previous line-transect estimate, with better precision. Barlow *et al*. (2021) note that the largest source of uncertainty in their estimates is estimation of the effective area surveyed by floating hydrophones.

### Minimum Population Estimate

The minimum population estimate is based on the lower 20th percentile of the posterior distribution reported in Barlow *et al*. (2021), or 4,214 whales.

### Current Population Trend

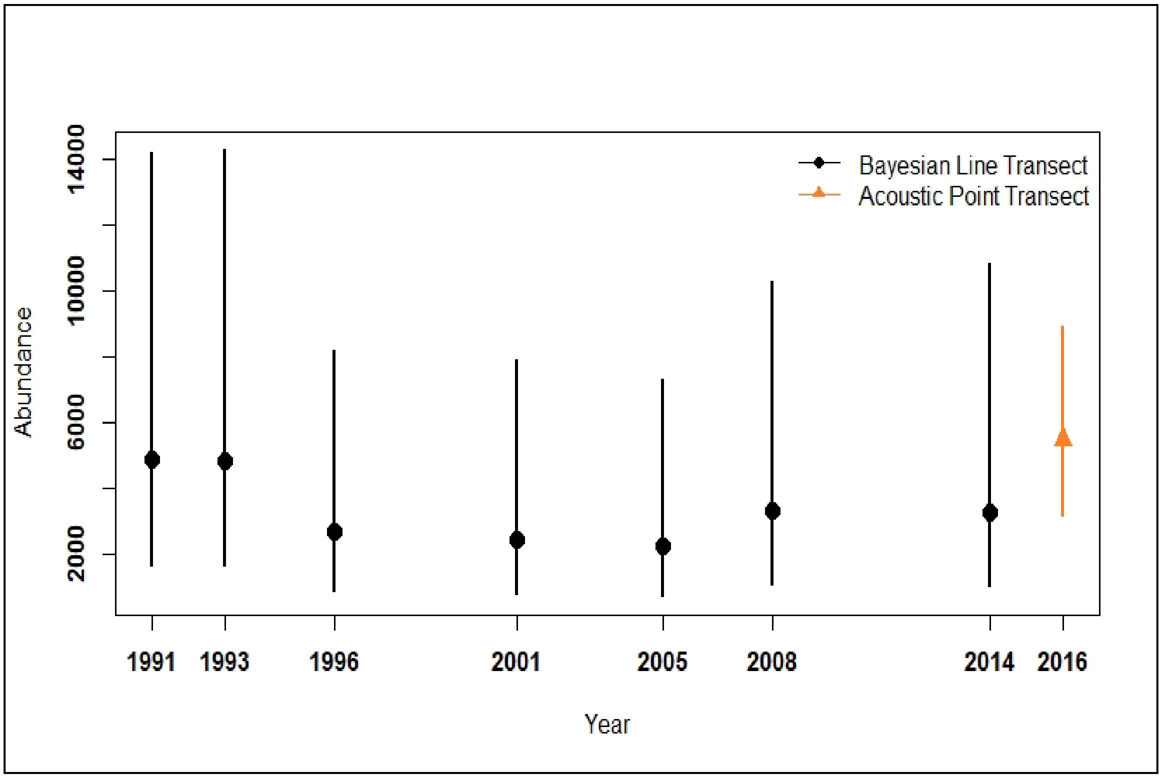


**Figure 2.** Abundance estimates for Cuvier's beaked whales in the California Current, 1991-2016 (Moore and Barlow 2017, Barlow *et al*. 2021). For each year, the Bayesian posterior median (●) and mean (▲) abundance estimates are shown, along with 95% CRIs.

There is substantial evidence, based on line-transect survey data and the historical stranding record off the U.S. west coast, that the estimated abundance of Cuvier's beaked whales in waters off California, Oregon and Washington was lower between 2001 and 2014 than in the early 1990s (Moore and Barlow 2013, 2017, Fig. 2). Statistical analysis of line-transect survey data from 1991 - 2014 indicates a 0.85 probability of decline during this period (Moore and Barlow 2017), with the mean annual rate of population change estimated to have been -3.0% per year (95% CRI: -10% to +3%, regression model results), although abundance throughout the 2000s appears stable, and estimates have not been updated following the 2018 survey. The 2016 acoustic based estimate represents the highest point estimate of the time series (Fig. 2), but it is unknown if this reflects differences in methodology between line transect and acoustic methods, a true increase in abundance, or both. Patterns in the historical stranding record alone provide limited information about beaked whale abundance trends, but the stranding record appears generally consistent rather than at-odds with results of the line-transect survey analysis. Regional stranding networks along the Pacific coast of the U.S. and Canada originated during the 1980s, and beach coverage and reporting rates are thought to have increased throughout the 1990s and in to the early 2000s. Therefore, for a stable or increasing population, an overall increasing trend in stranding reports between the 1980s and 2000s would be expected. Patterns of Cuvier's beaked whale strandings data are highly variable across stranding network regions, but an overall increasing trend from the 1980s through 2000s is not evident within the California Current area, contrary to patterns for Baird's beaked whales (Moore and Barlow 2013) and for cetaceans in general (e.g., Norman *et al.* 2004, Danil *et al.* 2010). Taylor *et al*. (2007) highlighted difficulties in assessing trends in abundance for beaked whales from visual surveys due to the rarity of sightings and relative imprecision of estimates. The addition of a new acoustically-derived abundance estimate for 2016 that is higher than all previous line-transect estimates (Barlow *et al*. 2021) does not aid in the assessment of trends for this stock, as there are no comparable acoustic estimates that overlap with the line-transect estimates. Barlow *et al*. (2021) note the great potential to estimate trends in abundance with greater precision using acoustic methods, based on documenting changes in acoustic encounter rates through time.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this species.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (4,214), times one half the default maximum net growth rate for cetaceans (½ of 4%), times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997), resulting in a PBR of 42 Cuvier's beaked whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The California swordfish drift gillnet fishery has been the only fishery historically known to interact with this stock. Prior to the introduction of acoustic pingers into the fishery in 1996, there were 21 Cuvier's beaked whales observed entangled in approximately 3,300 drift gillnet fishery sets: 1992 (six animals), 1993 (three), 1994 (six) and 1995 (six) (Julian and Beeson 1998). Since acoustic pinger use, no observed Cuvier's beaked whale entanglements have been observed in over 5,900 observed fishing sets (Barlow and Cameron 2003, Carretta *et al.* 2008, Carretta and Barlow 2011, Carretta 2021). New model-based estimates of bycatch based on regression trees identify the use of acoustic pingers, latitude, and sea surface temperature as three variables influencing the bycatch of Cuvier's beaked whales in the fishery (Carretta 2021). Mean annual takes in Table 1 are based on 2015-2019 data. Although no Cuvier's beaked whales were observed entangled in the most recent 5-year time period, bycatch models produced a negligible estimate of bycatch for this 5-year period of 0.3 (CV=1.4) whales. This results in an average estimated annual mortality of 0.06 (CV= 1.4) Cuvier's beaked whales.

**Table 1.** Summary of available information on the incidental mortality and serious injury of Cuvier's beaked whales (California/ Oregon/Washington Stock) in commercial fisheries that might take this species. Mean annual takes are based on 2016-2020 data.

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality + ReleasedAlive | Estimated Annual Mortality / Mortality + Entanglements | Mean Annual Takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer data | 2016-2020 | 21% | 0 | 0.3 (1.4) | 0.06 (1.4) |
| **Minimum total annual takes** | | | | | | 0.06 (1.4) |

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.

### Other mortality

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to the remote nature of such activities and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. In Hawaiian waters, Faerber & Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt the foraging dives and echolocation activities of tagged beaked whales (Tyack *et al.* 2011, DeRuiter *et al.* 2013). Cuvier's beaked whales tagged and tracked during simulated mid-frequency sonar exposure showed avoidance reactions, including prolonged diving, cessation of echolocation click production associated with foraging, and directional travel away from the simulated sonar source (DeRuiter *et al.* 2013). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al.* 2011). Fernández *et al.* (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al.* 2008, Carretta and Barlow 2011, Carretta 2022).

**STATUS OF STOCK**

The status of Cuvier's beaked whales in California, Oregon and Washington waters relative to OSP is unknown, but Moore and Barlow (2013) indicated a substantial likelihood of population decline in the California Current since the early 1990s, at a mean rate of -2.9% per year, which corresponds to trend-fitted abundance levels in 2008 being at 61% of 1991 levels. New trend estimates also indicate evidence of a population decline between 1990 and 2014, with an 85% probability of a decline at a mean rate of -3.0% per year (Moore and Barlow 2017). Cuvier's beaked whales are not listed as "threatened" or "endangered" under the Endangered Species Act, nor designated as "depleted" under the MMPA. However, the long-term decline in Cuvier's beaked whale abundance in the California Current reported by Moore and Barlow (2013, 2017), and the degree of decline (trend-fitted 2014 abundance at approximately 67% of 1991 levels) suggest that this stock may be below its carrying capacity. Assessing changes in abundance for any species may also be confounded by distributional shifts within the California Current related to ocean-warming (Cavole *et al.* 2015). Given that the stock is not currently ESA listed or designated as depleted, and human-caused mortality is below PBR, it is not strategic. Moore and Barlow (2013) ruled out bycatch as a cause of the decline in Cuvier's beaked whale abundance and suggest that impacts from anthropogenic sounds such as naval sonar and deepwater ecosystem changes within the California Current are plausible hypotheses warranting further investigation. The average annual estimated human-caused mortality between 2016 and 2020 is negligible (0.06 whales annually) and reflects a small probability that true bycatch in this fishery may be greater than the zero observed from approximately 5,900 fishing sets since 1996 (Carretta 2021). The total fishery mortality and serious injury for this stock is less than 10% of the PBR and thus is considered to be insignificant and approaching zero. The impacts of anthropogenic sound on beaked whales remains a concern (Barlow and Gisiner 2006, Cox *et al.* 2006, Hildebrand *et al.* 2005, Weilgart 2007).

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Revised 2/10/2017

# PYGMY SPERM WHALE (*Kogia breviceps*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Pygmy sperm whales are distributed throughout deep waters and along the continental slopes of the North Pacific and other ocean basins (Ross 1984; Caldwell and Caldwell 1989). Along the U.S. west coast, sightings of this species and of animals identified only as *Kogia* sp. have been rare (Figure 1). However, this probably reflects their pelagic distribution, small body size and cryptic behavior, rather than a measure of rarity. Strandings of pygmy sperm whales in this region are known from California, Oregon and Washington (Roest 1970; Caldwell and Caldwell 1989; NMFS, Northwest Region, unpublished data; NMFS, Southwest Region, unpublished data), while strandings of dwarf sperm whales (*Kogia sima*) are rare in this region. At-sea sightings in this region have all been either of pygmy sperm whales or unidentified *Kogia* sp. Available data are insufficient to identify any seasonality in the distribution of pygmy sperm whales, or to delineate possible stock boundaries. For the Marine Mammal Protection Act (MMPA) stock assessment reports, pygmy sperm whales within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Hawaiian waters.

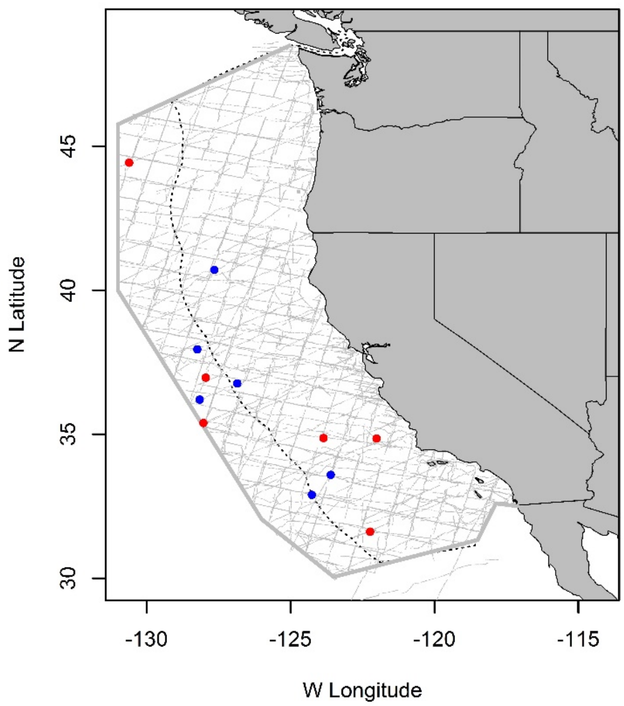


**Figure 1.** *Kogia* sightings based on shipboard surveys off California, Oregon and Washington, 1991-2014. Key: ● = *Kogia breviceps*, ● = *Kogia* spp. Dashed line represents the U.S. EEZ, thin lines indicate completed transect effort of all surveys combined.

## POPULATION SIZE

Most sightings of *Kogia* in the California Current are only identified to genus due to their cryptic nature, but based on positively-identified sightings from previous surveys and historical stranding data, most of these sightings were probably pygmy sperm whales; *K. breviceps*. The rarity of sightings likely reflects the cryptic nature of this species (they are detected almost exclusively in extremely calm sea conditions), rather than an absence of animals in the region. The best estimate of abundance for this stock is the geometric mean of 2008 and 2014 shipboard line-transect surveys, or 4,111 (CV=1.12) animals. This estimate is considerably higher than previous abundance estimates for the genus *Kogia* and results from a new and lower estimate of *g(0)*, the trackline detection probability (Barlow 2015). Only 3% of *Kogia* groups were estimated to have been detected on the trackline during 1991-2014 surveys (Barlow 2016).

### Minimum Population Estimate

The minimum population estimate is taken as the log-normal 20th percentile of the 2008 and 2014 average abundance estimate for California, Oregon, and Washington waters, or 1,924 animals.

### Current Population Trend

Due to the rarity of sightings of this species on surveys along the U.S. West coast, no information exists regarding trends in abundance of this population.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this species.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,924) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality during the last five years; Wade and Angliss 1997), resulting in a PBR of 19 pygmy sperm whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

A summary of recent fishery mortality and injury for pygmy sperm whales and unidentified Kogia, which may have been pygmy sperm whales, is shown in Table 1. In the California swordfish drift gillnet fishery (the only U.S. west coast fishery likely to interact with Kogia), no mortality of pygmy sperm whales or unidentified Kogia was observed during the most recent five years of monitoring (Carretta et al. 2017). Over 8,600 fishing sets have been monitored in the California swordfish drift gillnet fishery between 1990 and 2014 and only 2 pygmy sperm whales were observed entangled (Carretta et al. 2017). Both animals were entangled in years that predated the use of acoustic pingers in the fishery to reduce bycatch (Barlow and Cameron 2003), but the small sample size of Kogia breviceps bycatch in the fishery precludes any conclusions regarding the effectiveness of acoustic pingers in reducing bycatch of this species (Carretta and Barlow 2011). Mean annual takes in Table 1 are based on 2010-2014 data. This results in an average estimated annual mortality of zero pygmy sperm whales.

One pygmy sperm whale stranded in California in 2002 with evidence that it died as a result of a shooting (positive metal detector scan). Due to the cryptic and pelagic nature of this species, it is likely that the shooting resulted from an interaction with an unknown entangling net fishery. Although there are no records of fishery-related strandings of pygmy sperm whales along the U.S. west coast in recent years (Carretta *et al*. 2013, 2014, 2015, 2016a), compared with other more coastal cetaceans, the probability of a pygmy sperm whale carcass coming ashore and being detected would be quite low (Carretta *et al*. 2016b).

### Other mortality

Unknown levels of injuries and mortality of pygmy sperm whales may occur as a result of anthropogenic sound, such as military sonars. Atypical multispecies mass strandings, sometimes involving pygmy and/or dwarf sperm whales have been associated with military sonar use. One 1988 event from the Canary Islands included 2 pygmy sperm whales and the species *Ziphius cavirostris* and *Hyperoodon ampullatus* (reviewed in D'Amico *et al*. 2009). Another mass stranding and unusual mortality event (UME) in North Carolina, USA in 2005 included 2 dwarf sperm whales, in addition to 33 short-finned pilot whales and a minke whale (Hohn *et al*. 2006). This UME coincided in time and space with military activity using mid-frequency active sonar, although the authors note that a definitive association between the UME and sonar use is lacking (Hohn *et al.* 2006). Such injuries or mortality to pygmy sperm whales would rarely be documented, due to the remote nature of many of these activities and the low probability that an injured or dead pygmy sperm whale would strand.

## STATUS OF STOCK

The status of pygmy sperm whales in California, Oregon and Washington waters relative to OSP is not known, and there are insufficient data to evaluate potential trends in abundance. Although the impacts of anthropogenic sounds such as sonar are often focused on beaked whales (Barlow and Gisiner 2006), the impacts of such sounds on deep-diving pygmy beaked whales also warrants concern. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Given the rarity of sightings and lack of recent documented fishery interactions in U.S. west coast waters, pygmy sperm whales are not classified as a "strategic" stock under the MMPA.

**Table 1.** Summary of available information on the incidental mortality and injury of pygmy sperm whales and unidentified *Kogia* sp. (California/Oregon/Washington Stock) in commercial fisheries that might take this species. Coefficients of variation for mortality estimates are provided in parentheses. Mean annual takes are based on 2010-2014 data unless noted otherwise (Carretta *et al.* 2017).

| Fishery Name | Data Type | Year(s) | Percent Observer Coverage | Observed Mortality *K. breviceps /Kogia* sp. | *Estimated Annual Mortality of K. breviceps/Kogia* sp. | *Mean Annual Takes (CV in parentheses)* |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer data | 2010<br>2011<br>2012<br>2013<br>2014 | 12%<br>20%<br>19%<br>37%<br>24% | 0<br>0<br>0<br>0<br>0 | 0 | 0 |
| **Minimum total annual takes** | | | | | | 0 |

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Revised 2/10/2017

# DWARF SPERM WHALE (*Kogia sima*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Dwarf sperm whales are distributed throughout deep waters and along the continental slopes of the North Pacific and other ocean basins (Caldwell and Caldwell 1989; Ross 1984). This species was only recognized as being distinct from the pygmy sperm whale in 1966 (Handley, 1966), and early records for the two species are confounded. Along the U.S. west coast, no at-sea sightings of this species have been reported; however, this may be partially a reflection of their pelagic distribution, small body size and cryptic behavior. A few sightings of animals identified only as *Kogia* sp. have been reported (Figure 1), and some of these may have been dwarf sperm whales. At least five dwarf sperm whales stranded in California between 1967 and 2000 (Roest 1970; Jones 1981; J. Heyning, pers. comm.; NMFS, Southwest Region, unpublished data), and one stranding is reported for western Canada (Nagorsen and Stewart 1983). It is unclear whether records of dwarf sperm whales are so rare because they are not regular inhabitants of this region, or merely because of their cryptic habits and offshore distribution. Available data are insufficient to identify any seasonality in the distribution of dwarf sperm whales, or to delineate possible stock boundaries. For the Marine Mammal Protection Act (MMPA) stock assessment reports, dwarf sperm whales within the Pacific U.S. Exclusive Economic Zone are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington (this report), and 2) Hawaiian waters.

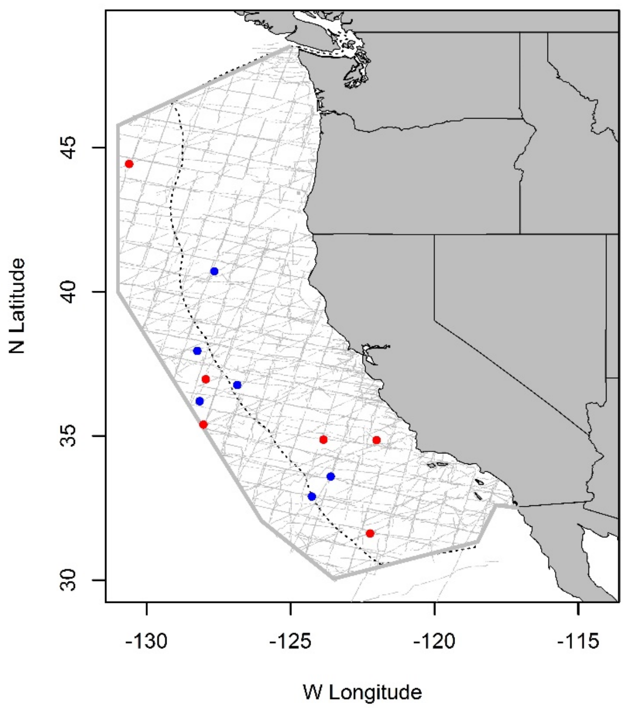


**Figure 1.** *Kogia* sightings based on shipboard surveys off California, Oregon and Washington, 1991-2014. Key: ● = *Kogia breviceps*; ● = *Kogia* spp. Dashed line represents the U.S. EEZ, thin lines indicate completed transect effort of all surveys combined.

## POPULATION SIZE

No information is available to estimate the population size of dwarf sperm whales off the U.S. west coast, as no sightings of this species have been documented despite numerous vessel surveys of this region (Barlow 1995; Barlow and Gerrodette 1996; Barlow and Forney 2007; Forney 2007; Barlow 2010, Barlow 2016). Based on previous sighting surveys and historical stranding data, it is likely that recent ship survey sightings were of pygmy sperm whales; *K. breviceps*.

### Minimum Population Estimate

No information is available to obtain a minimum population estimate for dwarf sperm whales.

### Current Population Trend

Due to the rarity of records for this species along the U.S. West coast, no information exists regarding trends in abundance of this population.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No information on current or maximum net productivity rates is available for this species.

## POTENTIAL BIOLOGICAL REMOVAL

Based on this stock's unknown status and growth rate, the recovery factor ($F_r$) is 0.5, and ½$R_{max}$ is the default value of 0.02. However, due to the lack of abundance estimates for this species, no potential biological removal (PBR) can be calculated.

## ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The fishery most likely to interact with dwarf sperm whales in the California Current is the swordfish drift gillnet fishery. There have been no observed dwarf sperm whale entanglements in over 8,600 monitored fishing sets from 1990 to 2014 (Carretta *et al.* 2017). Although there are no records of fishery-related strandings of dwarf sperm whales along the U.S. west coast in recent years (Carretta *et al*. 2013, 2014, 2015, 2016a), compared with other more coastal cetaceans, the probability of a dwarf sperm whale carcass coming ashore and being detected would be quite low (Carretta *et al*. 2016b).

**Table 1.** Summary of available information on the incidental mortality and injury of dwarf sperm whales and unidentified *Kogia* sp. (California/Oregon/Washington Stock) in commercial fisheries that might take this species. Coefficients of variation for mortality estimates are provided in parentheses. Mean annual takes are based on 2010-2014 data.

| **Fishery Name** | **Data Type** | **Year(s)** | **Percent Observer Coverage** | **Observed Mortality *K. breviceps* /*Kogia* sp.** | ***Estimated Annual Mortality of K. breviceps/Kogia* sp.** | ***Mean Annual Takes (CV in parentheses)*** |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | observer data | 2010-2014 | 12% to 37% | 0 | 0 | 0 |
| **Minimum total annual takes** | | | | | | 0 |

### Other Mortality

Unknown levels of injuries and mortality of dwarf sperm whales may occur as a result of anthropogenic sound, such as military sonars. Atypical multispecies mass strandings, sometimes involving dwarf and/or pygmy sperm whales have been associated with military sonar use. One 1988 event from the Canary Islands included 2 pygmy sperm whales and the species *Ziphius cavirostris* and *Hyperoodon ampullatus* (reviewed in D'Amico *et al.* 2009). Another mass stranding and unusual mortality event (UME) in North Carolina, USA in 2005 included 2 dwarf sperm whales, in addition to 33 short-finned pilot whales and a minke whale (Hohn *et al.* 2006). This UME coincided in time and space with military activity using mid-frequency active sonar, although the authors note that a definitive association between the UME and sonar use is lacking (Hohn *et al*. 2006). Such injuries or mortality would rarely be documented, due to the remote nature of many of these activities and the low probability that an injured or dead dwarf sperm whale would strand.

## STATUS OF STOCK

The status of dwarf sperm whales in California, Oregon and Washington waters relative to OSP is not known, and there are insufficient data to evaluate potential trends in abundance. Although the impacts of anthropogenic sounds such as sonar are often focused on beaked whales (Barlow and Gisiner 2006), the impacts of such sounds on deep-diving dwarf beaked whales also warrants concern. They are not listed as "threatened" or "endangered" under the Endangered Species Act nor as "depleted" under the MMPA. Given that this species rarely occurs off the U.S. west coast and a lack of recent documented fishery mortality, dwarf sperm whales off California, Oregon and Washington are not classified as a "strategic" stock under the MMPA.

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Revised 5/14/2024

# SPERM WHALE (*Physeter macrocephalus*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Sperm whales are distributed across the entire North Pacific and into the southern Bering Sea in summer, but the majority are thought to be south of 40ºN in winter (Rice 1974; Rice 1989; Gosho *et al.* 1984; Miyashita *et al.* 1995). The International Whaling Commission (IWC) historically divided the North Pacific into two management regions (Donovan 1991) defined by a zig-zag line which starts at 150ºW at the equator, is 160ºW between 40-50ºN, and ends up at 180ºW north of 50ºN; however, the IWC has not reviewed this stock boundary recently (Donovan 1991). Sperm whales are found year-round in California waters (Dohl *et al.* 1983; Barlow 1995; Forney *et al.* 1995), but they reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974). Sperm whales are seen off Washington and Oregon in every season except winter (Green *et al.* 1992). Of 176 sperm whales that were marked with Discovery tags off southern California in winter between 1962 and 1970, only three were recovered by whalers: one off northern California in June, one off Washington in June, and another far off British Columbia in April (Rice 1974). Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance declines westward towards the middle of the tropical Pacific (near the IWC stock boundary at 150ºW) and declines northward towards the tip of Baja California. Sperm whale population structure in the eastern tropical Pacific is unknown, but the only photographic matches of known individuals from this area have been between the Galapagos Islands and coastal waters of South America (Dufault and Whitehead 1995) and between the Galapagos Islands and the southern Gulf of California (Jaquet *et al.* 2003), suggesting that eastern tropical Pacific animals constitute a distinct stock. No apparent distributional hiatus was found between the U.S. Exclusive Economic Zone (EEZ) off California and Hawaii during a survey designed specifically to investigate stock structure and abundance of sperm whales in the northeastern temperate Pacific (Barlow and Taylor 2005). Sperm whales in the California Current have been identified as demographically independent from animals in Hawaii and the Eastern Tropical Pacific, based on genetic analyses of single-nucleotide polymorphisms (SNPs), microsatellites, and mtDNA (Mesnick *et al.* 2011). For the Marine Mammal Protection Act (MMPA) stock assessment reports, sperm whales within the Pacific U.S. EEZ are divided into three discrete, non-contiguous areas: 1) California, Oregon and Washington waters (this report), 2) waters around Hawaii, and 3) Alaska waters.

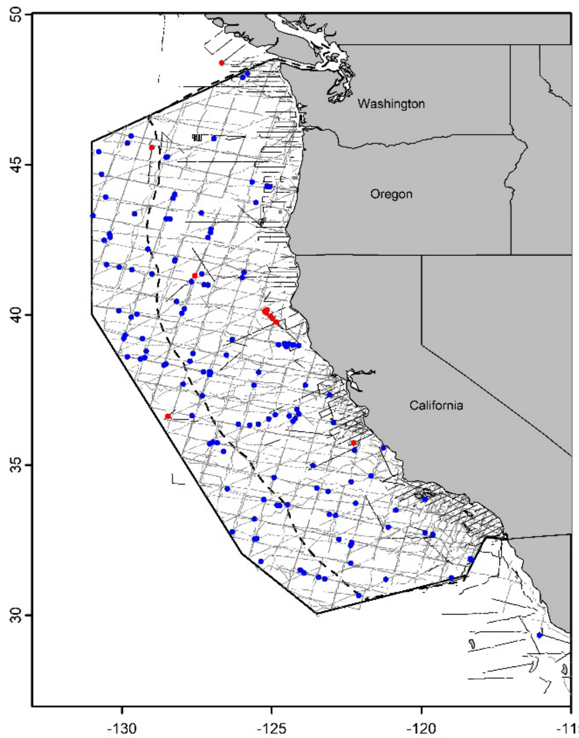


Figure 1. Sperm whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

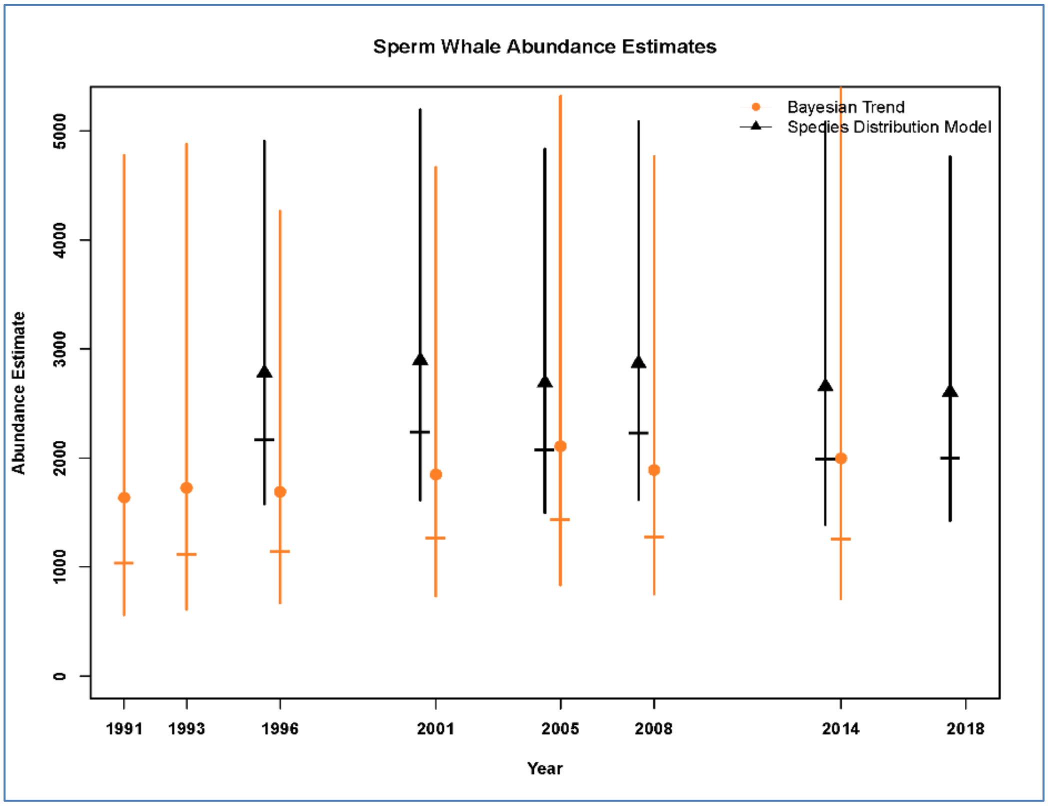


Figure 2. Trend-based and habitat-based abundance estimates of sperm whale abundance in the California Current, 1991-2018 (Moore and Barlow 2017, Becker *et al.* 2020). Abundance estimates (posterior medians [●] and 95% CRIs) from the trend model and mean estimated abundance [▲] and 95% confidence limits from the habitat model are shown. Horizontal hatch marks represent lower 80% percentiles of each estimate, corresponding to the minimum population estimate.

A series of abundance estimates are available from Bayesian trend models derived from line-transect surveys between 1991-2014 (Moore and Barlow 2017) and habitat-based density models from 1991-2018 (Becker *et al.* 2020) (Figure 2). Estimates from the two methods largely-overlap, though estimates from habitat models are, on average, higher. The most-recent estimate of sperm whale abundance for this stock is based on a 2018 survey and a habitat density model that is informed by 1991-2018 data, or 2,606 (CV = 0.135) whales (Becker *et al*. 2020).

### Minimum Population Estimate

The minimum population estimate for sperm whales is taken as the lower 20th percentile of the 2018 abundance estimate, or 2,011 whales (Becker *et al*. 2020).

### Current Population Trend

Moore and Barlow (2014) reported that sperm whale abundance appeared stable from 1991 to 2008 (Figure 2) and additional data from a 2014 survey does not change that conclusion (Moore and Barlow 2017). Estimated growth rates of the population include high uncertainty levels: the growth rate parameter from a Markov model has a posterior median and mean of +0.01 (SD = 0.06) with a broad 95% credible interval (CRI) ranging from -0.11 to +0.13 and a 60% chance of being positive. Another growth rate estimated from a regression model has a posterior mean of +0.01 with 95% CRI ranging from −0.06 to +0.07 (62% chance that growth has been positive), indicating that for the 1991-2014 study period, conclusions about whether the population has increased or decreased are uncertain (Moore and Barlow 2017). Habitat model estimate of abundance from Becker *et al.* (2020) show the same equivocal trend in abundance (Figure 2).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

A reliable estimate of the maximum net productivity rate is not unavailable for the CA/OR/WA stock of sperm whales. Hence, until additional data become available, it is recommended that the cetacean maximum net productivity rate ($R_{max}$) of 4% be employed for this stock at this time (Wade and Angliss 1997).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (2,011) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (for an endangered stock with $N_{min}$ >1,500 and stable trend in abundance; Taylor *et al.* 2003), resulting in a PBR of 4 animals per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The fishery most likely to injure or kill sperm whales from this stock is the California thresher shark/swordfish drift gillnet fishery (Julian and Beeson 1998, Carretta 2022). Observed serious injury and mortality is rarely observed in the fishery (10 animals from 6 events observed during >9,246 fishing sets between 1990 and 2021, Carretta 2022). While there has not been an observed entanglement of sperm whales in this fishery since 2010, there is a positive estimate of sperm whale bycatch in the fishery for the most-recent 5-year period of 2017-2021, based on a data model that uses 1990-2021 data (Carretta 2022). This estimate is 1.58 (CV=2.8) whales, or 0.32 whales annually (Carretta 2022).

Strandings of sperm whales are rare and it is expected that documented anthropogenic deaths and injuries due to entanglements within unknown fisheries or ingestion of marine debris represent a small fraction of the true number of cases, due to the low probability that the carcass of a highly-pelagic species washes ashore (Williams *et al.* 2011, Carretta *et al.* 2016a). Prior cases of observed mortality and serious injury of sperm whales due to interactions with unidentified fisheries and marine debris have been reported by Jacobsen *et al.* (2010) and Carretta *et al.* (2013, 2014). In the most recent 5-year period (2017 to 2021), there was one observation of a seriously-injured sperm whale in unidentified fishing gear (large gauge line) (Carretta *et al*. 2023). There were 3 reports of sperm whales feeding on catch in the limited entry sablefish hook and line fishery, but there was no evidence of entanglement or hooking (Carretta *et al*. 2023). Total mean annual commercial fishery-related serious injury and mortality of sperm whales from 2017-2021 is the sum of mean annual California drift gillnet fishery serious injury and mortality (0.32 whales), plus unidentified fisheries (0.2 whales), or 0.52 whales per year. (Table 1).

**Table 1.** Summary of available information on the incidental mortality and serious injury of sperm whales (CA/OR/WA stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual serious injury and mortality for the California swordfish drift gillnet fishery are based on 2017-2021 data unless stated otherwise (Carretta 2022, Carretta *et al*. 2023, Jannot *et al*. 2022).

| Fishery Name | Year(s) | Data Type | Observer Coverage | Observed mortality and (serious injury) | Estimated mortality and serious injury (CV) | Mean annual mortality and serious injury (CV) |
|---|---|---|---|---|---|---|
| **CA thresher shark/swordfish drift gillnet fishery** | 2017<br>2018<br>2019<br>2020<br>2021 | observer | 0.186<br>0.251<br>0.226<br>0.222<br>0.228 | 0<br>0<br>0<br>0<br>0 | 1.58 (2.8)<br>0 (n/a)<br>0 (n/a)<br>0 (n/a)<br>0 (n/a)<br>0 (n/a) | 0.32 (2.8) |
| **CA/OR/WA limited entry sablefish hook and line** | 2015-2019 | observer | n/a | 0 | 0 | 0 |
| **Unidentified fishery** | 2020 | Sighting | n/a | 0 (1) | 1 (n/a) | 0.2 |
| **Total annual takes** | | | | | | 0.52 (n/a) |

Sperm whales from the North Pacific stock depredate longline sablefish catch in the Gulf of Alaska and sometimes incur serious injuries from becoming entangled in gear (Sigler *et al.* 2008, Allen and Angliss 2011). An unknown number of whales from the CA/OR/WA stock probably venture into waters where Alaska longline fisheries operate, but the amount of temporal and spatial overlap is unknown. Thus, the risk of serious injury to CA/OR/WA stock sperm whales resulting from longline fisheries cannot be quantified.

**Vessel Strikes**

For the most recent 5-year period of 2017-2021, no vessel strike deaths or serious injuries were observed, though one was recorded in 2007 (Carretta *et al.* 2013). Due to the low probability of a sperm whale carcass washing ashore, estimated vessel strike deaths are likely underestimated.

**Other removals**

Whaling removed at least 436,000 sperm whales from the North Pacific between 1800 and the end of legal commercial whaling for this species in 1987 (Best 1976; Ohsumi 1980; Brownell 1998; Kasuya 1998). Of this total, an estimated 33,842 were taken by Soviet and Japanese pelagic whaling operations in the eastern North Pacific from the longitude of Hawaii to the U.S. West coast, between 1961 and 1976 (Allen 1980), and approximately 1,000 were reported taken in land-based U.S. West coast whaling operations between 1919 and 1971 (Ohsumi 1980; Clapham *et al*. 1997). There has been a prohibition ban on taking sperm whales in the North Pacific since 1988, but large-scale pelagic whaling stopped in 1980.

## STATUS OF STOCK

Sperm whales are listed as "endangered" under the U.S. Endangered Species Act (ESA), and consequently this stock is automatically considered as "depleted" and "strategic" under the MMPA. The status of sperm whales with respect to carrying capacity and optimum sustainable population (OSP) is unknown. The observed annual rate of documented mortality and serious injury ($\geq 0.52$ per year) is less than the calculated PBR (4.0) for this stock, but anthropogenic mortality and serious injury is likely underestimated due to incomplete detection of carcasses and injured whales. Total human-caused mortality from commercial fisheries is greater than 10% of the calculated PBR and, therefore, is not insignificant and approaching zero mortality and serious injury rate. Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for deep-diving whales like sperm whales that feed in the ocean's sound channel.

## OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

The NMFS (2015) 5-year review of sperm whale populations noted potential negative impacts to populations that include prey distribution and type changes due to climate change, anthropogenic sound, fishery interactions, oil spills, and pollutants, including heavy metals.

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# GRAY WHALE (*Eschrichtius robustus*): Eastern North Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Once common throughout the Northern Hemisphere, the gray whale was extinct in the Atlantic by the early 1700s (Fraser 1970; Mead and Mitchell 1984), but recent sightings in the Mediterranean Sea in 2010 and off Namibia in 2013 are documented (Scheinin *et al.* 2011, Elwen and Gridley 2013). Gray whales are commonly found in the North Pacific. Genetic studies indicate there are distinct "Eastern North Pacific" (ENP) and "Western North Pacific" (WNP) population stocks, with differentiation in both mtDNA haplotype and microsatellite allele frequencies (LeDuc *et al.* 2002; Lang *et al.* 2014; Weller *et al.* 2013). Brüniche-Olsen *et al*. (2018a) used nuclear single nucleotide polymorphisms (SNPs) from whales sampled off Sakhalin and Mexico breeding lagoons to conclude that genetic differentiation between the two regions was small, but statistically-significant, despite the presence of admixed individuals. These authors conclude that gray whale population structure is not determined by simple geography and may be in flux due to evolving migratory dynamics. Contemporary gray whale genomes, both eastern and western, contain less nucleotide diversity than most other marine mammals and evidence of inbreeding is greater in the Western Pacific than in the Eastern Pacific populations (Brüniche-Olsen *et al*. 2018b).

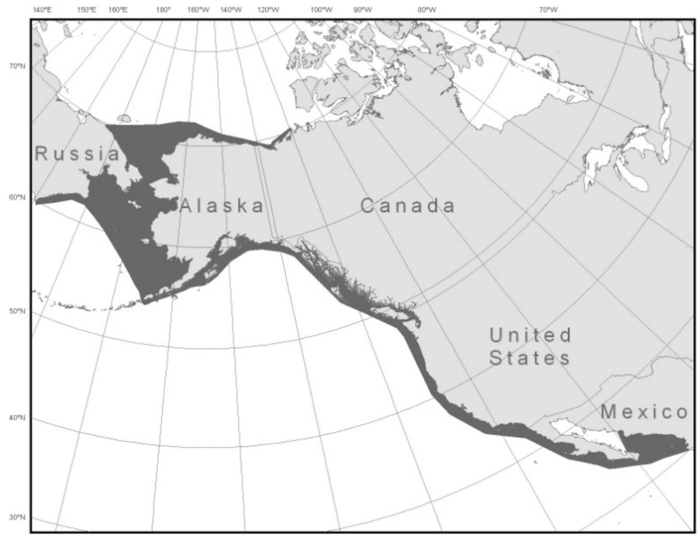


**Figure 1.** Approximate distribution of the Eastern North Pacific stock of gray whales (shaded area).

During summer and fall, most whales in the ENP population feed in the Chukchi, Beaufort and northwestern Bering Seas (Fig. 1). An exception to this is the relatively small number of whales that summer and feed along the Pacific coast between Kodiak Island, Alaska and northern California (Darling 1984, Gosho *et al.* 2011, Calambokidis *et al.* 2017). Three primary wintering lagoons in Baja California, Mexico are utilized, and some females are known to make repeated returns to specific lagoons (Jones 1990). Genetic substructure on the wintering grounds is indicated by significant differences in mtDNA haplotype frequencies between females (mothers with calves) using two primary calving lagoons and females sampled in other areas (Goerlitz *et al.* 2003). Other research has identified a small, but significant departure from panmixia between two lagoons using nuclear data, although no significant differences were identified using mtDNA (Alter *et al.* 2009).

Tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the ENP, including coastal waters of Canada, the U.S. and Mexico (Lang 2010; Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2013, Mate *et al.* 2015, Urbán *et al.* 2019). Photographs of 379 individuals identified on summer feeding grounds off Russia (316 off Sakhalin; 150 off Kamchatka), were compared to 10,685 individuals identified in Mexico breeding lagoons. A total of 43 matches were found, including the following links: 14 Sakhalin-Kamchatka-Mexico, 25 Sakhalin-Mexico, and 4 Kamchatka-Mexico (Urban *et al.* 2019). The number of whales documented moving between the WNP and ENP represents 14% of gray whales identified off Sakhalin Island and Kamchatka according to Urban *et al.* (2019).

In 2010, the IWC Standing Working Group on Aboriginal Whaling Management Procedure noted that different names had been used to refer to gray whales feeding along the Pacific coast, and agreed to designate animals that spend the summer and autumn feeding in coastal waters of the Pacific coast of North America from California to southeast Alaska as the "Pacific Coast Feeding Group" or PCFG (IWC 2012). This definition was further refined for purposes of abundance estimation, limiting the geographic range to the area from northern California to northern British Columbia (from 41°N to 52°N), and limiting the temporal range from June 1 to November 30, and counting only those whales seen in more than one year within this geographic and temporal range (IWC 2012). The IWC adopted this definition in 2011, but noted that "not all whales seen within the PCFG area at this time will be PCFG whales and some PCFG whales will be found outside of the PCFG area at various times during the year." (IWC 2012).

Photo-identification studies between northern California and northern British Columbia provide data on the abundance and population structure of PCFG whales (Calambokidis *et al.* 2017). Gray whales using the study area in summer and autumn include two components: (1) whales that frequently return to the area, display a high degree of intra-seasonal "fidelity" and account for a majority of the sightings between 1 June and 30 November. Despite movement and interchange among sub-regions of the study area, some whales are more likely to return to the same sub-region where they were observed in previous years; (2) "visitors" from the northbound migration that are sighted only in one year, tend to be seen for shorter time periods in that year, and are encountered in more limited areas. Photo-identification (Gosho *et al.* 2011; Calambokidis *et al.* 2017) and satellite tagging (Lagerquist *et al*. 2019; Ford *et al.* 2012) studies have documented some PCFG whales off Kodiak Island, the Gulf of Alaska and Barrow, Alaska, well to the north of the pre-defined 41°N to 52°N boundaries used in PCFG abundance estimation analyses. Lagerquist *et al.* (2019) noted that PCFG whales tagged in autumn in northern California and Oregon waters utilized feeding areas from northern California to Icy Bay, Alaska, with one male remaining in the vicinity of the California/Oregon border for almost a year. The highest use areas for these tagged whales were identified as northern California, central Oregon, and southern Washington waters.

Frasier *et al.* (2011) found significant differences in mtDNA haplotype distributions between PCFG whales and the rest of the ENP stock, in addition to differences in long-term effective population size, and concluded that the PCFG qualifies as a separate management unit under the criteria of Moritz (1994) and Palsbøll *et al.* (2007). The authors noted that PCFG whales probably mate with the rest of the ENP population and that their findings were the result of maternally-directed site fidelity of whales to different feeding grounds.

Lang *et al.* (2014) assessed stock structure from different ENP feeding grounds using mtDNA and eight microsatellite markers. Significant mtDNA differentiation was found when samples from individuals (n=71) sighted over ≥2 years within the seasonal range of the PCFG were compared to samples from whales feeding north of the Aleutians (n=103), and when PCFG samples were compared to samples collected off Chukotka, Russia (n=71). No significant differences were found when the same comparisons were made using microsatellite data. The authors concluded that (1) the significant differences in mtDNA haplotype frequencies between the PCFG and whales sampled in northern areas indicates that use of some feeding areas is influenced by internal recruitment (e.g., matrilineal fidelity), and (2) the lack of significance in nuclear comparisons suggests that individuals from different feeding grounds may interbreed. The level of mtDNA differentiation identified, while statistically significant, was low and the mtDNA haplotype diversity found within the PCFG was similar to that found in the northern strata. Lang *et al.* (2014) suggested this could indicate recent colonization of the PCFG but could also be consistent with external recruitment into the PCFG. An additional comparison of whales sampled off Vancouver Island, British Columbia (representing the PCFG) and whales sampled at the calving lagoon at San Ignacio also found no significant differences in microsatellite allele frequencies, providing further support for interbreeding between the PCFG and the rest of the ENP stock (D'Intino *et al.* 2012). Lang and Martien (2012) investigated potential immigration levels into the PCFG using simulations and produced results consistent with the empirical (mtDNA) analyses of Lang *et al.* (2014). Simulations indicated that immigration of >1 and <10 animals per year into the PCFG was plausible, and that annual immigration of 4 animals/year produced results most consistent with empirical data.

While the PCFG is recognized as a distinct feeding aggregation (Calambokidis *et al.* 2017; Lagerquist *et al.* 2019; Frasier *et al.* 2011; Lang *et al.* 2014; IWC 2012), the status of the PCFG as a population stock is unresolved (Weller *et al.* 2013). A NMFS 2012 gray whale stock identification workshop included a review of photo-identification, genetic, and satellite tag data. The workshop report states "there remains a substantial level of uncertainty in the strength of the lines of evidence supporting demographic independence of the PCFG." (Weller *et al.* 2013). The NMFS task force, charged with evaluating PCFG stock status, noted that "both the photo-identification and genetics data indicate that the levels of internal versus external recruitment are comparable, but these are not quantified well enough to determine if the population dynamics of the PCFG are more a consequence of births and deaths within the group (internal dynamics) rather than related to immigration and/or emigration (external dynamics)." Further, given the lack of significant differences found in nuclear DNA markers between PCFG whales and the rest of the ENP stock, the task force found no evidence to suggest that PCFG whales breed exclusively or primarily with each other, but interbreed with the rest of the ENP stock, including potentially other PCFG whales. Additional research to better identify recruitment levels into the PCFG and further assess the stock status of PCFG whales is needed (Weller *et al.* 2013). In contrast, the task force noted that WNP gray whales should be recognized as a population stock under the MMPA, and NMFS prepared a separate stock assessement report for WNP gray whales in 2014. Because the PCFG appears to be a distinct feeding aggregation and may one day warrant consideration as a distinct stock, separate PBRs are calculated for the PCFG to assess whether levels of human-caused mortality are likely to cause local depletion.

The IWC Scientific Committee completed annual (2014-2018) range-wide workshops on the status of North Pacific gray whales. The primary objectives were to identify plausible stock structure hypotheses and create a foundation for developing range-wide conservation advice.

The Scientific Committee reported on the plausibility of various stock structure hypotheses in 2020 (IWC 2020). These hypotheses include up to three feeding groups or aggregations: the Pacific Coast Feeding Group (PCFG), the Western Feeding Group (WFG), and the North Feeding Group (NFG). The PCFG is defined above. The WFG consists of whales that feed off Sakhalin Island as documented via photo-ID. The NFG includes whales found feeding in the Bering and Chukchi Seas where photo-ID and genetic data are sparse. The IWC Scientific Committee's stock structure hypotheses also consider up to three extant breeding stocks: the Western Breeding Stock (WBS), the Eastern Breeding Stock (EBS), and a third unnamed stock that includes WFG whales that interbreed largely with each other while migrating to the Mexico wintering grounds. The IWC summarizes three 'high plausibility' hypotheses as follows:

Hypothesis 3a is characterized by maternal feeding ground fidelity, one migratory route/wintering region used by Sakhalin whales, and random mating. Under this hypothesis, a single breeding stock (EBS) exists that includes three feeding groups: NFG, PCFG, and WFG. Areas off Southern Kamchatka and the Northern Kuril Islands are used by some whales that belong to the WFG and some whales that belong to the NFG. Although two breeding stocks (WBS and EBS) may once have existed, the WBS is assumed to have been extirpated.

Hypothesis 4a is characterized by maternal feeding ground fidelity, one migratory route/wintering region used by Sakhalin whales, and non-random mating. Under this hypothesis, two breeding stocks exist and overwinter in Mexico. One breeding stock (EBS) includes NFG and PCFG whales, and a second, unnamed breeding stock includes WFG whales that mate largely with each other while migrating to Mexico. Areas off Southern Kamchatka and the Northern Kuril Islands are used by some whales that belong to the breeding stock comprised of WFG whales and some whales that belong to the NFG. Although a third breeding stock (the WBS) may once have existed, under this hypothesis the WBS is assumed to have been extirpated.

Hypothesis 5a is characterized by maternal feeding ground fidelity and two migratory routes/wintering grounds used by Sakhalin whales. Under this hypothesis, two breeding stocks exist: EBS and WBS. The EBS includes three feeding groups: PCFG, NFG, and the WFG that feeds off Northeastern Sakhalin Island. The WBS whales feed off Northeastern Sakhalin Island, Southern Kamchatka, the Northern Kuril Islands and other areas of the Okhotsk Sea and then migrate to the South China Sea to overwinter. Under this hypothesis, areas off Southern Kamchatka and the Northern Kuril Islands are used by the WFG, the NFG, and the feeding whales that are part of the WBS.

**POPULATION SIZE**

Systematic counts of gray whales migrating south along the central California coast have been conducted by shore-based observers at Granite Canyon most years since 1967 (Fig. 2). The most recent estimate of abundance for the ENP population is from the 2015/2016 southbound survey and is 26,960 (CV=0.05) whales (Durban *et al.* 2017) (Fig. 2).

Photographic mark-recapture abundance estimates for PCFG gray whales between 1998 and 2015, including estimates for a number of smaller geographic areas within the IWC-defined PCFG region (41°N to 52°N), are reported in Calambokidis *et al.* (2017). The 2015 abundance estimate for the defined range of the PCFG between 41°N to 52°N is 243 whales (SE=18.9; CV= 0.08).

Eastern North Pacific gray whales experienced an unusual mortality event (UME) beginning in 2019 (which is ongoing), when large numbers of whales stranded from Mexico to Alaska (NOAA 2020a). Necropsies conducted on a subset of stranded whales indicated that many animals showed evidence of nutritional stress. NOAA is coordinating an independent team of scientists to review the stranding data and samples as part of the Working Group on Marine Mammal Unusual Mortality Events. NOAA continues to monitor the gray whale population through abundance and calf production surveys. The current UME is similar to that of 1999 and 2000, when large numbers of animals also stranded along the west coast of North America (Moore et al., 2001; Gulland et al., 2005). Stranding numbers during the 1999-2000 UME exceeded that of the current UME to date, although estimated population size at the time of the 1999-2000 UME was between 15,000 to 18,000 animals. During the 1999-2000 UME, >60% of the dead whales were adults, compared with previous years when calf strandings were more common. Several factors following the 1999-2000 UME suggest that the high mortality rate observed was a short-term, acute event: 1) in 2001 and 2002, strandings decreased to levels below UME levels (Gulland et al., 2005); 2) average calf production returned to levels seen before 1999; and 3) in 2001, living whales no longer appeared emaciated. Oceanographic factors that limited food availability for gray whales were identified as likely causes of the UME (LeBouef et al. 2000; Moore et al. 2001; Minobe 2002; Gulland et al. 2005), with resulting declines in survival rates of adults during this period (Punt and Wade 2012). Investigations on the causes of the current UME may yield similar conclusions. The ENP gray whale population has recovered to levels seen prior to the UME of 1999-2000 and the current estimate of abundance is the highest in the 1967-2015 time series (Fig. 2).

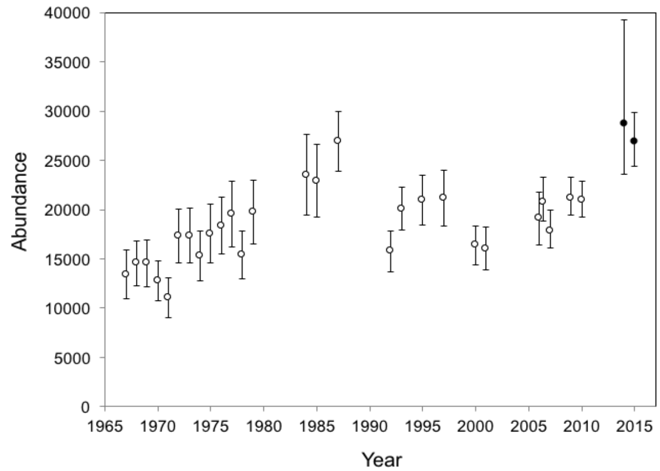


**Figure 2.** Estimated abundance of Eastern North Pacific gray whales from NMFS counts of migrating whales past Granite Canyon, California. Open circles represent abundance estimates and 95% confidence intervals reported by Laake *et al.* (2012) and Durban *et al.* (2015). Closed circles represent estimates and 95% posterior highest density intervals reported by Durban *et al.* (2017) for the 2014/2015 and 2015/2016 migration seasons.

**Minimum Population Estimate**

The minimum population estimate ($N_{MIN}$) for the ENP stock is calculated from Equation 1 from the PBR Guidelines (Wade and Angliss 1997): $N_{MIN} = N/\exp(0.842\times[\ln(1 + [CV(N)]^2)]^{½})$. Using the 2015/2016 abundance estimate of 26,960 and its associated CV of 0.05 (Durban *et al.* 2017), $N_{MIN}$ for this stock is 25,849.

The minimum population estimate for PCFG gray whales is calculated as the lower 20[th] percentile of the log-normal distribution of the 2015 mark-recapture estimate of 243 (CV=0.08), or 227 animals.

**Current Population Trend**

The population size of the ENP gray whale stock has increased over several decades despite an UME in 1999 and 2000 (see Fig. 2). Durban *et al.* (2017) noted that a recent 22% increase in ENP gray whale abundance over 2010/2011 levels is consistent with high observed and estimated calf production (Perryman *et al.* 2017). Recent increases in abundance also support hypotheses that gray whales may experience more favorable feeding conditions in arctic waters due to an increase in ice-free habitat that might result in increased primary productivity in the region (Perryman *et al.* 2002, Moore 2016). Abundance estimates of PCFG whales increased from 1998 through 2004, remained stable for the period 2005-2010, and have steadily increased during the 2011-2015 time period (Calambokidis *et al.* 2017).

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

Using abundance data through 2006/07, an analysis of the ENP gray whale population led to an estimate of Rmax of 0.062, with a 90% probability the value was between 0.032 and 0.088 (Punt and Wade 2012). This value of Rmax is also applied to PCFG gray whales, as it is currently the best estimate of Rmax available for gray whales in the ENP.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for the ENP stock of gray whales is calculated as the minimum population size (25,849), times one-half of the maximum theoretical net population growth rate (½ x 6.2% = 3.1%), times a recovery factor of 1.0 for a stock above MNPL (Punt and Wade 2012), or 801 animals per year.

The potential biological removal (PBR) level for PCFG gray whales is calculated as the minimum population size (227 animals), times one half the maximum theoretical net population growth rate (½ x 6.2% = 3.1%), times a recovery factor of 0.5 (for a population of unknown status), resulting in a PBR of 3.5 animals per year. Use of the recovery factor of 0.5 for PCFG gray whales, rather than 1.0 used for ENP gray whales, is based on uncertainty regarding stock structure and guidelines for preparing marine mammal stock assessments which state that "Recovery factors of 1.0 for stocks of unknown status should be reserved for cases where there is assurance that $N_{min}$, $R_{max}$, and the kill are unbiased and where the stock structure is unequivocal" (NMFS 2005, Weller *et al.* 2013). Given uncertainties in external versus internal recruitment levels of PCFG whales, the equivocal nature of the stock structure, and the small estimated population size of the PCFG, NMFS will continue to use the default recovery factor of 0.5 for PCFG gray whales.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

A total of 62 gray whale records involving human-caused deaths or serious injuries were assessed for the 5-year period 2014-2018 (Carretta *et al.* 2020). These included commercial fishery-related cases (n=50), vessel strikes (n=9), marine debris entanglements (n=2), and illegal hunts (n=1). These records are summarized in the report sections below.

**Fisheries Information**

The California large-mesh drift gillnet fishery for swordfish and thresher shark includes five observed entanglement records of gray whales from 9,085 observed fishing sets from1990-2018 (Carretta 2020). The estimated bycatch of gray whales in this fishery for the most recent 5-year period is 2.6 (CV=0.37) whales, or 0.52 whales annually (Carretta 2020). By comparison, the more coastal set gillnet fishery for halibut and white seabass has no observations of gray whale entanglements from over 10,000 observed sets for the same time period. This compares with 13 opportunistically documented unidentified gillnet fishery entanglements of gray whales in U.S. west coast waters during the most recent 5 year period of 2014-2018, including one self-report from a set gillnet vessel operator (Carretta 2020, Carretta *et al*. 2020). Alaska gillnet fisheries also interact with gray whales, but these fisheries largely lack observer programs. Some gillnet entanglements involving gray whales along the coasts of Washington, Oregon, and California may involve gear set in Alaska and/or Mexico waters and carried south and/or north during annual migration.

Entanglement in commercial pot and trap fisheries is another source of gray whale mortality and serious injury (Carretta *et al.* 2018, 2020). Most data on human-caused mortality and serious injury of gray whales are from strandings, including at-sea reports of entangled animals alive or dead (Carretta *et al.* 2018, 2020). Strandings represent only a fraction of actual gray whale deaths (natural or human-caused), as reported by Punt and Wade (2012), who estimated that only 3.9% to 13.0% of gray whales that die in a given year end up stranding and being reported. This estimate of carcass detection, however, also included sparsely-populated coastlines of Baja California, Canada, and Alaska, for which the rate of carcass detection is expected to be low. Since most U.S. cases of human-caused serious injury and mortality are documented from Washington, Oregon, and California waters, the Punt and Wade (2012) estimate of carcass recovery is not applicable to U.S. West Coast waters. An appropriate correction factor for undetected anthropogenic mortality and serious injury of gray whales is unavailable.

A summary of human-caused mortality and serious injury from fishery sources is given in Table 1 for the most recent 5-year period of 2014 to 2018 (Carretta *et al.* 2018b, 2020). Total observed and estimated entanglement-related human-caused mortality and serious injury for ENP gray whales is 9.3 whales annually, which includes PCFG entanglements (Table 1). The mean annual entanglement-related serious injury and mortality level for PCFG gray whales is 1.1 whales (Table 1). Gray whale serious injuries in unidentified fishing gear during 2014-2018 totaled 20.25, or 4 whales annually (Table 1, Carretta *et al*. 2020). Additionally, there were 21 *unidentfied whale* entanglements during 2014-2018, of which, 4.1 were prorated as gray whales using the method reported by Carretta (2018). Of these 4.1 entanglements, 1.2 occurred within the geographic and seasonal limit range considered to represent PCFG gray whales. Unidentified whale entanglements typically involve whales seen at-sea with unknown gear configurations that are prorated to represent 0.75 serious injuries per entanglement case. Thus it is estimated that at least 3.9 x 0.75 = 2.9 additional ENP gray whale and 1.2 x 0.75 = 0.9 PCFG serious injuries occurred from the 21 unidentified whale entanglement cases during 2014-2018. This represents 0.6 ENP gray whales and 0.2 PCFG gray whales annually. The 5-year total of 2.9 prorated ENP gray whale serious injuries from 2014 to 2018 includes 1.2 prorated PCFG serious injuries as PCFG whales are included in abundance and PBR calculations for the larger ENP stock. Total ENP gray whale serious injury and mortality from Table 1 totals 9.3 whales annually, and 1.1 annually for PCFG gray whales.

Table 1. Entanglement mortality and serious injury of gray whales, 2014-2018 (Carretta 2020, Carretta *et al.* 2020). Entanglement in most fisheries is derived from strandings and at-sea sightings of entangled whales and thus represent minimum impacts because they are documented opportunistically (Carretta *et al.* 2020). Mortality and injury information, where possible, is assigned to either the ENP gray whale stock or PCFG whales. Total mortality and

injury of ENP gray whales includes records attributable to PCFG gray whales, because abundance estimates and calculated PBR for ENP gray whales includes PCFG whales.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed mortality (+ serious injury) | Estimated mortality (CV) | Mean annual takes 2014-2018 (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet** | 2014-2018 | observer | 21% | ENP 1 (0) | 2.6 (0.37) | 0.52 (0.37) (ENP stock) |
| **CA halibut and white seabass set gillnet** | 2014-2018 | vessel self-report in 2015 | n/a | ENP 0 (0.75) | n/a | ENP 0.15 (n/a) |
| **CA Dungeness crab pot** | | strandings + sightings | n/a | ENP 1 (0.75) | n/a | ENP 0.35 (n/a) |
| **WA Dungeness crab pot** | | | | ENP 3 (1.75)<br>PCFG 0 (0.75) | | ENP 0.95 (n/a)<br>PCFG 0.15 (n/a) |
| **Unidentified pot/trap fishery** | | | | ENP 1 (3.75)<br>PCFG 0 (1.5) | | ENP 0.95 (n/a)PCFG 0.3 (n/a) |
| **Unidentified gillnet fishery** | | | | ENP 1 (9.5) | | ENP 2.1 (n/a) |
| **Unidentified fishery interactions involving *gray whales*** | | | | ENP 3 (15.25)<br>PCFG 1 (1) | | ENP 3.65 (n/a)<br>PCFG 0.4 (n/a) |
| **Unidentified fishery interactions involving *unidentified whales* prorated to gray whale** | | | | ENP 0 (2.9)<br>PCFG 0 (1.2) | | ENP 0.6 (n/a)<br>PCFG 0.2 (n/a) |
| **Totals** | | | | ENP 10 (34.65)<br>PCFG 1 (4.45) | | ENP 9.3 (n/a)<br>PCFG 1.1 (n/a) |

**Subsistence/Native Harvest Information**

Subsistence hunters in Russia, Canada, and the United States have traditionally harvested whales from the ENP stock, although only the Russian hunt has persisted in recent years (Huelsbeck 1988; Monks *et al*. 2001, Reeves 2002). NMFS has proposed to grant a waiver of the Marine Mammal Protection Act's moratorium on the take of marine mammals to allow the Makah Indian Tribe to take a limited number of Eastern North Pacific gray whales (NOAA 2019, 2020a, 2020b). The proposed rule includes a potential maximum removal of an average of 2.5 whales annually over a 10-year period (NOAA 2019). Proposed regulations include considerations of the estimated probabilities of a Makah hunt taking a WNP gray whale (Moore and Weller 2018) and safeguards to minimize the probability of taking either WNP or PCFG whales. The proposed rule states: "*there is about a 6 percent probability of hunters striking one WNP gray whale over the 10 years of the regulations (Moore and Weller, 2018). This probability is the most likely point estimate; the 95 percent confidence interval ranges from 3.0 percent to 9.3 percent. Stated another way, the most likely point estimates indicate that one in 17 10-year hunt periods (i.e., one year out of 170) would result in an individual WNP gray whale being struck by Makah hunters, if the Tribe made the maximum number of strike attempts allowed in even-year hunts and if ENP and WNP population sizes and migration patterns remained constant (Moore and Weller, 2018)*". A formal hearing occurred in November 2019 and NMFS is awaiting a recommended decision from the Administrative Law Judge overseeing that hearing (NOAA 2020b). The IWC Scientific Committee reviewed the proposed U.S. management plan for the Makah hunt of gray whales and stated that "*the performance of the Management Plan was adequate to meet the Commission's conservation objectives for the Pacific Coast Feeding Group, Western Feeding Group and Northern Feeding Group gray whales*" (IWC 2018).

In 2018, the IWC approved a 7-year Aboriginal Subsistence Whaling catch limit (2019-2025) of 980 gray whales landed, with an annual cap of 140 strikes (subject to a carry forward provision), for Russian and U.S. (Makah Indian Tribe) Native hunters based on the joint request and needs statements submitted by the U.S. and the Russian Federation. The U.S. and the Russian Federation have agreed that the quota will be shared with an average annual harvest of 135 whales by the Russian Chukotka people and 5 whales by the Makah Indian Tribe. Total takes by the Russian hunt during the past five years were: 124 in 2014, 125 in 2015, 120 in 2016, 119 in 2017 and 107 in 2018 (International Whaling Commission). There were no whales taken by the Makah Indian Tribe during that period because their hunt request is still under review. Based on this information, the annual subsistence take averaged 119 whales during the 5-year period from 2014 to 2018. The IWC reports a total of 4,013 gray whales harvested from annual aboriginal subsistence hunts for the 34-year period 1985 to 2018, which includes struck and lost whales. The estimated population size of ENP gray whales has increased during this same period (Fig. 2).

**Other Mortality**

Vessel strikes are a source of mortality and serious injury for gray whales. During the most recent five-year period, 2014-2018, serious injury and mortality of ENP gray whales attributed to vessel strikes totaled 9 animals ( 7 deaths and 2 serious injuries) or 1.8 whales annually (Carretta *et al.* 2020). Total vessel strike serious injury and mortality of gray whales observed in the PCFG range and season was 3 animals, or 0.6 whales per year (Carretta *et al.* 2020). Vessel strikes attributed to PCFG whales are also included in ENP totals. Additional mortality from vessel strikes probably goes unreported because the whales either do not strand, are undetected, or lack obvious signs of trauma.

Marine debris entanglements account for a small observed percentage of gray whale serious injuries and deaths. During 2014-2018, there were a total of 2 serious injuries/deaths, or 0.4 serious injuries/deaths annually attributed to marine debris entanglement for the ENP stock of gray whales (Carretta *et al*. 2020).

One gray whale was illegally killed in 2017 by Alaska native hunters. NOAA closed the investigation on this incident in 2018. The 5-year annual average for illegal hunts is 0.2 whales.

**HABITAT CONCERNS**

Nearshore industrialization and shipping congestion throughout gray whale migratory corridors represent risks due to increased likelihood of exposure to pollutants and vessel strikes, as well as a general habitat degradation.

The Arctic climate is changing significantly, resulting in reductions in sea ice cover that are likely to affect gray whale populations (Perryman *et al.* 2002, Johannessen *et al.* 2004, Comiso *et al.* 2008, Gailey *et al.* 2020). For example, the summer range of gray whales has greatly expanded (Rugh *et al.* 2001). Bluhm and Gradinger (2008) examined the availability of pelagic and benthic prey in the Arctic and concluded that pelagic prey is likely to increase while benthic prey is likely to decrease in response to climate change. They noted that marine mammal species that exhibit trophic plasticity (such as gray whales, which feed on both benthic and pelagic prey) will adapt better than trophic specialists. Annual sea ice conditions in arctic foraging grounds have been linked to variability in gray whale calf survival and production in both Western (Gailey *et al*. 2020), and Eastern (Perryman *et al*. 2002) North Pacific populations. Following years of high sea-ice coverage on foraging grounds, calf survival and production decline. Decreased spatial and temporal access to foraging grounds as a result of heavy ice cover is hypothesized as the responsible factor.

Global climate change is likely to increase human activity in the Arctic as sea ice decreases, including oil and gas exploration and shipping (Hovelsrud *et al.* 2008). Such activity will increase the risk of oil spills and vessel strikes in this region. Gray whales demonstrate avoidance behavior to anthropogenic sounds associated with oil and gas exploration (Malme *et al.* 1983, 1984) and low-frequency active sonar during acoustic playback experiments (Buck and Tyack 2000, Tyack 2009). Ocean acidification could reduce the abundance of shell-forming organisms (Fabry *et al.* 2008, Hall-Spencer *et al.* 2008), many of which are important in the gray whales' diet (Nerini 1984).

**STATUS OF STOCK**

In 1994, the ENP stock of gray whales was removed from the List of Endangered and Threatened Wildlife (the List), as it was no longer considered endangered or threatened under the Endangered Species Act (NMFS 1994). Punt and Wade (2012) estimated the 2009 ENP population was at 85% of carrying capacity (K) and at 129% of the maximum net productivity level (MNPL), with a probability of 0.884 that the population was above MNPL and therefore within the range of its optimum sustainable population (OSP).

The ongoing (UME) that began in 2019 resulted in elevated levels of stranded gray whales in poor body condition. NOAA continues to monitor this population through calf production surveys during the northbound migration and abundance surveys during the southbound migration (see Population Size). Even though the stock is within OSP, abundance will fluctuate as the population adjusts to natural and human-caused factors affecting carrying capacity (Punt and Wade 2012). It is expected that a population close to or at carrying capacity will be more susceptible to environmental fluctuations (Moore *et al.* 2001). The correlation between gray whale calf production and environmental conditions in the Bering Sea may reflect this (Perryman *et al.* 2002; Perryman and Weller 2012). Overall, the population nearly doubled in size over the first 20 years of monitoring, and has fluctuated for the last 30 years, with a recent increase to over 26,000 whales. Carrying capacity for this stock was estimated at 25,808 whales in 2009 (Punt and Wade 2012), however the authors noted that carrying capacity was likely to vary with environmental conditions.

Based on 2014-2018 data, the estimated annual level of human-caused mortality and serious injury for ENP gray whales includes Russian harvest (119), mortality and serious injury from commercial fisheries (9.3), marine debris (0.4), vessel strikes (1.8), and illegal hunts (0.2) totals 131 whales per year, which does not exceed the PBR (801). Therefore, the ENP stock of gray whales is not classified as a strategic stock.

PCFG gray whales do not currently have a formal status under the MMPA. Abundance estimates of PCFG whales increased from 1998 through 2004, remained stable during 2005-2010, and have steadily increased from 2011-2015 (Calambokidis *et al.* 2017). Total annual human-caused mortality of PCFG gray whales from 2014 to 2018 includes mortality and serious injuries due to commercial fisheries (1.1/yr ), and vessel strikes (0.6/yr), or 1.7 whales annually. This does not exceed the calculated PBR level of 3.5 whales for this population. However, observed levels of human-caused mortality and serious injury from commercial fisheries and vessel strikes for both ENP and PCFG whales represent minimum estimates because not all cases are detected.

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# GRAY WHALE (*Eschrichtius robustus*): Western North Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Gray whales occur along the eastern and western margins of the North Pacific. In the western North Pacific (WNP), gray whales feed during summer and fall in the Okhotsk Sea off northeast Sakhalin Island, Russia, and off southeastern Kamchatka in the Bering Sea (Weller *et al.* 1999, 2002; Vertyankin *et al.* 2004; Tyurneva *et al.* 2010; Burdin *et al.* 2017; Figure 1). Historical evidence indicates that the coastal waters of eastern Russia, the Korean Peninsula and Japan were once part of the migratory route in the WNP and that areas in the South China Sea may have been used as wintering grounds (Weller *et al.* 2002; Weller *et al.* 2013a). Present day records of gray whales off Japan (Nambu *et al.* 2010; Nakamura *et al.* 2017a; Nakamura *et al.* 2017b) and China are infrequent (Wang 1984; Zhu 2002; Wang *et al.* 2015) and the last known record from Korea was in 1977 (Park 1995; Kim *et al.* 2013). While recent observations of gray whales off the coast of Asia remain sporadic, observations off Japan, mostly from the Pacific coast, appear to be increasing in the past two decades (Nakamura *et al.* 2017b).

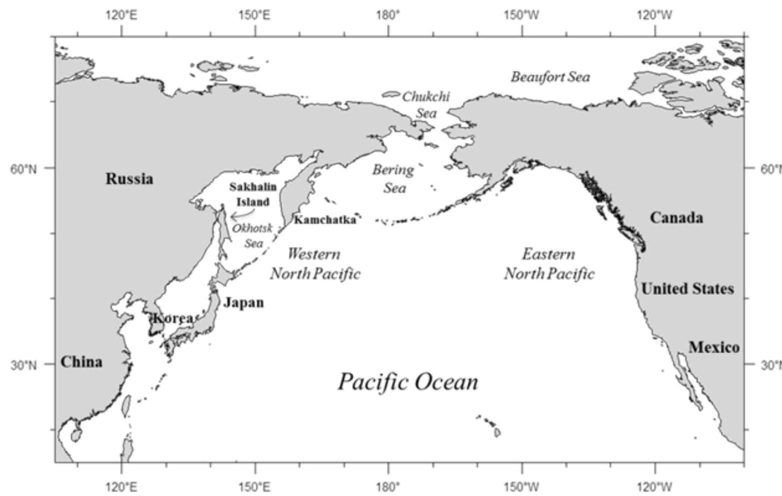


**Figure 1.** Range map of the Western North Pacific Stock of gray whales, including summering areas off Russia and wintering areas in the western and eastern Pacific.

Information from tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the eastern North Pacific (ENP), including coastal waters of Canada, the U.S. and Mexico (Lang *et al*. 2014; Weller *et al.* 2012; Mate *et al.* 2015, Urbán *et al.* 2019). Photographs of 379 individuals identified on summer feeding grounds off Russia (316 off Sakhalin; 150 off Kamchatka), were compared to 10,685 individuals identified in Mexico breeding lagoons. A total of 43 matches were found, including the following area matches: 14 Sakhalin-Kamchatka-Mexico, 25 Sakhalin-Mexico, and 4 Kamchatka-Mexico (Urban *et al.* 2019). The number of whales documented moving between the WNP and ENP represents 14% of gray whales identified off Sakhalin Island and Kamchatka according to Urban *et al.* (2019). Some whales that feed off Sakhalin Island in summer migrate east across the Pacific to the west coast of North America in winter, while others migrate south to waters off Japan and China (Weller *et al.* 2016). Cooke *et al.* (2019) note that the fraction of the WNP population that migrates to the ENP is estimated at 45-80% and note "therefore it is likely that a western breeding population that migrates through Asian waters still exists." These authors further state that at least 20% of WNP gray whales probably migrate elsewhere, likely to wintering areas in Asian waters. Despite these estimates of cross-basin movements, analysis of photo-identification data, including data on mother-calf pairs and paternity assessments, suggest that gray whales summering in the WNP may constitute a demographically self-contained subpopulation where mating occurs at least preferentially and possibly exclusively within the subpopulation (Broker *et al.* 2020, Cooke *et al.* 2017). Despite the observed movements of some gray whales between the WNP and ENP, significant differences in their mitochondrial and nuclear DNA exist (LeDuc *et al.* 2002; Lang *et al.* 2014). Taken together, these observations indicate that not all gray whales in the WNP share a common wintering ground. Brüniche-Olsen *et al.* (2018a) reassessed the genetic differentiation of gray whales feeding off Sakhalin and ENP whales from the Mexican breeding lagoons using nuclear Single Nucleotide Polymorphisms (SNPs). The degree of differentiation between these two regions was small but significant despite the existence of some admixed individuals. In conclusion, these authors suggested that gray whale population structure is not currently determined by simple geography and may be in flux as a result of emerging migratory dynamics. Contemporary gray whale genomes, both eastern and western, contain less nucleotide diversity than most other marine mammals and evidence of inbreeding is greater in the Western Pacific than in the Eastern Pacific populations (Brüniche-Olsen et al. 2018b).

In 2012, the National Marine Fisheries Service convened a scientific task force to appraise the currently recognized and emerging stock structure of gray whales in the North Pacific (Weller *et al.* 2013b). The charge of the task force was to evaluate gray whale stock structure as defined under the Marine Mammal Protection Act (MMPA) and implemented through the National Marine Fisheries Service's Guidelines for Assessing Marine Mammal Stocks (GAMMS; NMFS 2005). Significant differences in both mitochondrial and nuclear DNA between whales sampled off Sakhalin Island (WNP) and whales sampled in the ENP provided convincing evidence that resulted in the task force advising that WNP gray whales should be recognized as a population stock under the MMPA and GAMMS guidelines. Given the interchange of some whales between the WNP and ENP, including seasonal occurrence of WNP whales in U.S. waters, the task force agreed that a stand-alone WNP gray whale population stock assessment report was warranted.

The IWC Scientific Committee completed annual (2014-2018) range-wide workshops on the status of North Pacific gray whales. The primary objectives of these meetings were to identify plausible stock structure hypotheses and create a foundation for developing range-wide conservation advice.

The Scientific Committee reported on the plausibility of various stock structure hypotheses in 2020 (IWC 2020). These hypotheses include up to three feeding groups or aggregations: the Pacific Coast Feeding Group (PCFG), the Western Feeding Group (WFG), and the North Feeding Group (NFG). The PCFG is defined above. The WFG consists of whales that feed off Sakhalin Island as documented via photo-ID. The NFG includes whales found feeding in the Bering and Chukchi Seas where photo-ID and genetic data are sparse. The IWC Scientific Committee's stock structure hypotheses also consider up to three extant breeding stocks: the Western Breeding Stock (WBS), the Eastern Breeding Stock (EBS), and a third unnamed stock that includes WFG whales that interbreed largely with each other while migrating to the Mexico wintering grounds. The IWC summarizes three 'high plausibility' hypotheses as follows:

**Hypothesis 3a** is characterized by maternal feeding ground fidelity, one migratory route/wintering region used by Sakhalin whales, and random mating. Under this hypothesis, a single breeding stock (EBS) exists that includes three feeding groups: NFG, PCFG, and WFG. Areas off Southern Kamchatka and the Northern Kuril Islands are used by some whales that belong to the WFG and some whales that belong to the NFG. Although two breeding stocks (WBS and EBS) may once have existed, the WBS is assumed to have been extirpated.

**Hypothesis 4a** is characterized by maternal feeding ground fidelity, one migratory route/wintering region used by Sakhalin whales, and non-random mating. Under this hypothesis, two breeding stocks exist and overwinter in Mexico. One breeding stock (EBS) includes NFG and PCFG whales, and a second, unnamed breeding stock includes WFG whales that mate largely with each other while migrating to Mexico. Areas off Southern Kamchatka and the Northern Kuril Islands are used by some whales that belong to the breeding stock comprised of WFG whales and some whales that belong to the NFG. Although a third breeding stock (the WBS) may once have existed, under this hypothesis the WBS is assumed to have been extirpated.

**Hypothesis 5a** is characterized by maternal feeding ground fidelity and two migratory routes/wintering grounds used by Sakhalin whales. Under this hypothesis, two breeding stocks exist: EBS and WBS. The EBS includes three feeding groups: PCFG, NFG, and the WFG that feeds off Northeastern Sakhalin Island. The WBS whales feed off Northeastern Sakhalin Island, Southern Kamchatka, the Northern Kuril Islands and other areas of the Okhotsk Sea and then migrate to the South China Sea to overwinter. Under this hypothesis, areas off Southern Kamchatka and the Northern Kuril Islands are used by the WFG, the NFG, and the feeding whales that are part of the WBS.

## POPULATION SIZE

Estimated population size from photo-ID data for Sakhalin and Kamchatka in 2016 was estimated at 290 whales (90% percentile intervals = 271 – 311) (Cooke 2017, Cooke *et al.* 2018). Of these, 175-192 whales are estimated to be predominantly part of a Sakhalin feeding aggregation. These estimates represent animals in the 1-year plus age category. Cooke (2017) notes that not all of these animals belong to the Western North Pacific stock of gray whales and proposes an upper limit of approximately 100 whales from Sakhalin that could belong to the Western North Pacific breeding population.

### Minimum Population Estimate

The minimum population size estimate is taken as the lower 5$^{th}$ percentile of the estimate from Cooke (2017), or 271 animals. This is a more conservative estimate of minimum population size than using the lower 20$^{th}$ percentile of a population estimate, however, Cooke (2017) did not provide such an estimate in his analysis.

**Current Population Trend**

The combined Sakhalin Island and Kamchatka populations were estimated to be increasing from 2005 through 2016 at an average rate between 2-5% annually (Cooke 2017).

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

An analysis of the ENP gray whale population provided an estimate of $R_{max}$ of 0.062, with a 90% probability the value was between 0.032 and 0.088 (Punt and Wade 2012). This value of $R_{max}$ is also applied to WNP gray whales, as it is currently the best estimate of $R_{max}$ available for any gray whale population.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (271) times one-half the estimated maximum annual growth rate for a gray whale population (½ of 6.2% for the ENP stock, Punt and Wade 2012), times a recovery factor of 0.1 (for an endangered stock with $N_{min} < 1{,}500$, Taylor *et al.* 2003), and also multiplied by estimates for the proportion of the stock that uses U.S. EEZ waters (0.575), and the proportion of the year that those animals are in the U.S. EEZ (3 months, or 0.25 years) (Moore and Weller 2013), resulting in a PBR of 0.12 WNP gray whales per year, or approximately 1 whale every 8 years (if abundance and other parameters in the PBR equation remained constant over that time period).

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Fisheries Information**

The decline of gray whales in the WNP is attributable to commercial hunting off Korea and Japan between the 1890s and 1960s. Pre-exploitation abundance of WNP gray whales is unknown, but has been estimated to be between 1,500 and 10,000 individuals (Yablokov and Bogoslovskaya 1984). By 1910, after some commercial exploitation had already occurred, it is estimated that only 1,000 to 1,500 gray whales remained in the WNP population (Berzin and Vladimirov 1981). The basis for how these two estimates were derived, however, is not apparent (Weller *et al.* 2002). By the 1930s, gray whales in the WNP were considered by many to be extinct (Mizue 1951; Bowen 1974).

A significant threat to gray whales in the WNP are incidental catches in coastal net fisheries (Weller *et al.* 2002; Nakamura *et al.* 2017b; Weller *et al.* 2008; Weller *et al.* 2013a; Lowry *et al.* 2018). Between 2005 and 2007, four female gray whales (including one mother-calf pair and one yearling) died in fishing nets on the Pacific coast of Japan. In addition, one adult female gray whale died as a result of a fisheries interaction in November 2011 off Pingtan County, China (Wang *et al.* 2015). An analysis of anthropogenic scarring of gray whales photographed off Sakhalin Island found that at least 18.7% (n=28) of 150 individuals identified between 1994 and 2005 had evidence of previous entanglements in fishing gear but where the scars were acquired is unknown (Bradford *et al.* 2009). Trap nets for Pacific salmon have been deployed in the feeding area off northeastern Sakhalin Island since 2013, resulting in two known entanglements and one probable entanglement mortality (Lowry *et al.* 2018).

Given that some WNP gray whales occur in U.S. waters, there is some probability of WNP gray whales being killed or injured by vessel strikes or entangled in fishing gear within U.S. waters.

**Subsistence/Native Harvest Information**

NMFS has proposed to grant a waiver of the Marine Mammal Protection Act's moratorium on the take of marine mammals to allow the Makah Indian Tribe to take a limited number of Eastern North Pacific gray whales (NOAA 2019, 2020b). The **proposed rule** includes a potential maximum removal of an average of 2.5 whales annually over a 10-year period (NOAA 2019). Proposed regulations include considerations of the estimated probabilities of a Makah hunt taking a WNP gray whale (Moore and Weller 2018) and safeguards to minimize the probability of taking either WNP or PCFG whales. The proposed rule states "*there is about a 6 percent probability of hunters striking one WNP gray whale over the 10 years of the regulations (Moore and Weller, 2018). This probability is the most likely point estimate; the 95 percent confidence interval ranges from 3.0 percent to 9.3 percent. Stated another way, the most likely point estimates indicate that one in 17 10-year hunt periods (i.e., one year out of 170) would result in an individual WNP gray whale being struck by Makah hunters, if the Tribe made the maximum number of strike attempts allowed in even-year hunts and if ENP and WNP population sizes and migration patterns remained constant (Moore and Weller, 2018)*". A formal hearing occurred in November 2019 and NMFS is awaiting a recommended decision from the Administrative Law Judge overseeing that hearing (NOAA 2020b). The IWC Scientific Committee reviewed the proposed U.S. management plan for the Makah hunt of gray whales and stated that "*the performance of the Management Plan was adequate to meet the Commission's conservation objectives for the Pacific Coast Feeding*

*Group, Western Feeding Group and Northern Feeding Group gray whales*" (IWC 2018).

## HABITAT ISSUES

Near shore industrialization and shipping congestion throughout the migratory corridors of the WNP gray whale stock represent risks by increasing the likelihood of exposure to pollutants and vessel strikes as well as a general degradation of the habitat. In addition, the summer feeding area off Sakhalin Island is a region rich with offshore oil and gas reserves. Two major offshore oil and gas projects now directly overlap or are in near proximity to this important feeding area, and more development is planned in other parts of the Okhotsk Sea that include the migratory routes of these whales. Operations of this nature have introduced new sources of underwater noise, including seismic surveys, increased shipping traffic, habitat modification, and risks associated with oil spills (Weller *et al.* 2002). During the past decade, a Western Gray Whale Advisory Panel, convened by the International Union for Conservation of Nature (IUCN), has been providing scientific advice on the matter of anthropogenic threats to gray whales in the WNP. Ocean acidification could reduce the abundance of shell-forming organisms (Fabry *et al.* 2008, Hall-Spencer *et al.* 2008), many of which are important in the gray whales' diet (Nerini 1984). An **unusual mortality event (UME)** that began in 2019 and which is ongoing (NOAA 2020a), resulted in elevated levels of stranded gray whales in poor body condition, however, it is unknown if oceanographic conditions related to this UME affected WNP and ENP gray whales similarly. Annual sea ice conditions in arctic foraging grounds have been linked to variability in gray whale calf survival and production in both Western (Gailey *et al.* 2020), and Eastern (Perryman *et al.* 2002) North Pacific populations. Following years of high sea-ice coverage on foraging grounds, calf survival and production decline. Decreased spatial and temporal access to foraging grounds as a result of heavy ice cover is hypothesized as the responsible factor.

## STATUS OF STOCK

The WNP stock is listed as "Endangered" under the U.S. Endangered Species Act of 1973 (ESA) and is therefore also considered "strategic" and "depleted" under the MMPA. At the time the ENP stock was delisted, the WNP stock was thought to be geographically isolated from the ENP stock. NOAA (2018) initiated a 5-yr Status Review of WNP gray whales to ensure that the listing classification is accurate. This review is ongoing. Documentation of some whales moving between the WNP and ENP indicates otherwise (Lang *et al*. 2014; Mate *et al.* 2011; Weller *et al.* 2012; Urbán *et al.* 2019). Other research findings, however, provide continued support for identifying two separate stocks of North Pacific gray whales, including: (1) significant mitochondrial and nuclear genetic differences between whales that feed in the WNP and those that feed in the ENP (LeDuc *et al.* 2002; Lang *et al.* 2014), (2) recruitment into the WNP stock is almost exclusively internal (Cooke *et al.* 2013), (3) a SNP study that indicates the gray whale gene pool is differentiated into two populations (Brüniche-Olsen *et al.* 2018a) and (4) the abundance of the WNP stock remains low while the abundance of the ENP stock grew steadily following the end of commercial whaling (Cooke *et al.*2017). As long as the WNP stock remains listed as endangered under the ESA, it continues to be considered as depleted under the MMPA.The IWC Scientific Committee stock structure hypotheses are summarized in the Stock Definition and Geographic Range section of this report. Cooke *et al.* (2017) conducted an assessment of gray whales in the WNP using an individually-based stage-structured population model with modified stock definitions that allows for the possibility of multiple feeding/breeding groups. Cooke *et al.* (2017) noted that "there is preferential, but not exclusive, mating within the Sakhalin feeding aggregation. The hypothesis of mating exclusively within the Sakhalin feeding population is just rejected ($p < 0.05$). We conclude that the Sakhalin feeding aggregation is probably not genetically closed but that the Sakhalin and Kamchatka feeding aggregations, taken together, may be genetically closed. However, genetic data from Kamchatka would be required to confirm this." In this scenario, whales identified feeding off Sakhalin represent about 2/3 of the combined Sakhalin Island-Kamchatka subpopulation. Further substructure within the subpopulation was not excluded by Cooke *et al.* (2017), including the possibility of less than 50 mature whales that breed only in the WNP. Other IWC hypotheses include the possibility that the Western Breeding stock has been extirpated (IWC 2020).

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Revised 5/30/2023

# Humpback Whale (*Megaptera novaeangliae kuzira*) Central America / Southern Mexico - California-Oregon-Washington Stock

### Stock Definition and Geographic Range

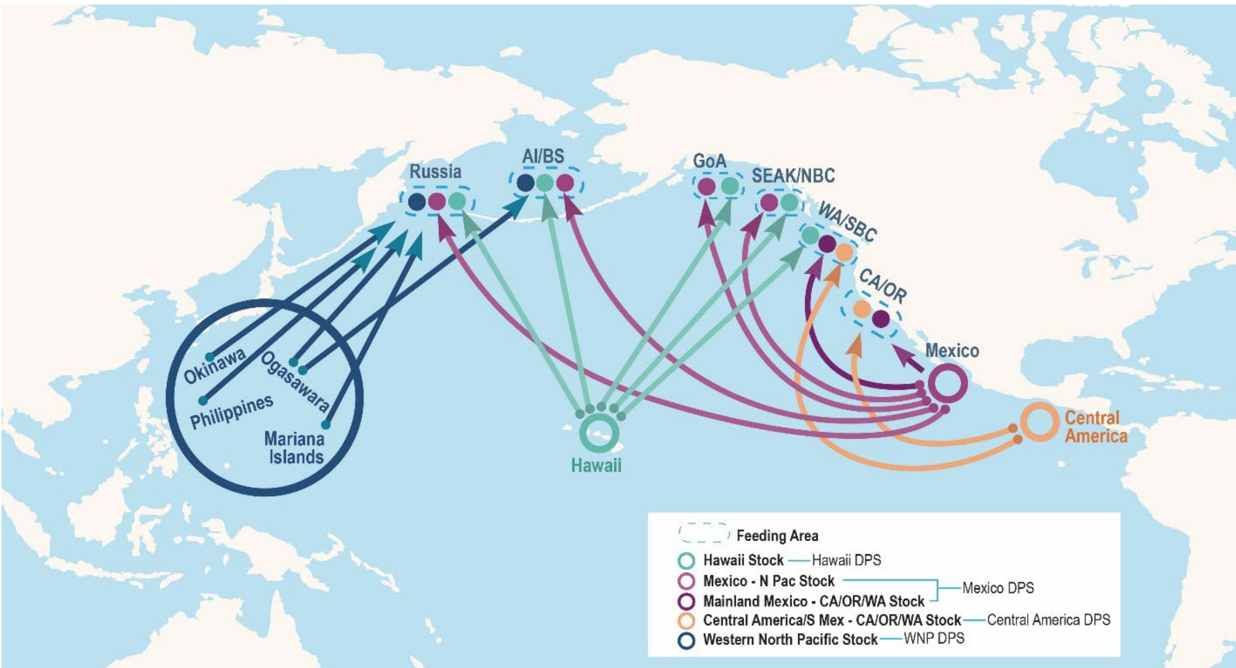


**Figure 1.** Pacific basin map showing wintering areas of five humpback whale stocks mentioned in this report. Also shown are summer feeding areas mentioned in the text. High-latitude summer feeding areas include Russia, Aleutian Islands / Bering Sea (AI/BS), Gulf of Alaska (GoA), Southeast Alaska / Northern British Columbia (SEAK/NBC), Washington / Southern British Columbia (WA/SBC), and California / Oregon (CA/OR).

Humpback whales occur worldwide and migrate seasonally from high latitude subarctic and temperate summering areas to low latitude subtropical and tropical wintering areas. Three subspecies are recognized globally (North Pacific, Atlantic, and Southern Hemisphere), based on restricted gene flow between ocean basins (Jackson *et al.* 2014). The North Pacific subspecies (*Megaptera novaeangliae kuzira*) occurs basin-wide, with summering areas in waters of the Russian Far East, Beaufort Sea, Bering Sea, Chukchi Sea, Gulf of Alaska, Western Canada, and the U.S. West Coast. Known wintering areas include waters of Okinawa and Ogasawara in Japan, Philippines, Mariana Archipelago, Hawaiian Islands, Revillagigedos Archipelago, Mainland Mexico, and Central America (Baker *et al.* 2013, Barlow *et al.* 2011, Calambokidis *et al.* 2008, Clarke *et al.* 2013, Fleming and Jackson 2011, Hashagen *et al.* 2009). In describing humpback whale population structure in the Pacific, Martien *et al.* (2020, 2023) note that 'migratory whale herds', defined as groups of animals that share the same summering and wintering area, are likely to be demographically independent due to their strong, maternally-inherited fidelity to migratory destinations. Despite whales from multiple wintering areas sharing some summer feeding areas, Baker *et al.* (2013) reported significant genetic differences between North Pacific summering and wintering areas, driven by strong maternal site fidelity to feeding areas and natal philopatry to wintering areas. This differentiation is supported by photo ID studies showing little interchange of whales between summering areas (Calambokidis *et al.* 2001).

NMFS has identified 14 distinct population segments (DPSs) of humpback whales worldwide under the Endangered Species Act (ESA) (81 FR 62259, September 8, 2016), based on genetics and movement data (Baker *et al.* 2013, Calambokidis *et al.* 2008, Bettridge *et al.* 2015). In the North Pacific, 4 DPSs are recognized (with ESA listing status), based on their respective low latitude wintering areas: "Western North Pacific" (endangered), "Hawai'i" (not listed), "Mexico" (threatened), and "Central America" (endangered). The listing status of each DPS was determined following an evaluation of the ESA section 4(a)(1) listing factors as well as an evaluation of demographic risk factors. The evaluation is summarized in the final rule revising the ESA listing status of humpback whales (81 FR 62259, September 8, 2016).

In prior stock assessments, NMFS designated three stocks of humpback whales in the North Pacific: the California/Oregon/Washington (CA/OR/WA) stock, consisting of winter populations in coastal Central America and coastal Mexico which migrate to the coast of California and as far north as southern British Columbia in summer; 2) the Central North Pacific stock, consisting of winter populations in the Hawaiian Islands which migrate primarily to northern British Columbia/Southeast Alaska, the Gulf of Alaska, and the Bering Sea/Aleutian Islands; and 3) the Western North Pacific stock, consisting of winter populations off Asia which migrate primarily to Russia and the Bering Sea/Aleutian Islands. These stocks, to varying extents, were not aligned with the more recently identified ESA DPSs (e.g., some stocks were composed of whales from more than one DPS), which led NMFS to reevaluate stock structure under the Marine Mammal Protection Act (MMPA).

NMFS evaluated whether these North Pacific DPSs contain one or more demographically independent populations (DIPs), where demographic independence is defined as "…the population dynamics of the affected group is more a consequence of births and deaths within the group (internal dynamics) rather than immigration or emigration (external dynamics)" (NMFS 2023). Evaluation of the four DPSs in the North Pacific by NMFS resulted in the delineation of three DIPs, as well as four "units" that may contain one or more DIPs (Martien *et al.* 2021, Taylor *et al.* 2021, Wade *et al.* 2021, Oleson *et al.* 2022, Table 1). Delineation of DIPs is based on evaluation of 'strong lines of evidence' such as genetics, movement data, and morphology (Martien *et al.* 2019). From these DIPs and units, NMFS designated five stocks. North Pacific DIPs / units / stocks are described below, along with the lines of evidence used for each. In some cases, multiple units may be combined into a single stock due to lack of sufficient data and/or analytical tools necessary for effective management or for pragmatic reasons (NMFS 2019).

Table 1. DPS of origin for North Pacific humpback whale DIPs, units, and stocks. Names are based on their general winter and summering area linkages. The stock included in this report is shown in bold font. All others appear in separate reports.

| DPS | ESA Status | DIPs / units | Stocks |
|---|---|---|---|
| Central America | Endangered | Central America - CA-OR-WA DIP | **Central America / Southern Mexico - CA-OR-WA stock** |
| Mexico | Threatened | Mainland Mexico - CA-OR-WA DIP | Mainland Mexico - CA-OR-WA stock |
| | | Mexico - North Pacific unit | Mexico - North Pacific stock |
| Hawai'i | Not Listed | Hawai'i - North Pacific unit | Hawai'i stock |
| | | Hawai'i - Southeast Alaska / Northern British Columbia DIP | |

| | | | |
|---|---|---|---|
| Western North Pacific | Endangered | Philippines / Okinawa - North Pacific unit | Western North Pacific stock |
| | | Marianas / Ogasawara - North Pacific unit | |

Delineation of the Central America/Southern Mexico – California/Oregon/Washington DIP is based on two strong lines of evidence indicating demographic independence: genetics and movement data (Taylor *et al.* 2021). The DIP was designated as a stock because available data make it feasible to manage as a stock and because there are conservation and management benefits to doing so (NMFS 2019, NMFS 2022a, NMFS 2023). Whales in this stock winter off the Pacific coast of Nicaragua, Honduras, El Salvador, Guatemala, Panama, Costa Rica and likely southern coastal Mexico (Taylor *et al.* 2021). Summer destinations for whales in this DIP include the U.S. West Coast waters of California, Oregon, and Washington (including the Salish Sea, Calambokidis *et al.* 2017).

Delineation of the Mainland Mexico – California/Oregon/Washington DIP is based on two strong lines of evidence indicating demographic independence: genetics and movement data (Martien *et al.* 2021). The DIP was designated as a stock because available data make it feasible to manage as a stock and because there are conservation and management benefits to doing so (NMFS 2019, NMFS 2022b, NMFS 2023). Whales in this stock winter off the mainland Mexico states of Nayarit and Jalisco, with some animals seen as far south as Colima and Michoacán. Summer destinations for whales in the Mainland Mexico DPS include U.S. West Coast waters of California, Oregon, Washington (including the Salish Sea, Martien *et al.* 2021), Southern British Columbia, Alaska, and the Bering Sea.

The Mexico – North Pacific unit is likely composed of multiple DIPs, based on movement data (Martien *et al.* 2021, Wade 2021, Wade *et al.* 2021). However, because currently available data and analyses are not sufficient to delineate or assess DIPs within the unit, it was designated as a single stock (NMFS 2019, NMFS 2022b, NMFS 2023). Whales in this stock winter off Mexico and the Revillagigedo Archipelago and summer primarily in Alaska waters (Martien *et al.* 2021).

The Hawaiʻi stock consists of one DIP - Hawaiʻi - Southeast Alaska / Northern British Columbia DIP and one unit - Hawaiʻi - North Pacific unit, which may or may not be composed of multiple DIPs (Wade *et al.* 2021). The DIP and unit are managed as a single stock at this time, due to the lack of data available to separately assess them and lack of compelling conservation benefit to managing them separately (NMFS 2019, NMFS 2022c, NMFS 2023). The DIP is delineated based on two strong lines of evidence: genetics and movement data (Wade *et al.* 2021). Whales in the Hawaiʻi - Southeast Alaska/Northern British Columbia DIP winter off Hawaiʻi and largely summer in Southeast Alaska and Northern British Columbia (Wade *et al*. 2021). The group of whales that migrate from Russia, western Alaska (Bering Sea and Aleutian Islands), and central Alaska (Gulf of Alaska excluding Southeast Alaska) to Hawaiʻi have been delineated as the Hawaiʻi-North Pacific unit (Wade *et al.* 2021). There are a small number of whales that migrate between Hawaiʻi and southern British Columbia/Washington, but current data and analyses do not provide a clear understanding of which unit these whales belong to (Wade et al. 2021).

The Western North Pacific (WNP) stock consists of two units- the Philippines / Okinawa - North Pacific unit and the Marianas / Ogasawara - North Pacific unit. The units are managed as a single stock at this time, due to a lack of data available to separately assess them (NMFS 2019, NMFS 2022d, NMFS 2023). Recognition of these units is based on movements and genetic data (Oleson *et al.* 2022). Whales in the Philippines/Okinawa - North Pacific unit winter near the Philippines and in the Ryukyu Archipelago and migrate to summer feeding areas primarily off the Russian mainland (Oleson *et al.* 2022). Whales that winter off the Mariana Archipelago, Ogasawara, and other areas not yet identified and then migrate to summer feeding areas off the Commander Islands, and to the Bering Sea and Aleutian Islands comprise the Marianas/Ogasawara - North Pacific unit.

This stock assessment report includes information on the **Central America/Southern Mexico – California-Oregon-Washington stock** (Figure 2). In previous marine mammal stock assessments, humpback whales that summer and feed off California, Oregon, and Washington were treated as a single stock ("California-Oregon-Washington"), that included whales from three DPSs (Central America, Mexico, Hawaiʻi), defined by separate wintering areas. Some Hawai'i stock whales occur in Washington state and Southern British Columbia waters during summer (Calambokidis and Barlow 2020, Wade 2021), but the proportions using Washington vs Southern British Columbia waters during summer is unknown. The previous "California-Oregon-Washington" stock included animals from multiple DIPs (Central America – California-Oregon-Washington DIP, Mainland Mexico – California-Oregon-Washington DIP, and Hawai'i), which is inconsistent with management goals under the MMPA (NMFS 2019).

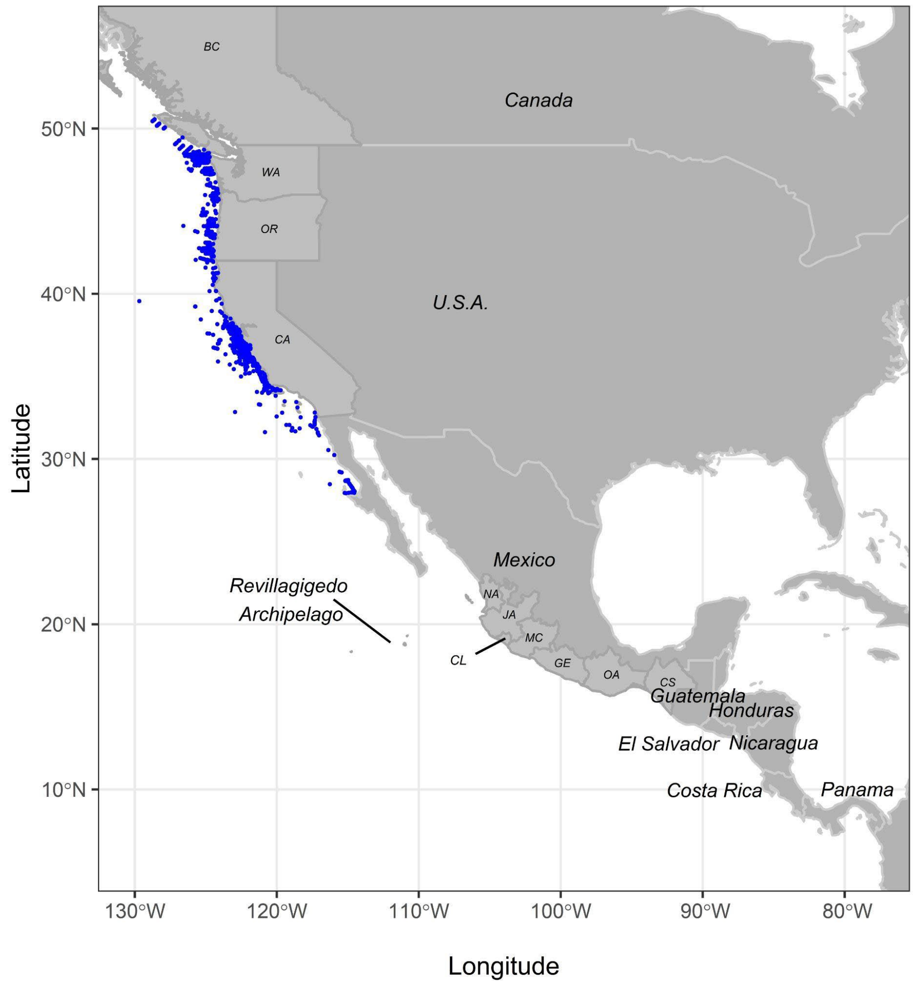


**Figure 2**. Wintering and summering areas for the Central America / Southern Mexico - CA-OR-WA stock of humpback whales. The primary wintering areas of the Central America / Southern Mexico - CA-OR-WA stock include the Pacific coasts of Nicaragua, Honduras, El Salvador, Guatemala, Panama, Costa Rica, with animals sometimes sighted as far north as Michoacán and Colima. Primary summering areas of whales from this stock include California and Oregon, with only a few individuals identified in the northern Washington/southern British Columbia feeding areas. Summering area sightings from 1991 - 2018 NMFS/SWFSC research vessel line-transect surveys are shown as blue dots and primarily represent whales from two stocks: the Central America / Southern Mexico - CA-OR-WA stock and Mainland Mexico - CA-OR-WA stock, although small numbers of whales from the Hawai'i stock also have been matched to WA and Southern British Columbia (Wade 2021). Country and state names abbreviations from north to south are: BC = British Columbia, WA = Washington state, OR = Oregon, CA = California, U.S.A. = United States of America, NA = Nayarit, JA = Jalisco, CL = Colima, MC = Michoacán, GE = Guerrero, OA = Oaxaca, and CS = Chiapas.

**Population Size**

Curtis *et al.* (2022) estimated the population size of whales wintering in southern Mexico and Central America using spatial capture-recapture methods based on photographic data collected between 2019 and 2021. Their estimate of abundance is 1,496 (CV=0.171) whales and this represents the best estimate of abundance for the Central America / Southern Mexico - CA-OR-WA stock of humpback whales.

**Minimum Population Estimate**

The minimum population estimate for this stock is taken as the lower 20th percentile of the capture-recapture estimate from Curtis *et al.* (2022), or 1,284 whales.

**Current Population Trend**

The 2019-2021 abundance estimate for the Central America / Southern Mexico - CA/OR/WA stock (1,496, CV=0.171) is almost double the 2004-2006 estimate that excludes whales from southern Mexico (755 whales, CV=0.242) (Wade 2021). Given the time elapsed between the two estimates, if the increase were due purely to population growth, it would suggest an annual growth rate of approximately 4.7% (Curtis *et al.* 2022), which is lower than the 8.2% annual increase observed for the U.S. West Coast since 1989 (Calambokidis and Barlow 2020). Given inclusion of whales from southern Mexico in the current estimate, Curtis *et al.* (2022) derived a population growth rate for Central America / Southern Mexico whales based on differences between 2004-2006 estimates and the current estimate by excluding whales from southern Mexico, yielding an annual growth rate of 1.6% (SD = 2.0%) for this stock of humpback whales. However, the estimate has high uncertainty (Curtis *et al.* 2022).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Calambokidis and Barlow (2020) reported humpback whale abundance increased 8.2% annually in the California Current since 1989, based on mark-recapture estimates largely restricted to whales summering in California and Oregon waters. However, these estimates include whales from the Central America / Southern Mexico - CA/OR/WA stock and the Mainland Mexico - CA/OR/WA stocks. The maximum net productivity rate for the Central America / Southern Mexico - CA/OR/WA stock is unknown. However, the maximum net productivity rate can be taken to be at least as high as the maximum observed for the combined stocks, or 8.2% annually, though it could be higher if one of the stocks is growing faster than another.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,284), times one-half the estimated population growth rate for this stock of humpback whales (½ of 8.2%) times a recovery factor of 0.1 (for an endangered stock with $N_{min}$<1,500; NMFS 2023), resulting in a PBR of 5.2. Ryan *et al.* (2019) summarizes sighting and acoustic data, noting that humpbacks are present in central California waters at least 8 months annually, with December and April representing 'transition months', where whales are moving out of / into the region. Counting December and April each as one-half month of residency time during migration, plus the 7 months of May through November when whales are abundant, yields 8 months of residency time, or ⅔ of the year. This may be considered a minimum residency time, as some whales are still in U.S. waters from December to April. Therefore, the total PBR for this stock (5.2) is prorated by ⅔, to yield a PBR in U.S. waters of 3.5 whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Human-caused mortality and serious injury (MSI) of humpback whales in CA-OR-WA summer feeding areas includes whales from three stocks: **Central America / Southern Mexico – CA-OR-WA**; **Mainland Mexico – CA-OR-WA**; and **Hawai'i**. Where multiple stocks share a summer feeding area, total human-caused MSI for the mixed-stock area may be prorated to each stock using point estimates of summer to winter area movement probabilities in Wade (2021) (Table 2). Human-caused MSI from CA-OR-WA waters for the Hawai'i stock is summarized in the Alaska stock assessments (Young *et al*. 2023).

**Table 2**. Summer to winter area movement probabilities from Wade (2021) used for prorating human-caused MSI to stocks of humpback whales using CA/OR/WA waters in summer.

| | **Location of MSI** | |
|---|---|---|
| **Stock** | **California or Oregon** | **Washington** |
| Central America/Southern Mexico - CA/OR/WA | 0.423 (CV=0.23) | 0.059 (CV=0.935) |
| Hawai'i | 0.00 | 0.688 (CV=0.13) |
| Mainland Mexico – CA/OR/WA | 0.577 (CV=0.169) | 0.254 (CV=0.278) |

**Fishery Information**
U.S. Commercial Fisheries

**Table 3**. Sources of humpback whale MSI in California (CA), Oregon (OR), and Washington (WA) commercial fisheries from 2016-2020, unless noted otherwise (Carretta 2022, Carretta *et al.* 2022, Jannot *et al.* 2021). Records include entanglements detected outside of U.S. waters, but confirmed to involve U.S. fisheries. Most cases represent strandings and at-sea sightings of entangled whales. Cases of entangled *unidentified whales* are prorated to humpback whale based on location, depth, and time of year (Carretta 2018). Sources derived from observer programs with statistical estimates of bycatch and uncertainty are shown with coefficients of variation (CV) where available. Totals for the Central America / Southern Mexico – CA-OR-WA stock are based on prorating CA-OR-WA cases by summer to winter area movement probabilities in Table 2. In 2017, there was one non-serious injury of a humpback involving two gear types: CA Coonstripe Shrimp Pot and WA/OR/CA Sablefish Pot.

| Fishery | Cases | All CA-OR-WA Humpback Stocks ∑ MSI | Central America/Southern Mexico – CA-OR-WA Stock ∑ MSI | Central America/Southern Mexico – CA-OR-WA Stock Mean Annual MSI |
|---|---|---|---|---|
| CA Spot Prawn Trap | 5 | 3.25 | 1.37 | 0.275 |
| CA Dungeness Crab Pot | 34 | 23.75 | 10.05 | 2.01 |
| Dungeness Crab Pot (Commercial, state unknown) | 2 | 2 | 0.846 | 0.169 |
| OR Dungeness Crab Pot | 2 | 1.75 | 0.740 | 0.148 |
| WA Coastal Dungeness Crab Pot | 7 | 5.5 | 0.324 | 0.065 |
| Gillnet Fishery | 6 | 2 | 0.846 | 0.169 |
| Unidentified Fishery Interaction (whales identified as humpback) | 58 | 43.75 | 17.60 | 3.52 |
| Unidentified Fishery Interaction (unidentified whales prorated to humpback) | 7 | 5.25 | 2.22 | 0.44 |
| Unidentified Pot/Trap Fishery Entanglement | 13 | 9.5 | 3.11 | 0.622 |
| WA/OR/CA Sablefish Pot[1] | 2 | 7.82 (CV>0.8) | 3.31 (CV>0.8) | 0.661 (CV>0.8) |
| CA Swordfish and Thresher Shark Drift Gillnet (Observer Program)[2] | 0 | 0.1 (CV>0.8) | 0.042 (CV>0.8) | ≈0.01 (CV>0.8) |
| **Totals** | **136** | **104.7** | **40.45** | **8.1** |

**Other human-caused mortality and serious injury**

Non-commercial sources of human-caused MSI, including tribal fisheries, recreational fisheries, marine debris (including research buoys) and vessel strikes are also responsible for a fraction of reported cases annually (Carretta *et al.* 2022). These sources and case totals are summarized in Tables 4 and 5.

[1] Estimates are based on 2015-2019 data (Jannot et al. 2021) for the limited entry (LE) and open-access (OA) sablefish pot sectors combined. Two observer program entanglements since 2002 informed the bycatch estimates, both of which occurred in CA + OR waters. Other sablefish pot cases opportunistically reported (at-sea sightings of entangled whales, strandings) also occurred in CA/OR waters (Carretta et al. 2022). Estimates from Jannot et al. (2021) are used in this stock assessment report because annual MSI totals are higher than those reported based on opportunistic sightings (Carretta et al. 2022). Annual observer coverage varies between 14% and 72% for the LE fleet and between 2% and 12% for the OA fleet (Somers et al., 2020).

[2] There were no observed entanglements during 2016-2020 with 21% observer coverage, however the model-based estimate of bycatch is based on pooling 1990-2000 data, resulting in a small positive estimate (Carretta 2022).

Marine Debris, Recreational and Tribal Fisheries

**Table 4.** Sources of MSI from marine debris, recreational, and tribal fisheries from 2016-2020 summarized in Carretta *et al*. (2022).

| Source | Cases | All CA-OR-WA Humpback Stocks ∑ MSI | Central America/Southern Mexico – CA-OR-WA Stock ∑ MSI | Central America/Southern Mexico – CA-OR-WA Stock Mean Annual MSI |
|---|---|---|---|---|
| Dungeness Crab Pot Fishery (Recreational) | 2 | 1 | 0.423 | 0.085 |
| Gillnet Fishery, Tribal | 3 | 2.5 | 0.148 | 0.0295 |
| Hook And Line Fishery | 1 | 0.75 | 0.317 | 0.063 |
| Marine Debris | 1 | 1 | 0.423 | 0.085 |
| Pot Fishery, Tribal | 1 | 1 | 0.423 | 0.085 |
| Spot Prawn Trap/Pot Fishery (Recreational) | 1 | 0 | 0 | 0 |
| **Totals** | **9** | **6.25** | **1.73** | **0.35** |

**Vessel Strikes**

Fourteen vessel strike cases involving humpback whales were observed in CA-OR-WA waters during 2016-2020 (8 in CA, 1 in OR, and 5 in WA), totaling 13.2 MSI, or 2.6 whales per year (Carretta *et al.* 2022). Most vessel strikes are likely undetected and thus, we use estimates of vessel strike mortality reported by Rockwood *et al.* (2017) for this region. The estimated number of annual vessel strike deaths was 22 humpback whales, though this includes only the period July – November when whales are most likely to be present in the U.S. West Coast EEZ and the season that overlaps with survey effort used in species distribution models (Becker *et al.* 2016, Rockwood *et al.* 2017). This estimate is based on an assumption of a moderate level of vessel avoidance by humpback whales, as measured by the behavior of satellite-tagged whales in the presence of vessels (McKenna *et al.* 2015). Based on estimates of 22 deaths due to vessel strikes annually, the number attributed to the Central America / Southern Mexico - CA-OR-WA stock during 2016-2020 is 6.45 whales per year (Table 5). The estimated mortality of 6.45 humpback whales annually due to vessel strikes represents approximately 0.4% of the stock's estimated population size (6.45 deaths / 1,496 whales). The ratio of mean annual observed to estimated vessel strike deaths and serious injuries of humpback whales during 2016-2020 is 2.6 / 22 = 0.11, implying that vessel strike counts from opportunistic observations represent a small fraction of overall incidents.

**Table 5.** Summary of humpback whale vessel strike MSI during 2016-2020 (Carretta *et al*. 2022). Estimates are based on prorating annual estimates of humpback vessel strike mortality in this region (22/yr, Rockwood *et al*. 2017) by the fraction of observed vessel strikes in different feeding areas (WA vs CA/OR), which are then prorated to stock by summer to winter area movement probabilities from Wade (2021).

| State Detected | Observations | Fraction of Observations | Fraction of Observations *times* 22 MSI/yr estimated by Rockwood et al. (2017) | Central America/Southern Mexico – CA-OR-WA stock prorated ∑ MSI based on summer to winter area movement probabilities (Wade 2021) |
|---|---|---|---|---|
| WA | 5 | 0.357 | 7.86 | 0.463 |
| CA/OR | 9 | 0.643 | 14.14 | 5.98 |
| **Total** | **14** | | | **6.45** |

Vessel strikes in U.S. West Coast EEZ waters continue to impact large whales (Redfern *et al.* 2013; 2019; Moore *et al.* 2018). A complex of diverse vessel types, speeds, and destination ports all contribute to variability in vessel traffic and these factors may be influenced by economic and regulatory changes. For example, Moore *et al.*

(2018) found that primary routes traveled by vessels changed when emission control areas (ECAs) were established off the U.S. West Coast. They also found that large vessels typically reduced their speed by 3-6 kts in ECAs between 2008 and 2015. The speed reductions are thought to be a strategy to reduce operating costs associated with more expensive, cleaner burning fuels required within the ECAs. In contrast, Moore *et al.* (2018) noted that some vessels increased speed when transiting longer routes to avoid the ECAs. Further research is ongoing to understand how variability in vessel traffic affects vessel strike risk and mitigation strategies, though Redfern *et al.* (2019) note that a combination of vessel speed reductions and expansion of areas to be avoided should be considered. Rockwood *et al.* (2017) note that 82% of humpback whale vessel strike mortalities occur within 10% of the region, implying that vessel strike mitigation measures may be effective if applied over relatively small regions.

**Historic whaling**

Approximately 15,000 humpback whales were taken from the North Pacific from 1919 to 1987 (Tonnessen and Johnsen 1982), of these, approximately 8,000 were from the west coast of Baja California, California, Oregon and Washington (Rice 1978). Shore-based whaling depleted the humpback whale stock off California twice: once prior to 1925 (Clapham *et al.* 1997) and again between 1956 and 1965 (Rice 1974). There has been a prohibition on taking humpback whales since 1966.

**Habitat Concerns**

Increasing levels of anthropogenic sound in the world's oceans (Andrew *et al.* 2002), such as those produced by shipping traffic, or Low Frequency Active sonar, is a habitat concern for whales, as it can reduce acoustic space used for communication (masking) (Clark *et al.* 2009, NOAA 2016c). This can be particularly problematic for baleen whales that may communicate using low-frequency sound (Erbe 2016). Based on vocalizations (Richardson *et al.* 1995; Au *et al.* 2006), reactions to sound sources (Lien *et al.* 1990, 1992; Maybaum 1993), and anatomical studies (Hauser *et al.* 2001), humpback whales also appear to be sensitive to mid-frequency sounds, including those used in active sonar military exercises (U.S. Navy 2007).

Seven important feeding areas for humpback whales are identified off the U.S. west coast by Calambokidis *et al.* (2015), including five in California, one in Oregon, and one in Washington. Humpback whales have increasingly reoccupied areas inside of Puget Sound (the 'Salish Sea'), a region where they were historically abundant prior to whaling (Calambokidis *et al.* 2017).

**STATUS OF STOCK**

The Central America / Southern Mexico - CA-OR-WA stock of humpback whales is a DIP delineated from the 'Central America DPS' of humpback whales listed as endangered under the ESA (Bettridge *et al.* 2015, Taylor *et al.* 2021), and is therefore considered 'depleted' and 'strategic' under the MMPA. Total annual human-caused serious injury and mortality of humpback whales is the sum of commercial fishery MSI (8.1/yr) + vessel strikes (6.45/yr), + non-commercial sources of MSI (0.35/yr), or 14.9 humpback whales annually. Total commercial fishery MSI (8.1/yr) is greater than the calculated PBR (3.5) for this stock, thus, it is not approaching zero mortality and serious injury rate. There is no estimate of the undocumented fraction of anthropogenic injuries and deaths to humpback whales on the U.S. West Coast, but for vessel strikes, a comparison of observed vs. estimated annual vessel strikes suggests that approximately 10% of vessel strikes are documented. The stock is estimated to have grown at 1.6% annually (SD = 2.0%) between 2004-2006 and 2019-2021 Curtis *et al.* (2022), but this estimate has high uncertainty. Habitat concerns include sensitivity to anthropogenic sound sources.

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Revised 5/30/2023

# Humpback Whale (*Megaptera novaeangliae kuzira*)
# Mainland Mexico - California-Oregon-Washington Stock

## Stock Definition and Geographic Range

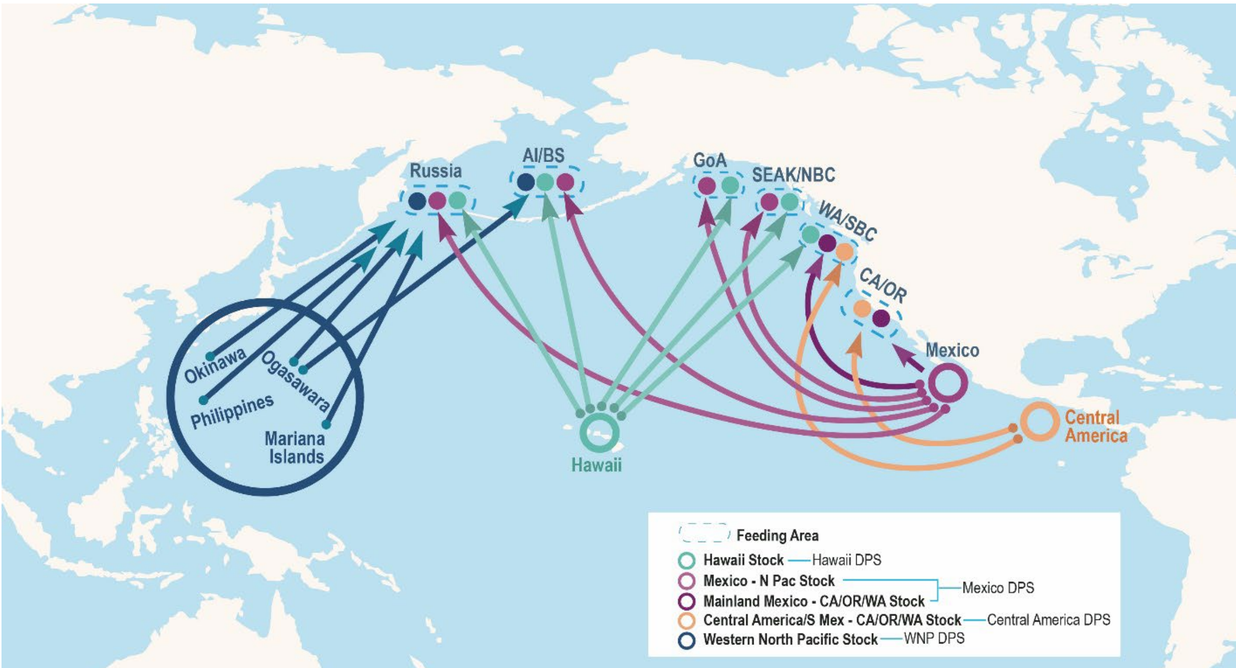


**Figure 1**. Pacific basin map showing wintering areas of five humpback whale stocks mentioned in this report. Also shown are summering feeding areas mentioned in the text. High-latitude summer feeding areas include Russia, Aleutian Islands / Bering Sea (AI/BS), Gulf of Alaska (GoA), Southeast Alaska / Northern British Columbia (SEAK/NBC), Washington / Southern British Columbia (WA/SBC), and California / Oregon (CA/OR).

Humpback whales occur worldwide and migrate seasonally from high latitude subarctic and temperate summering areas to low latitude subtropical and tropical wintering areas. Three subspecies are recognized globally (North Pacific, Atlantic, and Southern Hemisphere), based on restricted gene flow between ocean basins (Jackson *et al.* 2014). The North Pacific subspecies (*Megaptera novaeangliae kuzira*) occurs basin-wide, with summering areas in waters of the Russian Far East, Beaufort Sea, Bering Sea, Chukchi Sea, Gulf of Alaska, Western Canada, and the U.S. West Coast. Known wintering areas include waters of Okinawa and Ogasawara in Japan, Philippines, Mariana Archipelago, Hawaiian Islands, Revillagigedos Archipelago, Mainland Mexico, and Central America (Baker *et al.* 2013, Barlow *et al.* 2011, Calambokidis *et al.* 2008, Clarke *et al.* 2013, Fleming and Jackson 2011, Hashagen *et al.* 2009). In describing humpback whale population structure in the Pacific, Martien *et al.* (2020, 2023) note that 'migratory whale herds', defined as groups of animals that share the same summering and wintering area, are likely to be demographically independent due to their strong, maternally-inherited fidelity to migratory destinations. Despite whales from multiple wintering areas sharing some summer feeding areas, Baker *et al.* (2013) reported significant genetic differences between North Pacific summering and wintering areas, driven by strong maternal site fidelity to feeding areas and natal philopatry to wintering areas. This differentiation is supported by photo ID studies showing little interchange of whales between summering areas (Calambokidis *et al.* 2001).

NMFS has identified 14 distinct population segments (DPSs) of humpback whales worldwide under the Endangered Species Act (ESA) (81 FR 62259, September 8, 2016), based on genetics and movement data (Baker *et al.* 2013, Calambokidis *et al.* 2008, Bettridge *et al.* 2015). In the North Pacific, 4 DPSs are recognized (with ESA listing status), based on their respective low latitude wintering areas: "Western North Pacific" (endangered), "Hawai'i" (not listed), “Mexico” (threatened), and “Central America” (endangered). The listing status of each DPS was determined following an evaluation of the ESA section 4(a)(1) listing factors as well as an evaluation of demographic risk factors. The evaluation is summarized in the final rule revising the ESA listing status of humpback whales (81 FR 62259, September 8, 2016).

In prior stock assessments, NMFS designated three stocks of humpback whales in the North Pacific: the California/Oregon/Washington (CA/OR/WA) stock, consisting of winter populations in coastal Central America and coastal Mexico which migrate to the coast of California and as far north as southern British Columbia in summer; 2) the Central North Pacific stock, consisting of winter populations in the Hawaiian Islands which migrate primarily to northern British Columbia/Southeast Alaska, the Gulf of Alaska, and the Bering Sea/Aleutian Islands; and 3) the Western North Pacific stock, consisting of winter populations off Asia which migrate primarily to Russia and the Bering Sea/Aleutian Islands. These stocks, to varying extents, were not aligned with the more recently identified ESA DPSs (e.g., some stocks were composed of whales from more than one DPS), which led NMFS to reevaluate stock structure under the Marine Mammal Protection Act (MMPA).

NMFS evaluated whether these North Pacific DPSs contain one or more demographically independent populations (DIPs), where demographic independence is defined as “…the population dynamics of the affected group is more a consequence of births and deaths within the group (internal dynamics) rather than immigration or emigration (external dynamics)” (NMFS 2023). Evaluation of the four DPSs in the North Pacific by NMFS resulted in the delineation of three DIPs, as well as four “units” that may contain one or more DIPs (Martien *et al.* 2021, Taylor *et al.* 2021, Wade *et al.* 2021, Oleson *et al.* 2022, Table 1). Delineation of DIPs is based on evaluation of ‘strong lines of evidence’ such as genetics, movement data, and morphology (Martien *et al.* 2019). From these DIPs and units, NMFS designated five stocks. North Pacific DIPs / units / stocks are described below, along with the lines of evidence used for each. In some cases, multiple units may be combined into a single stock due to lack of sufficient data and/or analytical tools necessary for effective management or for pragmatic reasons (NMFS 2019).

Table 1. DPS of origin for North Pacific humpback whale DIPs, units, and stocks. Names are based on their general winter and summering area linkages. The stock included in this report is shown in bold font. All others appear in separate reports.

| DPS | ESA Status | DIPs / units | Stocks |
|---|---|---|---|
| Central America | Endangered | Central America - CA-OR-WA DIP | Central America / Southern Mexico - CA-OR-WA stock |
| Mexico | Threatened | Mainland Mexico - CA-OR-WA DIP | **Mainland Mexico - CA-OR-WA stock** |
| | | Mexico - North Pacific unit | Mexico - North Pacific stock |
| Hawai‘i | Not Listed | Hawai‘i - North Pacific unit | Hawai‘i stock |
| | | Hawai‘i - Southeast Alaska / Northern British Columbia DIP | |

| | | | |
|---|---|---|---|
| Western North Pacific | Endangered | Philippines / Okinawa - North Pacific unit | Western North Pacific stock |
| | | Marianas / Ogasawara - North Pacific unit | |

Delineation of the Central America/Southern Mexico – California/Oregon/Washington DIP is based on two strong lines of evidence indicating demographic independence: genetics and movement data (Taylor *et al.* 2021). The DIP was designated as a stock because available data make it feasible to manage as a stock and because there are conservation and management benefits to doing so (NMFS 2023, NMFS 2019, NMFS 2022a). Whales in this stock winter off the Pacific coast of Nicaragua, Honduras, El Salvador, Guatemala, Panama, Costa Rica and likely southern coastal Mexico (Taylor *et al.* 2021). Summer destinations for whales in this DIP include the U.S. West Coast waters of California, Oregon, and Washington (including the Salish Sea, Calambokidis *et al.* 2017).

Delineation of the Mainland Mexico – California/Oregon/Washington DIP is based on two strong lines of evidence indicating demographic independence: genetics and movement data (Martien *et al.* 2021). The DIP was designated as a stock because available data make it feasible to manage as a stock and because there are conservation and management benefits to doing so (NMFS 2023, NMFS 2019, NMFS 2022b). Whales in this stock winter off the mainland Mexico states of Nayarit and Jalisco, with some animals seen as far south as Colima and Michoacán. Summer destinations for whales in the Mainland Mexico DPS include U.S. West Coast waters of California, Oregon, Washington (including the Salish Sea, Martien *et al.* 2021), Southern British Columbia, Alaska, and the Bering Sea.

The Mexico – North Pacific unit is likely composed of multiple DIPs, based on movement data (Martien *et al.* 2021, Wade 2021, Wade *et al.* 2021). However, because currently available data and analyses are not sufficient to delineate or assess DIPs within the unit, it was designated as a single stock (NMFS 2023, NMFS 2019, NMFS 2022b). Whales in this stock winter off Mexico and the Revillagigedo Archipelago and summer primarily in Alaska waters (Martien *et al.* 2021).

The Hawai‘i stock consists of one DIP - Hawai‘i - Southeast Alaska / Northern British Columbia DIP and one unit - Hawai‘i - North Pacific unit, which may or may not be composed of multiple DIPs (Wade *et al.* 2021). The DIP and unit are managed as a single stock at this time, due to the lack of data available to separately assess them and lack of compelling conservation benefit to managing them separately (NMFS 2023, NMFS 2019, NMFS 2022c). The DIP is delineated based on two strong lines of evidence: genetics and movement data (Wade *et al.* 2021). Whales in the Hawai‘i - Southeast Alaska/Northern British Columbia DIP winter off Hawai‘i and largely summer in Southeast Alaska and Northern British Columbia (Wade *et al*. 2021). The group of whales that migrate from Russia, western Alaska (Bering Sea and Aleutian Islands), and central Alaska (Gulf of Alaska excluding Southeast Alaska) to Hawai‘i have been delineated as the Hawai‘i-North Pacific unit (Wade *et al.* 2021). There are a small number of whales that migrate between Hawai‘i and southern British Columbia/Washington, but current data and analyses do not provide a clear understanding of which unit these whales belong to (Wade et al. 2021).

The Western North Pacific (WNP) stock consists of two units- the Philippines / Okinawa - North Pacific unit and the Marianas / Ogasawara - North Pacific unit. The units are managed as a single stock at this time, due to a lack of data available to separately assess them (NMFS 2023, NMFS 2019, NMFS 2022d). Recognition of these units is based on movements and genetic data (Oleson *et al.* 2022). Whales in the Philippines/Okinawa - North Pacific unit winter near the Philippines and in the Ryukyu Archipelago and migrate to summer feeding areas primarily off the Russian mainland (Oleson *et al.* 2022). Whales that winter off the Mariana Archipelago, Ogasawara, and other areas not yet identified and then migrate to summer feeding areas off the Commander Islands, and to the Bering Sea and Aleutian Islands comprise the Marianas/Ogasawara - North Pacific unit.

This stock assessment report includes information on the **Mainland Mexico – California-Oregon-Washington stock** (Figure 2). In previous marine mammal stock assessments, humpback whales that summer and feed off California, Oregon, and Washington were treated as a single stock (“California-Oregon-Washington”), but included whales from three DPSs (Central America, Mexico, Hawai‘i) defined by separate wintering areas. Some Hawai’i stock whales occur in Washington state and Southern British Columbia waters during summer (Calambokidis and Barlow 2020, Wade 2021), but the proportions using Washington vs Southern British Columbia waters during summer is unknown. The previous "California-Oregon-Washington stock" also included multiple DIPs (Central America – California-Oregon-Washington DIP and Mainland Mexico – California-Oregon-Washington DIP**)**, which is inconsistent with management goals under the MMPA (NMFS 2019).

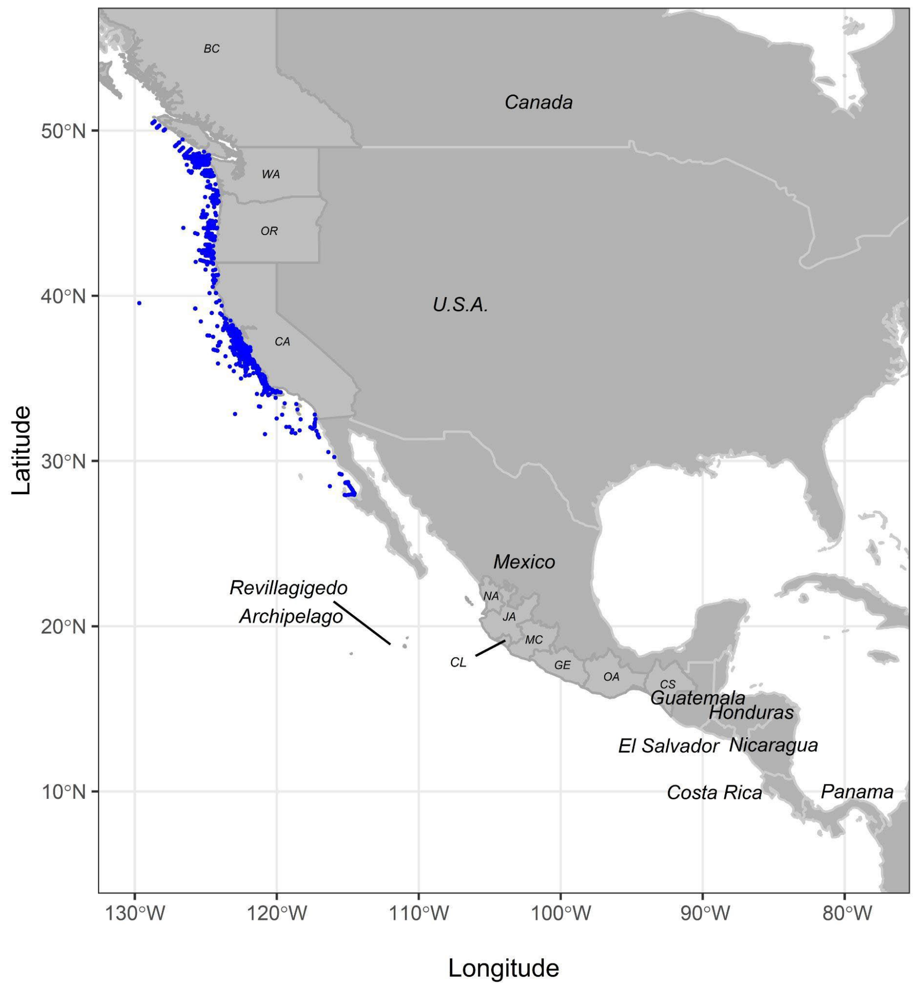


**Figure 2**. Wintering and summering areas for the Mainland Mexico - CA-OR-WA stock of humpback whales. The primary wintering areas of the Mainland Mexico - CA-OR-WA stock include the mainland Mexico states of Nayarit and Jalisco, with some animals seen as far south as Colima and Michoacán. Summer destinations for whales in the Mainland Mexico - CA-OR-WA stock include U.S. West Coast waters of California, Oregon, Washington, Southern British Columbia, Alaska, and the Bering Sea. Summering area sightings from 1991 - 2018 NMFS/SWFSC research vessel line-transect surveys are shown as blue dots and primarily represent whales from two stocks: the Central America / Southern Mexico - CA-OR-WA stock and Mainland Mexico - CA-OR-WA stock, although whales from the Hawai'i stock also have been matched to WA and Southern British Columbia (Wade 2021). Country and state names abbreviations from north to south are: BC = British Columbia, WA = Washington state, OR = Oregon, CA = California, U.S.A. = United States of America, NA = Nayarit, JA = Jalisco, CL = Colima, MC = Michoacán, GE = Guerrero, OA = Oaxaca, and CS = Chiapas.

**Population Size**

Curtis *et al.* (2022) estimated the abundance of whales wintering in southern Mexico and Central America using spatial capture-recapture methods based on photo-ID data collected between 2019 and 2021. Their estimate of abundance for the Central America / Southern Mexico – CA-OR-WA DIP is 1,496 (CV=0.171) whales. Given the availability of this estimate and a recent estimate of total abundance in the U.S. West Coast EEZ of 4,973 (CV=0.048) whales from mark-recapture (Calambokidis and Barlow 2020), Curtis *et al*. (2022) also estimated the abundance of whales from the Mainland Mexico – CA-OR-WA DIP as the difference, or 3,477 animals (CV=0.101). This may be an underestimate, because the estimate from Calambokidis and Barlow (2020) did not include photo-IDs off Washington state, but these authors state their estimate likely includes whales from Washington waters, since there is movement of whales between Washington and California and Oregon. Another estimate, based on a species distribution model from 2018 line-transect data, <u>resulted in a</u> <u>lower abundance of 4,784 whales</u> (CV=0.31) (Becker *et al.* 2020). This lends support to the statement of Calambokidis and Barlow (2020) that their estimate includes whales using Washington waters. Of those two estimates, the mark-recapture estimate of Calambokidis and Barlow (2020) has been previously used to represent U.S. West Coast abundance, as it is more precise, while the species distribution model reflects only whale densities and oceanographic conditions within the study area during summer and autumn of 2018. The best estimate of abundance for the Mainland Mexico – CA-OR-WA stock of humpback whales is considered to be the difference between the mark-recapture estimates of Calambokidis and Barlow (2020) and the Central America / Southern Mexico DIP reported by Curtis *et al.* (2022), or 3,477 animals (CV=0.101). Although the CA-OR-WA summer feeding area estimate includes some Hawai'i stock whales in Washington state, the abundance of the Hawai'i stock (11,278, CV = 0.56) is based on wintering area estimates (Becker *et al*. 2022), and more information on that stock, including prorated human-related mortality and serious injury totals from Washington state, is included in the Alaska region marine mammal stock assessments (Young *et al*. 2023).

**Minimum Population Estimate**

The minimum population estimate for this stock is taken as the lower 20th percentile of the 'difference' estimate from Curtis *et al.* (2022) cited above, or 3,185 whales.

**Current Population Trend**

Calambokidis and Barlow (2020) report that humpback whale abundance appears to have increased within the California Current at approximately 8.2% annually since 1989. This is consistent with observed increases for the entire North Pacific from ~1,200 whales in 1966 to 18,000 - 20,000 whales during 2004 to 2006 (Calambokidis *et al*. 2008). However, multiple humpback whale stocks utilize this region and a stock-specific population trend for the Mainland Mexico – CA-OR-WA stock of humpbacks has not been estimated.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Calambokidis and Barlow (2020) estimated that humpback whale abundance increased approximately 8.2% annually in the California Current since 1989, based on mark-recapture estimates largely restricted to whales summering in California and Oregon waters. However, these estimates included whales from at least two stocks; the Central America / Southern Mexico - CA/OR/WA stock and the Mainland Mexico - CA/OR/WA stock. The current net productivity rate for the Mainland Mexico - CA/OR/WA stock is unknown. However, the theoretical maximum net productivity rate can be taken to be at least as high as the maximum observed for the combined stocks, or 8.2% annually (Calambokidis and Barlow 2020), though it could be higher if one of the stocks is growing faster than another.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (3,185) times one half the estimated population growth rate for this stock of humpback whales (½ of 8.2%) times a recovery factor of 0.5, for a threatened stock with increasing population trend (NMFS 2023), resulting in a PBR of 65. Ryan *et al.* (2019) summarizes sighting and acoustic data, noting that humpbacks are present in central California waters at least 8 months annually, with December and April representing 'transition months', where whales are moving out of / into the region. Counting December and April each as one-half month of residency time during migration, plus the 7 months of May through November when whales are abundant, yields 8 months of residency time, or ⅔ of the year. This may be considered a minimum residency time, as some whales are still in U.S. waters from December to April. Therefore, the total PBR for this stock (65) is prorated by ⅔, to yield a PBR in U.S. waters of 43 whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Human-caused mortality and serious injury (MSI) of humpback whales in CA-OR-WA summer feeding areas includes whales from three stocks: **Central America / Southern Mexico – CA-OR-WA**; **Mainland Mexico – CA-OR-WA**; and **Hawai'i**. Where multiple stocks share a summer feeding area, total human-caused MSI for the mixed-stock area may be prorated to each stock using point estimates of summer to winter area movement probabilities in Wade (2021) (Table 2). Human-caused MSI from CA-OR-WA waters for the Hawai'i stock is reviewed in the Alaska stock assessments (Young *et al*. 2023).

**Table 2**. Summer to winter area movement probabilities from Wade (2021) used for prorating human-caused MSI to stocks of humpback whales using CA/OR/WA waters in summer.

| | **Location of MSI** | |
|---|---|---|
| **Stock** | **California or Oregon** | **Washington** |
| Central America/Southern Mexico - CA/OR/WA | 0.423 (CV=0.23) | 0.059 (CV=0.935) |
| Hawai'i | 0.00 | 0.688 (CV=0.13) |
| Mainland Mexico – CA/OR/WA | 0.577 (CV=0.169) | 0.254 (CV=0.278) |

**Fishery Information**

U.S. Commercial Fisheries

**Table 3.** Sources of humpback whale MSI in California (CA), Oregon (OR), and Washington (WA) commercial fisheries from 2016-2020, unless noted otherwise (Carretta 2022, Carretta *et al.* 2022, Jannot *et al.* 2021). Records include entanglements detected outside of U.S. waters, but confirmed to involve U.S. fisheries. Most cases represent strandings and at-sea sightings of entangled whales. Cases of entangled *unidentified whales* are prorated to humpback whale based on location, depth, and time of year (Carretta 2018). Sources derived from observer programs with statistical estimates of bycatch and uncertainty are shown with coefficients of variation (CV) where available. Totals for the Mainland Mexico – CA-OR-WA stock are based on prorating CA-OR-WA cases by summer to winter area movement probabilities in Table 2. In 2017, there was one non-serious injury of a humpback involving two gear types: CA Coonstripe Shrimp Pot and WA/OR/CA Sablefish Pot.

| Fishery | Cases | All CA-OR-WA Humpback Stocks ∑ MSI | Mainland Mexico – CA-OR-WA Stock ∑ MSI | Mainland Mexico – CA-OR-WA Stock Mean Annual MSI |
|---|---|---|---|---|
| CA Spot Prawn Trap | 5 | 3.25 | 1.88 | 0.375 |
| CA Dungeness Crab Pot | 34 | 23.75 | 13.7 | 2.74 |
| Dungeness Crab Pot (Commercial, state unknown) | 2 | 2 | 1.15 | 0.231 |
| OR Dungeness Crab Pot | 2 | 1.75 | 1.01 | 0.202 |
| WA Coastal Dungeness Crab Pot | 7 | 5.5 | 1.4 | 0.280 |
| Gillnet Fishery | 6 | 2 | 1.15 | 0.231 |
| Unidentified Fishery Interaction (whales identified as humpback) | 58 | 43.75 | 24.4 | 4.89 |
| Unidentified Fishery Interaction (unidentified whales prorated to humpback) | 7 | 5.25 | 3.03 | 0.606 |
| Unidentified Pot/Trap Fishery Entanglement | 13 | 9.5 | 4.67 | 0.935 |
| WA/OR/CA Sablefish Pot[1] | 2 | 7.82 (CV>0.8) | 4.51 (CV>0.8) | 0.902 (CV>0.8) |

[1] Estimates are based on 2015-2019 data (Jannot et al. 2021) for the limited entry (LE) and open-access (OA) sablefish pot sectors combined. Two observer program entanglements since 2002 informed the bycatch estimates, both of which occurred in CA + OR waters. Other sablefish pot cases opportunistically reported (at-sea sightings of entangled whales, strandings) also occurred in CA/OR waters (Carretta et al. 2022). Estimates from Jannot et al. (2021) are used in this stock assessment report because annual MSI totals are higher than those reported based on opportunistic sightings (Carretta et al. 2022). Annual observer coverage varies between 14% and 72% for the LE fleet and between 2% and 12% for the OA fleet (Somers et al., 2020).

| | | | | |
|---|---|---|---|---|
| CA Swordfish and Thresher Shark Drift Gillnet (Observer Program)[2] | 0 | 0.1 (CV>0.8) | 0.042 (CV>0.8) | ≈0.01 (CV>0.8) |
| **Totals** | **136** | **104.7** | **56.98** | **11.4** |

**Other human-caused mortality and serious injury**

Non-commercial sources of human-caused MSI, including tribal fisheries, recreational fisheries, marine debris (including research buoys) and vessel strikes are also responsible for a fraction of reported cases annually (Carretta *et al.* 2022). These sources and case totals are summarized in Tables 4 and 5.

Marine Debris, Recreational and Tribal Fisheries

**Table 4.** Sources of MSI from marine debris, recreational, and tribal fisheries from 2016-2020 summarized in Carretta *et al*. (2022).

| Source | Cases | All CA-OR-WA Humpback Stocks ∑ MSI | Mainland Mexico – CA-OR-WA Stock ∑ MSI | Mainland Mexico – CA-OR-WA Stock Mean Annual MSI |
|---|---|---|---|---|
| Dungeness Crab Pot Fishery (Recreational) | 2 | 1 | 0.577 | 0.1154 |
| Gillnet Fishery, Tribal | 3 | 2.5 | 0.635 | 0.127 |
| Hook And Line Fishery | 1 | 0.75 | 0.43275 | 0.08655 |
| Marine Debris | 1 | 1 | 0.577 | 0.1154 |
| Pot Fishery, Tribal | 1 | 1 | 0.577 | 0.1154 |
| Spot Prawn Trap/Pot Fishery (Recreational) | 1 | 0 | 0 | 0 |
| **Totals** | **9** | **6.25** | **2.80** | **0.56** |

**Vessel Strikes**

Fourteen vessel strike cases involving humpback whales were observed in CA-OR-WA waters during 2016-2020 (8 in CA, 1 in OR, and 5 in WA), totaling 13.2 MSI, or 2.6 whales per year (Carretta *et al.* 2022). Most vessel strikes are likely undetected and thus, we use estimates of vessel strike mortality reported by Rockwood *et al.* (2017) for this region. The estimated number of annual vessel strike deaths was 22 humpback whales, though this includes only the period July – November when whales are most likely to be present in the U.S. West Coast EEZ and the season that overlaps with survey effort used in species distribution models (Becker *et al.* 2016, Rockwood *et al.* 2017). This estimate is based on an assumption of a moderate level of vessel avoidance by humpback whales, as measured by the behavior of satellite-tagged whales in the presence of vessels (McKenna *et al.* 2015). Based on estimates of 22 deaths due to vessel strikes annually, the number attributed to the Mainland Mexico - CA-OR-WA stock during 2016-2020 is 10.15 whales per year (Table 5). The estimated mortality of 10.15 humpback whales annually due to vessel strikes represents approximately 0.3% of the stock's estimated population size (10.15 deaths / 3,477 whales). The ratio of mean annual observed to estimated vessel strike deaths and serious injuries of humpback whales during 2016-2020 is 2.6 / 22 = 0.11, implying that vessel strike counts from opportunistic observations represent a small fraction of overall incidents.

**Table 5.** Summary of humpback whale vessel strike MSI during 2016-2020 (Carretta *et al*. 2022). Estimates are based on prorating annual estimates of humpback vessel strike mortality in this region (22/yr, Rockwood *et al*. 2017) by the fraction of observed vessel strikes in different feeding areas (WA vs CA/OR), which are then prorated to stock by summer to winter area movement probabilities from Wade (2021).

[2] There were no observed entanglements during 2016-2020 with 21% observer coverage, however the model-based estimate of bycatch is based on pooling 1990-2000 data, resulting in a small positive estimate (Carretta 2022).

| State Detected | Observations | Fraction of Observations | Fraction of Observations *times* 22 MSI/yr estimated by Rockwood et al. (2017) | Mainland Mexico – CA-OR-WA stock prorated ∑ MSI based on summer to winter area movement probabilities (Wade 2021) |
|---|---|---|---|---|
| WA | 5 | 0.357 | 7.86 | 1.99 |
| CA/OR | 9 | 0.643 | 14.14 | 8.16 |
| **Total** | **14** | | | **10.15** |

Vessel strikes in U.S. West Coast EEZ waters continue to impact large whales (Redfern *et al.* 2013; 2019; Moore *et al.* 2018). A complex of diverse vessel types, speeds, and destination ports all contribute to variability in vessel traffic and these factors may be influenced by economic and regulatory changes. For example, Moore *et al.* (2018) found that primary routes travelled by vessels changed when emission control areas (ECAs) were established off the U.S. West Coast. They also found that large vessels typically reduced their speed by 3-6 kts in ECAs between 2008 and 2015. The speed reductions are thought to be a strategy to reduce operating costs associated with more expensive, cleaner burning fuels required within the ECAs. In contrast, Moore *et al.* (2018) noted that some vessels increased speed when transiting longer routes to avoid the ECAs. Further research is ongoing to understand how variability in vessel traffic affects vessel strike risk and mitigation strategies, though Redfern *et al.* (2019) note that a combination of vessel speed reductions and expansion of areas to be avoided should be considered. Rockwood *et al.* (2017) note that 82% of humpback whale vessel strike mortalities occur within 10% of the region, implying that vessel strike mitigation measures may be effective if applied over relatively small regions.

**Historic whaling**

Approximately 15,000 humpback whales were taken from the North Pacific from 1919 to 1987 (Tonnessen and Johnsen 1982), and, of these, approximately 8,000 were taken from the west coast of Baja California, California, Oregon and Washington (Rice 1978). Shore-based whaling apparently depleted the humpback whale stock off California twice: once prior to 1925 (Clapham *et al.* 1997) and again between 1956 and 1965 (Rice 1974). There has been a prohibition on taking humpback whales since 1966.

**Habitat Concerns**

Increasing levels of anthropogenic sound in the world's oceans (Andrew *et al.* 2002), such as those produced by shipping traffic, or Low Frequency Active sonar, is a habitat concern for whales, as it can reduce acoustic space used for communication (masking) (Clark *et al.* 2009, NOAA 2016c). This can be particularly problematic for baleen whales that may communicate using low-frequency sound (Erbe 2016). Based on vocalizations (Richardson *et al.* 1995; Au *et al.* 2006), reactions to sound sources (Lien *et al.* 1990, 1992; Maybaum 1993), and anatomical studies (Hauser *et al.* 2001), humpback whales also appear to be sensitive to mid-frequency sounds, including those used in active sonar military exercises (U.S. Navy 2007).

Seven important feeding areas for humpback whales are identified off the U.S. west coast by Calambokidis *et al.* (2015), including five in California, one in Oregon, and one in Washington. Humpback whales have increasingly reoccupied areas inside of Puget Sound (the 'Salish Sea'), a region where they were historically abundant prior to whaling (Calambokidis *et al.* 2017).

**STATUS OF STOCK**

The Mainland Mexico - CA-OR-WA stock of humpback whales is a DIP delineated from the 'Mexico DPS' of humpback whales listed as threatened under the ESA (Bettridge *et al.* 2015, Martien *et al.* 2021), and is therefore considered 'depleted' and 'strategic' under the MMPA. Total annual human-caused serious injury and mortality of humpback whales is the sum of commercial fishery (11.4/yr) + estimated vessel strikes (10.15/yr), + non-commercial sources (0.56/yr), or 22 humpback whales annually. Total commercial fishery mortality and serious injury (11.4/yr) is > 10% of the calculated PBR (43) for this stock, thus takes are not approaching zero mortality and injury rate. There is no estimate of the undocumented fraction of anthropogenic injuries and deaths to humpback whales on the U.S. West Coast, but for vessel strikes, a comparison of observed vs. estimated annual vessel strikes suggests that approximately 10% of vessel strikes are documented. There is no direct estimate of population trend for this stock, but Calambokidis and Barlow (2020) report that humpback whale abundance increased within the California Current at approximately 8.2% annually since 1989, which includes animals from three stocks: Central America / Southern Mexico – CA-OR-WA, Mainland Mexico – CA-OR-WA, and Hawai'i. Habitat concerns include sensitivity to anthropogenic sound sources.

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Revised 05/14/2024

# BLUE WHALE (*Balaenoptera musculus musculus*): Eastern North Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

North Pacific blue whales were once thought to belong to as many as five separate populations (Reeves *et al.* 1998), but acoustic evidence suggests only two populations, in the eastern and western North Pacific, respectively (Stafford *et al*. 2001, Stafford 2003, McDonald *et al*. 2006, Monnahan *et al*. 2014). North Pacific blue whales produce two distinct acoustic calls, referred to as "northwestern" and "northeastern" types. Stafford *et al.* 2001, Stafford 2003, and Monnahan *et al.* 2014 have proposed that these represent distinct populations with some geographic overlap. The northeastern call predominates in the Gulf of Alaska, along the U.S. West Coast, and in the eastern tropical Pacific, and the northwestern call predominates from south of the Aleutian Islands to Russia's Kamchatka Peninsula, though both call types have been recorded concurrently in the Gulf of Alaska (Stafford *et al*. 2001, Stafford 2003). Both call types occur in lower latitudes in the central North Pacific, but differ in seasonal patterns (Stafford *et al*. 2001). Blue whales satellite-tagged off California in summer have traveled to the eastern tropical Pacific and the Costa Rica Dome in winter (Mate *et al*. 1999, Bailey *et al.* 2009). Blue whales photographed off California have been matched to individuals photographed off the Queen Charlotte Islands in northern British Columbia and to one individual photographed in the northern Gulf of Alaska (Calambokidis *et al*. 2009a). Barlow (2010, 2016) noted a northward shift in blue whale distribution within the California Current, based on a series of vessel-based line-transect surveys between 1991 and 2014. Gilpatrick and Perryman (2008) reported that blue whales from California to Central America (the Eastern North Pacific stock) are on average, two meters shorter than blue whales measured from historic whaling records in the central and western North Pacific.

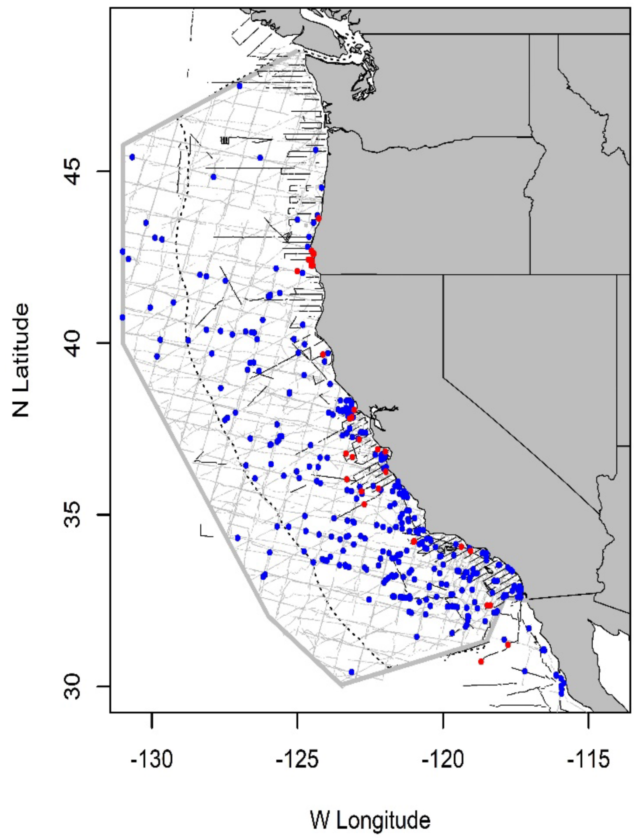


**Figure 1**. Blue whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, two stocks are currently recognized in the North Pacific: 1) the Eastern North Pacific Stock, and 2) the Central North Pacific Stock. Based on northeastern call type locations, some whales in the Eastern North Pacific stock may range as far west as Wake Island and as far south as the Equator (Stafford *et al*. 1999, 2001). The U.S. West Coast is an important feeding area in summer and fall (Fig. 1), but, increasingly, blue whales from the Eastern North Pacific stock are found feeding north and south of this area in summer and fall. Nine important areas for blue whale feeding have been identified off the California coast (Calambokidis *et al*. 2015), including six areas in southern California and three in central California.

Most of this stock is believed to migrate south to spend the winter and spring in high productivity areas off Baja California, the Gulf of California, and on the Costa Rica Dome.

## POPULATION SIZE

The size of the feeding stock of blue whales off the U.S. West Coast has been estimated by line-transect and mark-recapture methods. Because some fraction of the population is always outside the survey area, the line-transect and mark recapture estimation methods provide different measures of abundance for this stock. Line transect estimates reflect the average density and abundance of blue whales in the study area during summer and autumn surveys, while mark-recapture estimates can provide an estimate of total population size if differences in capture heterogeneity are addressed.

Abundance estimates from line-transect surveys have been highly-variable (Fig. 2), and this variability is attributed to northward distributional shifts of blue whales out of U.S. waters linked to warming ocean temperatures (Barlow and Forney 2007, Calambokidis *et al.* 2009a, Barlow 2010, 2016). Mark-recapture estimates of abundance are considered the more reliable and precise of the two methods for this transboundary population of blue whales because not all animals are within the U.S. Exclusive Economic Zone (EEZ) during summer and autumn line-transect surveys and mark-recapture estimates can be corrected for heterogeneity in sighting probabilities. Generally, the highest abundance estimates from line-transect surveys occurred in the mid-1990s, when ocean conditions were colder than present-day (Fig. 2). Since that time, line-transect abundance estimates within the California Current have declined, while estimates from mark-recapture studies have increased (Fig. 2). Evidence for a northward shift in blue whale distribution includes increasing numbers of blue whales found in Oregon and Washington waters during 1996-2014 line-transect surveys (Barlow 2016) and satellite tracks of blue whales in Gulf of Alaska and Canadian waters between 1994 and 2007 (Bailey *et al.* 2009). Calambokidis and Barlow (2020) estimated blue whale abundance for the U.S. West Coast based on updated photographic ID data through 2018 using mark-recapture methods. They reported that the best estimate of current abundance for CA/OR/WA waters is based the most-recent 4 years (2015-2018) of capture-recapture data and a Chao model that accounts for heterogeneity of capture probabilities, resulting in an estimate of 1,898 (CV=0.085) whales. Becker *et al*. (2020) also estimated blue whale abundance with habitat-based species distribution models from line-transect data collected between 1991 to 2018, using fixed and dynamic ocean variables (Becker *et al.* 2016, 2017). The most-recent species distribution model-based estimate is 670 (CV=0.43) blue whales for 2018 (Fig. 2). The mark-recapture estimate (1,898) is considered the best estimate of abundance for 2018 due to its higher precision and because estimates based on line-transect data reflect only animal densities within the study area at the time surveys are conducted.

### Minimum Population Estimate

The minimum population estimate of blue whales is calculated as the lower 20th percentile of the 2018 mark-recapture estimate, or 1,767 whales.

### Current Population Trend

Mark-recapture estimates provide the best gauge of population trends for this stock, because of recent northward shifts in blue whale distribution that negatively bias line-transect estimates. Based on mark-recapture estimates shown in Fig. 2, there may be evidence of a population size increase since the 1990s, but a formal trend analysis is lacking and the current population trend is unknown. Monnahan *et al*. (2015) used a population dynamics model to estimate that the eastern Pacific blue whale population was at 97% of carrying capacity in 2013 and suggested that density dependence, and not vessel strike impacts, explained the observed lack of a population size increase since the early 1990s. Monnahan *et al.* (2015) also estimated that the eastern North Pacific population likely did not drop below 460 whales during the last century, despite being targeted by commercial whaling. Monnahan *et al*. (2014) estimated that 3,411 blue whales (95% range 2,593 - 4,114) were removed via commercial whaling from the eastern North Pacific between 1905 and 1971.

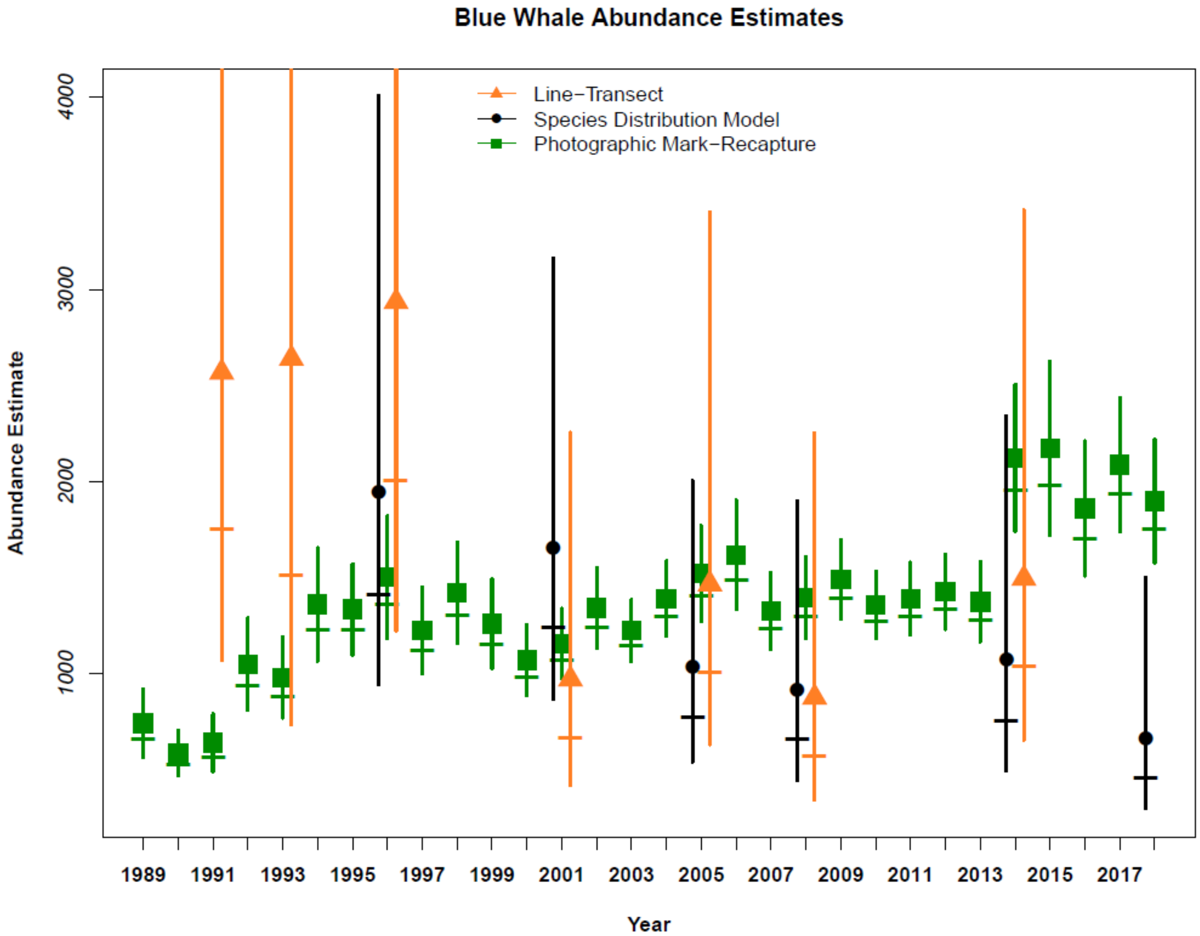


**Figure 2**. Estimated abundance of blue whales based on three methods (standard vessel-based line transect surveys, habitat-based species distribution models, and a photographic mark-recapture model). The line-transect estimates are based on surveys reported by Barlow (2016). Species distribution model estimates are based on the same line-transect surveys, but use fixed and dynamic ocean variables to model whale density (Becker *et al.* 2020). The mark recapture estimates reflects a Chao model that uses rolling 4-year periods and accounts for heterogeneity of capture probability (Calambokidis and Barlow 2020). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates, 95% confidence limits reported from species distribution model estimates, and ±2 standard errors of mark-recapture abundance estimates. Horizontal hatch marks represent minimum population size estimates based on 20$^{th}$ percentiles of mean estimates. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington. The y-axis has been truncated to better show the variability in mean estimates between methods. Upper 95% confidence limits for line-transect surveys in 1991, 1993 and 1995 not visible in this plot ranged between 6,000 and 9,500 whales.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Based on mark-recapture estimates from the U.S. West Coast and Baja California, Mexico, Calambokidis *et al*. (2009b) estimated an approximate rate of increase of 3% per year. This estimate is not considered a maximum net productivity rate because it does not account for the effects of anthropogenic mortality and serious injury on the population and therefore likely represents an underestimate of the maximum net productivity rate. For this reason and because an estimate of maximum net productivity is lacking for any blue whale population, the default rate of 4% is used for all blue whale stocks, based on NMFS guidelines for preparing stock assessments (NMFS 2016).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (1,767) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.2 (for an endangered species with a minimum abundance greater than 1,500 and unknown population trend), resulting in a PBR of 7 whales. Satellite telemetry deployments (Hazen *et al*. 2016) indicate that most blue whales are outside U.S. West Coast waters from November to March (5 months), so the PBR for U.S. waters is 7/12 of the total PBR, or 4.1 whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY
### Fisheries Information

Blue whales are occasionally documented entangled in pot/trap fisheries and other unidentified fishery gear on the U.S. West Coast (Table 1). The annual entanglement rate of blue whales (observed) during 2017-2021 is the sum of observed annual entanglements (0.60/yr), plus species probability assignments (Carretta 2018) from 4 unidentified whale entanglements (0.014/yr), totaling 0.61 blue whales annually (Table 1). Observed totals represent a negatively-biased accounting of the serious injury and mortality of blue whales in the region, because not all cases are detected and there is no correction factor available to account for undetected events.

**Table 1.** Summary of available information on observed incidental mortality and injury of blue whales (Eastern North Pacific stock) from commercial fisheries (Carretta *et al.* 2023, Carretta 2022). Values in this table represent observed deaths and serious injuries and totals are negatively-biased because not all cases are detected.

| Fishery Name | Year(s) | Data Type | Observer Coverage | Observed Mortality + Serious Injury | Estimated mortality and/or serious injury (CV in parentheses) | Mean Annual Mortality and Serious Injury (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA Dungeness crab pot** | 2017-2021 | Strandings + sightings | n/a | 0 + 0.75 | n/a | ≥ 0.15 (n/a) |
| **Unidentified fishery interactions involving identified blue whales** | 2017-2021 | Strandings + sightings | n/a | 0 + 2.25 | n/a | ≥ 0.45 (n/a) |
| **Unidentified fishery interactions involving *unidentified whales* prorated to blue whale** | 2017-2021 | Strandings + Sightings | n/a | n/a | 0.07 | ≥ 0.014 |
| **CA/OR thresher shark/swordfish drift gillnet fishery** | 2017<br>2018<br>2019<br>2020<br>2021 | observer | 0.186<br>0.251<br>0.226<br>0.222<br>0.228 | 0 | 0 | 0 (n/a) |
| **Total Annual Takes** | | | | | | ≥ 0.61(n/a) |

### Vessel Strikes

Three blue whale vessel strike deaths were observed during 2017-2021 (Carretta *et al*. 2023), resulting in an observed annual average of 0.6 vessel strike deaths. Observations of blue whale vessel strikes have been highly-variable in previous 5-year periods, with as many as 10 observed (9 deaths + 1 serious injury) during 2007-2011 (Carretta *et al*. 2013). The highest number of blue whale vessel strikes observed in a single year (2007) was 5 whales (Carretta *et al*. 2013). Since 2007, documented vessel strikes have totaled 14 blue whales and 10 unidentified whales (Carretta *et al*. 2013, 2023). Methods to prorate the number of unidentified whale vessel strike cases to species are not available, because observed sample sizes are small and identified cases are likely biased towards species that are large, easy to identify, and more likely to be detected, such as blue and fin whales. Most observed blue whale vessel strikes have been in southern California or off San Francisco, CA, where blue whales seasonally occur close to shipping ports (Berman-Kowalewski *et al*. 2010). Documented vessel strike deaths and serious injuries are derived from observed whale carcasses and at-sea sightings and are considered minimum values. Where evaluated, estimates of detection rates of cetacean carcasses are consistently quite low across different regions and species (<1% to 36%), highlighting that observed numbers are unrepresentative of true impacts (Kraus *et al*. 2005, Pace *et al*. 2021, Perrin *et al*. 2011, Williams *et al*. 2011, Prado *et al*. 2013). Due to this negative bias, Redfern *et al*. (2013) noted that the number of observed vessel strike deaths of blue whales in the U.S. West Coast EEZ likely exceeds PBR.

Vessel strike mortality was estimated for blue whales in the U.S. West Coast EEZ (Rockwood *et al*. 2017), using an encounter theory model (Martin *et al.* 2016) that combined species distribution models of whale density (Becker *et al*. 2016), vessel traffic characteristics (size + speed + spatial use), along with whale movement patterns obtained from satellite-tagged whales in the region to estimate encounters that would result in mortality. The estimated number of annual vessel strike deaths was 18 blue whales, though this includes only the period July – November when whales are most likely to be present in the U.S. West Coast EEZ and was based on cetacean habitat models generated from line-transect surveys (Becker *et al*. 2016, Rockwood *et al*. 2017). This estimate was also based on an assumption of a moderate level of vessel avoidance (55%) by blue whales, as measured by the behavior of satellite-tagged whales in the presence of vessels (McKenna *et al*. 2015). The estimated mortality of 18 blue whales annually due to vessel strikes represents approximately 1% of the most recent estimated population size of the stock (18 deaths / 1,898 whales). The results of Rockwood *et al*. (2017) also include a no-avoidance encounter model that results in a worst-case estimate of 40 annual blue whale vessel strike deaths, which represents 2.1% of the estimated population size. The authors note that 74% of blue whale vessel strike mortalities occur within 10% of the study area, implying that vessel avoidance mitigation measures can be effective if applied over relatively small regions. Using the moderate level of avoidance model from Rockwood *et al*. (2017), estimated vessel strike deaths of blue whales are 18 annually. A comparison of average annual vessel strikes observed over the period 2017-2021 (0.6/yr) versus estimated vessel strikes (18/yr) indicates that the rate of detection for blue whale vessel strikes is approximately 3%. Comparing the highest number of vessel strikes observed in a single year (5 in 2007) with the estimated annual number (18) implies that vessel strike detection rates have not exceeded 28% (5/18) in any single year.

Impacts of vessel strikes on population recovery of the eastern North Pacific blue whale population were assessed by Monnahan *et al*. (2015). Their population dynamics model incorporated data on historic whaling removals, vessel strike levels, and projected numbers of vessels using the region through 2050. The authors concluded (based on 10 vessel strike deaths per year) that this stock was at 97% of carrying capacity in 2013. These authors also analyzed the status of the blue whale stock based on a 'high case' of annual vessel strike deaths (35/yr) and concluded that under that scenario, the stock would have been at approximately 91% of carrying capacity in 2013. Caveats to the carrying capacity analysis include the assumption that the population was already at carrying capacity prior to commercial whaling of this stock in the early 20$^{th}$ century and that carrying capacity has not changed appreciably since that time (Monnahan *et al*. 2015).

Vessel strikes within the U.S. West Coast EEZ impact all large whale populations (Redfern *et al*. 2013; 2019; Moore *et al*. 2018). (Redfern *et al*. 2013; 2019; Moore *et al*. 2018). However, diverse vessel types, speeds, and destination ports all contribute to variability in vessel traffic and these factors may be influenced by economic and regulatory changes. For example, Moore *et al*. (2018) found that primary routes travelled by vessels changed when emission control areas (ECAs) were established off the U.S. West Coast. They also found that large vessels typically reduced their speed by 3-6 kts in ECAs between 2008 and 2015. The speed reductions are thought to be a strategy to reduce operating costs associated with more expensive, cleaner burning fuels required within the ECAs. In contrast, Moore *et al*. (2018) noted that some vessels increased their speed when they transited longer routes to avoid the ECAs. Further research is ongoing to understand how variability in vessel traffic affects vessel strike risk and mitigation strategies, though Redfern *et al*. (2019) note that a combination of vessel speed reductions and expansion of areas to be avoided should be considered.

**STATUS OF STOCK**

As a result of commercial whaling, blue whales were listed as "endangered" under the U.S. Endangered Species Conservation Act of 1969. This protection was transferred to the U.S. Endangered Species Act in 1973. Despite an analysis suggesting that the Eastern North Pacific population was at 91%-97% of carrying capacity in 2013 (Monnahan *et al*. 2015), blue whales are listed as "endangered", and consequently the Eastern North Pacific stock is automatically considered a "depleted" and "strategic" stock under the MMPA. Conclusions about the population's current status relative to carrying capacity depend upon assumptions that the population was already at carrying capacity before commercial whaling impacted the population in the early 1900s, and that carrying capacity has remained relatively constant since that time (Monnahan *et al*. 2015). If carrying capacity has changed significantly in the last century, conclusions regarding the status of this population would necessarily change (Monnahan *et al*. 2015).

The sum of observed and assigned annual incidental mortality and serious injury due to commercial fisheries (≥0.61/yr), plus estimated vessel strike deaths (18/yr), is 18.6 whales annually for 2017-2021. This exceeds the calculated PBR of 4.1 for this stock. Monnahan *et al*. (2015) proposed that estimated vessel strike levels of 10 – 35 whales annually did not pose a threat to the status of this stock, but estimates of carrying capacity of this blue whale stock differed depending on the level of vessel strikes: 97% of K with 10 annual strikes and 91% of K with 35 annual strikes. The highest estimates of blue whale vessel strike mortality (35/yr; Monnahan *et al*. (2015); 40/yr; Rockwood *et al*. (2017)) are similar, and annually represent approximately 2% of the estimated population size. Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ 0.61) exceed 10% of the stock's PBR (4.1), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate.

**OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY**

Increasing levels of anthropogenic sound in the world's oceans is a habitat concern for blue whales (Reeves *et al*. 1998, Andrew *et al*. 2002). Tagged blue whales exposed to simulated mid-frequency sonar and pseudo-random noise demonstrated a variety of behavioral responses, including no change in behavior, termination of deep dives, directed travel away from sound sources, and cessation of feeding (Goldbogen *et al*. 2013, Southall *et al.* 2019). Behavioral responses were highly dependent upon the type of sound source, distance from sound sources, and the behavioral state of the animal at the time of exposure. Deep-feeding and non-feeding whales reacted more strongly to experimental sound sources than surface-feeding whales that typically showed no change in behavior (Goldbogen *et al.* 2013, Southall *et al.* 2019). Both studies noted that behavioral responses to such sounds are influenced by a complex interaction of behavioral state, environmental context, and prior exposure of individuals to such sound sources. One concern expressed in both studies is if blue whales did not habituate to such sounds near feeding areas, that chronic cessation of feeding behavior could affect the fitness of individual whales, which could impact population fitness (Goldbogen *et al.* 2013, Southall *et al.* 2019). Currently, no evidence indicates that such reduced population health exists, but such evidence would be difficult to differentiate from natural sources of reduced fitness or mortality in the population. Nine blue whale feeding areas identified off the California coast by Calambokidis *et al*. (2015) represent a diversity of nearshore and offshore habitats that overlap with a variety of anthropogenic activities, including shipping, oil and gas extraction, and military activities.

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Revised 05/14/2024

# FIN WHALE (*Balaenoptera physalus velifera*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Fin whales are found from temperate to subpolar oceans worldwide, with a distributional hiatus between the Northern and Southern Hemispheres within 20° to 30° of the equator (Edwards *et al.* 2015). Fin whales occur throughout the North Pacific, from the northeastern Chukchi Sea (Crance *et al.* 2015) to the Tropic of Cancer (Mizroch *et al.* 2009), but their wintering areas are poorly known. Archer *et al*. (2019a) used mitochondrial DNA and single-nucleotide polymorphisms (SNPs) to demonstrate that North Atlantic and North Pacific genetic samples could be correctly assigned to their respective ocean basins with 99% accuracy. North Pacific whales are recognized as a separate subspecies: *Balaenoptera physalus velifera*. Mizroch *et al.* (2009) described eastern and western North Pacific populations, based on sightings data, catch statistics, recaptures of marked whales, blood chemistry, and acoustics. The two populations are thought to have separate wintering and mating grounds off Asia and North America and during summer, whales from each population may co-occur near the Aleutian Islands and Bering Sea (Mizroch *et al.* 2009). A non-migratory population occurs in the Gulf of California, based on evidence from photo-ID, genetics, satellite telemetry, and acoustics (Thompson *et al*. 1992; Tershy *et al.* 1993; Bérubé *et al.* 2002; Jiménez López *et al*. 2019; Nigenda-Morales 2008; Širović *et al*. 2017, Nigenda-Morales *et al.* 2023). Fin whales are scarce in the eastern tropical Pacific in summer and winter (Lee 1993, Wade and Gerrodette 1993). Fin whales occur year-round in the Gulf of Alaska (Stafford *et al.* 2007); the Gulf of California (Tershy *et al.* 1993; Bérubé *et al.* 2002); California (Dohl *et al.* 1983; Širović *et al*. 2017); and Oregon and Washington (Moore *et al.* 1998). Fin whales satellite-tagged in the Southern California Bight (SCB) use the region year-round, although they seasonally range to central California and Baja California before returning to the SCB (Falcone and Schorr 2013). The longest satellite track reported by Falcone and Schorr (2013) was a fin whale tagged in the SCB in January 2014 that moved south to central Baja California by February and north to the Monterey area by late June. Archer *et al.* (2013) present evidence for geographic separation of fin whale mtDNA clades near Point Conception, California. A significantly higher proportion of 'clade A' is composed of samples from the SCB and Baja California, while 'clade C' is largely represented by samples from central California, Oregon, Washington, and the Gulf of Alaska.

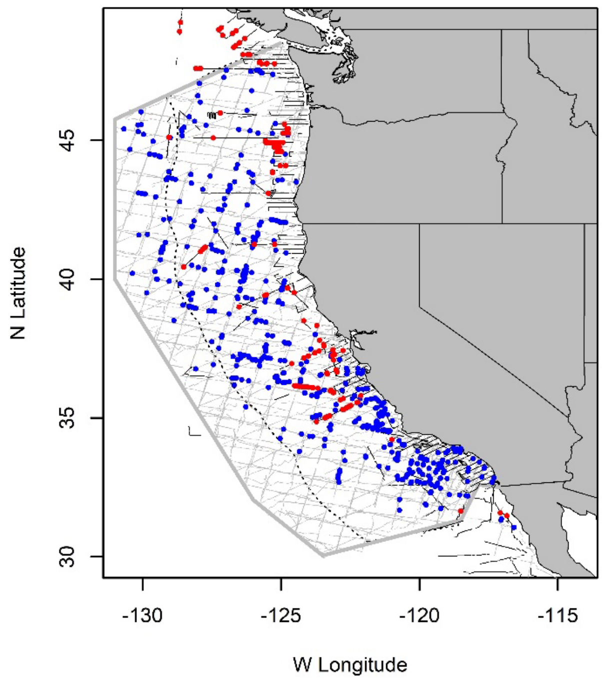


**Figure 1.** Fin whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018 (Barlow 2016, Henry *et al*. 2020). Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

While knowledge of North Pacific fin whale population structure from genetic and movement patterns is limited, passive acoustic data provides another line of evidence to assess population structure. For example, acoustic data (Širović *et al*., 2017; Thompson *et al*., 1992) support prior photo-ID (Tershy *et al.* 1993) and genetic conclusions (Bérubé *et al.* 2002; Nigenda-Morales *et al.* 2008; Rivera-León *et al.* 2019) that a resident fin whale population occurs in the Gulf of California, Mexico. Additionally, acoustic data indicate there may be a resident population in southern California waters, though this may be confounded by seasonal movements in the region (Širović *et al*., 2015, 2017). Oleson *et al*. (2014) report that fin whale songs recorded near Hawaii are similar to those from southern California and the Bering Sea, suggesting movement of animals throughout that range. Song structure throughout the North Pacific is characterized by seasonal and interannual variability (Delarue *et al*., 2013; Oleson *et al*., 2014; Širovic *et al*., 2017; Weirathmueller *et al*., 2017). Similarities of songs within and across years for multiple North Pacific pelagic areas (Hawaii, Bering Sea, Southern California) suggests that a single population may range throughout this oceanic basin; however, there is evidence for multiple song types in the Bering Sea (Delarue *et al*., 2013) and the northeast Pacific, including a possible resident population in inland waters of British Columbia (Koot, 2015). Archer *et al*. (2019b) developed an automated classification method for fin whale note types that revealed analysts have manually misclassified certain fin whale note types near Hawaii, which has implications for stock identification interpretation. These authors found that Hawaii had some of the most distinctive calls, with sequences characterized by "B" type calls with relatively long inter-note intervals. Archer *et al*. (2019b) also notes the similarity of B sequences from the Gulf of California in spring that match those described by Širović *et al*. (2017) as a "long singlet" pattern found in the southern Gulf of California and southern California Bight. In the Archer *et al*. (2019b) study, the B singlet pattern was most similar to Monterey Bay and northwest Pacific autumn sequences, perhaps reflecting a widespread pattern across populations in the North Pacific, or hinting at some population connectivity between the central and southern U.S. West Coast and southern Gulf of California and the northwest Pacific (Archer *et al*. 2019b). Acoustic evidence also suggests two populations that use the Chuckchi Sea and central Aleutian Islands area that mix seasonally in the southern Bering Sea (Archer *et al*. 2019b). Observed movements of fin whales from the southern and central Bering Sea to the Aleutian Islands and Kamchatka documented from Discovery tag recoveries are consistent with these acoustic findings (Mizroch *et al*. 2009). Further research is necessary to use multiple lines of evidence, such as acoustics, genetics, and satellite telemetry in order to identify population stocks in the North Pacific (Martien *et al.* 2020).

Insufficient data exists to determine population structure, but from a conservation perspective it may be risky to assume panmixia in the North Pacific. This report covers the stock of fin whales found along the coasts of California, Oregon, and Washington within 300 nmi of shore (Fig. 1). Because fin whale abundance appears lower in winter/spring in California (Dohl *et al.* 1983; Forney *et al.* 1995) and in Oregon (Green *et al.* 1992), it is likely that the distribution of this stock extends seasonally outside these coastal waters. The Marine Mammal Protection Act (MMPA) stock assessment reports recognize three stocks of fin whales in the North Pacific: (1) the California/Oregon/Washington stock (this report), (2) the Hawaii stock, and (3) the Northeast Pacific stock.

## POPULATION SIZE

Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2016, 2017, 2020, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 11,065 (CV=0.405) animals (Becker *et al.* 2020). This estimate is higher than those reported from Bayesian trend analyses by Moore and Barlow (2011) and Nadeem et al. (2016), but is consistent with their conclusion of increasing abundance. The estimates of Becker *et al*. (2020) also include sea-state specific correction factors to prorate unidentified large whale sightings to species that would otherwise result in negative estimation biases (Becker *et al*. 2017).

### Minimum Population Estimate

The minimum population estimate for fin whales is taken as the lower 20th percentile of the posterior distribution of 2018 abundance estimate, or 7,970 whales (Becker *et al*. 2020b).

### Current Population Trend

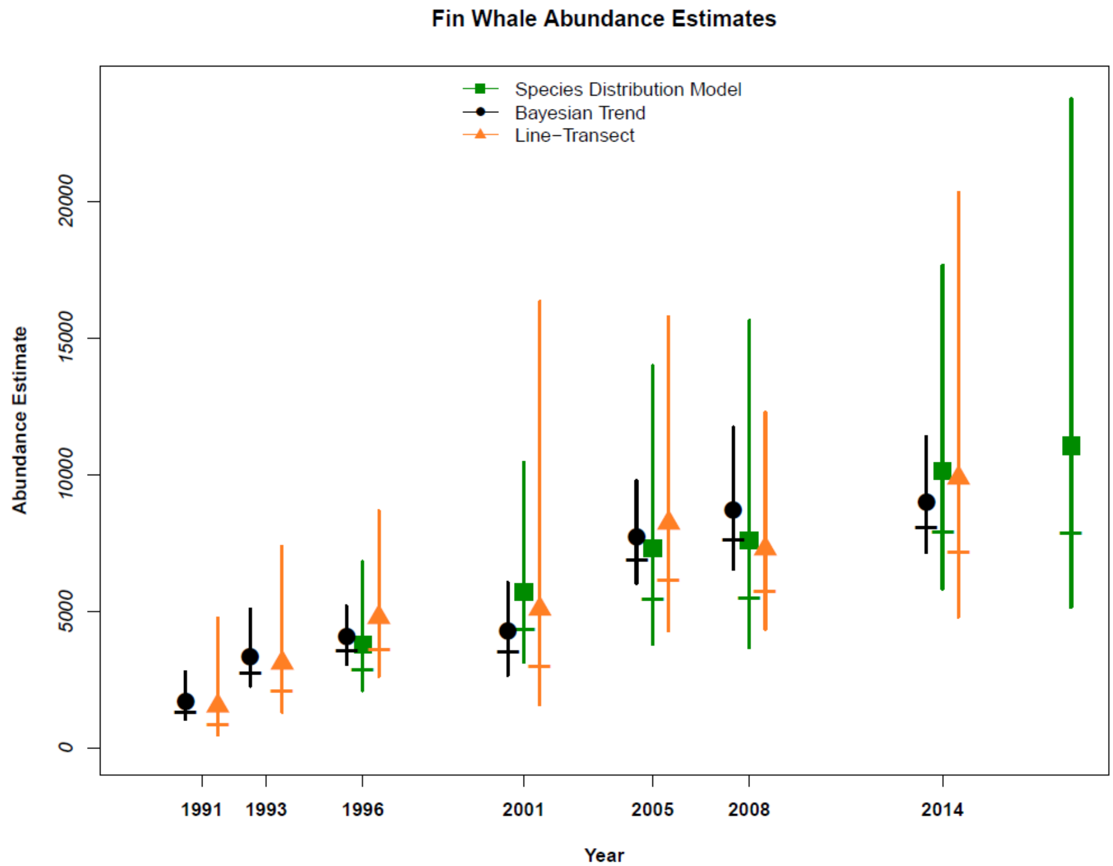


**Figure 2.** Fin whale abundance estimated from three methods (standard vessel-based line transect surveys (Barlow 2016), habitat-based species distribution models (Becker *et al*. 2020), and a Bayesian trend analysis (Nadeem *et al*. 2016). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates, 95% confidence limits reported from species distribution model estimates, and 95% prediction intervals from Nadeem *et al*. (2016). Horizontal hatch marks represent minimum population size estimates based on 20$^{th}$ percentiles of mean estimates. Line-transect surveys in 1991 and 1993 did not include the waters of Oregon and Washington.

Indications of recovery in CA coastal waters date back to 1979/80 (Barlow 1994), but there is now strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 based on analysis of line transect surveys (Moore and Barlow 2011, Nadeem *et al.* 2016, Becker *et al.* 2020a, Fig. 2). Nadeem *et al.* (2016) reported mean annual abundance increased 7.5% annually during 1991 to 2014.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

Estimated annual rates of increase in the California Current (California, Oregon, and Washington waters) averaged 7.5% from 1991 to 2014 (Nadeem *et al.* 2016). However, it is unknown how much of this growth is due to immigration rather than birth and death processes. A doubling of the abundance estimate in California waters between 1991 and 1993 cannot be explained by birth and death processes alone, and movement of individuals between U.S. west coast waters and other areas (e.g., Alaska, Mexico) have been documented (Mizroch *et al.* 1984).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (7,970) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for an endangered species, with $N_{min}$ > 5,000 and $CV_{Nmin}$ < 0.50, Taylor *et al.* 2003), resulting in a PBR of 80 whales.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fisheries Information

The California large-mesh drift gillnet fishery for swordfish and thresher shark includes one observed entanglement record (in 1999) of a fin whale from 9,246 observed fishing sets during 1990 - 2021 (Carretta 2022). The estimated bycatch of fin whales in this fishery for the most recent 5-year period is zero whales (Carretta 2022).

In addition to drift gillnets, fin whales are observed entangled in longline gear. One fin whale was observed entangled in 2015 in the Hawaii shallow-set longline fishery in waters between the U.S. West Coast and Hawaiian EEZs. The entanglement was determined to be a non-serious injury, based on the animal being cut free of the gear with superficial wounds caused by the line (Bradford 2018). The stock identity of this whale is unknown.

Two fin whale serious injuries were documented in unidentified fishing gear during 2017-2021, or 0.4 whales annually (Carretta *et al*. 2023). Additionally, there were 4 *unidentfied whale* entanglements during this period, of which, 0.05 were prorated as fin whales using the method reported by Carretta (2018). Unidentified whale entanglements typically involve whales seen at-sea with unknown gear configurations that are prorated to represent 0.75 serious injuries per entanglement case. Thus, approximately 0.05 x 0.75 = 0.04 fin whale serious injuries occurred from the 4 unidentified whale entanglement cases during 2017-2021 (Table 1). This represents a negligible annual estimate of ~ 0.01 prorated fin whales derived from sightings of unidentified entangled whales. Total mean annual fishery-related serious injury and mortality is the sum of observed (0.4) and prorated (0.01) mean annual deaths and serious injuries, or 0.41 fin whales annually (Table 1).

**Table 1**. Summary of available information on the incidental mortality and serious injury of fin whales (CA/OR/WA stock) for commercial fisheries that might take this species.

| Fishery Name | Data Type | Year(s) | Observer Coverage | Observed (or self-reported) | Estimated Mortality (and serious injury) | Mean Annual Mortality and Serious Injury (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA swordfish and thresher shark drift gillnet fishery** | 2017<br>2018<br>2019<br>2020<br>2021 | observer | 0.186<br>0.251<br>0.226<br>0.222<br>0.228 | 0 | 0 (n/a) | 0 (n/a) |
| **Unidentified fishery interactions involving *fin whales*** | 2017-2021 | at-sea sightings | n/a | 2 | 0 (2) | ≥ 0.4 |
| **Unidentified fishery interactions involving *unidentified whales* prorated to fin whale** | 2017-2021 | at-sea sightings | n/a | n/a | 0 (0.04) | ≥ 0.01 |
| **Minimum total annual takes** | | | | | | ≥ 0.41 (n/a) |

### Vessel Strikes

Vessel strikes were implicated in the deaths of 8 fin whales from 2017-2021 (Carretta *et al.* 2023). Additional mortality from vessel strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. The average observed annual mortality and serious injury due to vessel strikes is 1.6 fin whales per year during 2017-2021. Documented vessel strike deaths and serious injuries are derived from direct counts of whale carcasses and represent minimum impacts. Where evaluated, estimates of detection rates of cetacean carcasses are consistently low across different regions and species (<1% to 36%), highlighting that observed numbers underestimate true impacts (Carretta *et al.* 2016, Kraus *et al.* 2005, Williams *et al.* 2011, Prado *et al.* 2013, Wells *et al.* 2015, Pace *et al*. 2021). Vessel strike mortality was recently estimated for fin whales in the U.S. West Coast EEZ (Rockwood *et al.* 2017), using an encounter theory model (Martin *et al.* 2016) that combined species distribution models of whale density (Becker *et al.* 2016), vessel traffic characteristics (size + speed + spatial use), along with whale movement patterns obtained from satellite-tagged animals to estimate encounters that would result in mortality. The estimated number of annual vessel strike deaths was 43 fin whales, though this includes only the period July – November when whales are most likely to be present in the U.S. West Coast EEZ and the season that overlaps with cetacean habitat models generated from line-transect surveys (Becker *et al.* 2016, Rockwood *et al.* 2017). This estimate is based on an assumption of a moderate level of vessel avoidance (55%) by fin whales, as measured by the behavior of satellite-tagged *blue whales* in the presence of vessels (McKenna *et al.* 2015). The estimated mortality of 43 fin whales annually due to vessel strikes represents approximately 0.4% of the estimated population size (43 deaths / 11,065 whales). The results of Rockwood *et al.* (2017) also include a no-avoidance encounter model that results in a worst-case estimate of 95 fin whale vessel strike deaths per year, representing approximately 0.8% of the estimated population size. The authors also note that 65% of fin whale vessel strike mortalities occur within 10% of the study area, implying that vessel avoidance mitigation measures may be effective if applied over relatively small regions. Rockwood *et al.* (2017) also estimated a worst-case vessel strike carcass recovery rate of 5% for fin whales, but this estimate was based on a multi-species average from three species (gray, killer and sperm whales). Another way to estimate carcass recovery and/or documentation rates of fin whales killed or seriously injured by vessels is by directly comparing the documented number of vessel strike deaths and serious injuries with annual estimates of vessel strikes from Rockwood *et al.* (2017). Comprehensive coast-wide data on vessel strike deaths and serious injuries assumed to result in death are compiled in annual reports on observed anthropogenic morality for the 15-year period 2007-2021 (Carretta *et al.* 2013, 2018, 2020, 2021, 2022, 2023). During this 15-year period, there were 23 observations of fin whale vessel strike deaths and 1 serious injury, or 1.6 fin whales annually. The ratio of documented vessel strike deaths (1.6/yr) to estimated annual deaths from the moderate avoidance model (43) implies a carcass recovery/documentation rate of 3.7%, which is lower than the worst-case estimate of 5% from Rockwood *et al.* (2017). There is uncertainty regarding the estimated number of vessel strike deaths, however, it is apparent that carcass recovery rates of fin whales are low.

Vessel traffic within the U.S. West Coast EEZ continues to be a vessel strike threat to all large whale populations (Redfern *et al*. 2013, Moore *et al.* 2018). However, a complex of vessel types, speeds, and destination ports all contribute to variability in vessel traffic, and these factors may be influenced by economic and regulatory changes. For example, Moore *et al*. (2018) found primary vessel travel routes changed when emission control areas (ECAs) were established off the U.S. West Coast. They also found large vessels typically reduced their speed by 3-6 kts in ECAs between 2008 and 2015. The speed reductions are thought to be a strategy to reduce operating costs associated with more expensive, cleaner burning fuels required within the ECAs. In contrast, Moore *et al*. (2018) noted that some vessels increased their speed when they transited longer routes to avoid the ECAs. Further research is ongoing to understand how variability in vessel traffic affects vessel strike risk and mitigation strategies, though Redfern *et al*. (2019) note that a combination of vessel speed reductions and expansion of areas to be avoided should be considered.

**STATUS OF STOCK**

Fin whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently this stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. NMFS (2019) concluded in its 5-year status review under the ESA that fin whales satisfy the risk analysis criteria for downlisting from endangered to threatened status, which would require future rulemaking. The sum of observed incidental mortality and serious injury, due to commercial fisheries (0.41/yr, including identified and prorated fin whales), plus estimated vessel strikes (43/yr) is 43.4 whales annually, which is less than the calculated PBR (80). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate.

Estimated vessel strike mortality is 43 whales annually, or approximately 0.4% of the estimated population size. As these estimates are model-derived, they are inherently corrected for undocumented and undetected cases, but they represent only a portion of the year (July-December) for which habitat model data are available. The worst-case vessel strike estimate of mortality is 95 whales, based on no avoidance of vessels, or approximately 0.8% of the estimated population size. Neither vessel strike estimate includes incidents outside of the U.S. West Coast EEZ.

There is strong evidence that the population has increased since 1991 (Moore and Barlow 2011, Nadeem *et al.* 2016, Becker *et al*. 2020). Increasing levels of anthropogenic sound in the world's oceans is a habitat concern for whales, particularly for baleen whales that communicate using low-frequency sound

(Croll *et al.* 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged *blue* whales (Goldbogen *et al.* 2013), but it is unknown if fin whales respond in the same manner to such sounds.

**OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY**

Despite anthropogenic impacts from vessel strikes and fishery entanglements, estimates of population size from line-transect surveys and species distribution models have steadily increased from 1991 to 2018 (Moore and Barlow 2011, Nadeem *et al.* 2016, Becker *et al*. 2020, Figure 2). The NMFS (2019) 5-year review of fin whale populations noted that vessel strikes and impacts to their prey base due to climate and ecosystem change or shifts in habitat are two threats that require more study.

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# SEI WHALE (*Balaenoptera borealis borealis*): Eastern North Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) recognizes one stock of sei whales in the North Pacific (Donovan 1991, Wada and Numachi 1991), but evidence exists for multiple populations (Masaki 1977; Mizroch *et al.* 1984; Horwood 1987). Kanda *et al*. (2006) reported there is likely a single population of sei whales in the western North Pacific, based on microsatellite analyses, for the region 37ºN-45ºN and 147ºE-166ºE. Sei whales are distributed far out to sea in temperate waters worldwide and do not appear to be associated with coastal features. Whaling effort for this species was distributed continuously across the North Pacific between 45-55ºN (Masaki 1977). Two sei whales tagged off California were later killed off Washington and British Columbia (Rice 1974). Sei whales are rare in the California Current (Dohl *et al.* 1983; Barlow 2016; Forney *et al.* 1995; Green *et al.* 1992), but were the fourth most common whale taken by California coastal whalers in the 1950s-1960s (Rice 1974). They are extremely rare south of California (Wade and Gerrodette 1993; Lee 1993). Lacking additional data on sei whale population structure, sei whales in the eastern North Pacific (east of longitude 180º) are considered as a separate stock. For the Marine Mammal Protection Act (MMPA) stock assessment reports, sei whales within the Pacific U.S. EEZ are divided into two discrete areas: (1) California, Oregon and Washington waters (this report) and (2) waters around Hawaii. The Eastern North Pacific stock includes animals found within the U.S. west coast EEZ and in adjacent high seas waters; however, because comprehensive data on abundance, distribution, and human-caused impacts are lacking for high seas regions, the status of this stock is evaluated based on data from U.S. EEZ waters of the California Current (NMFS 2023).

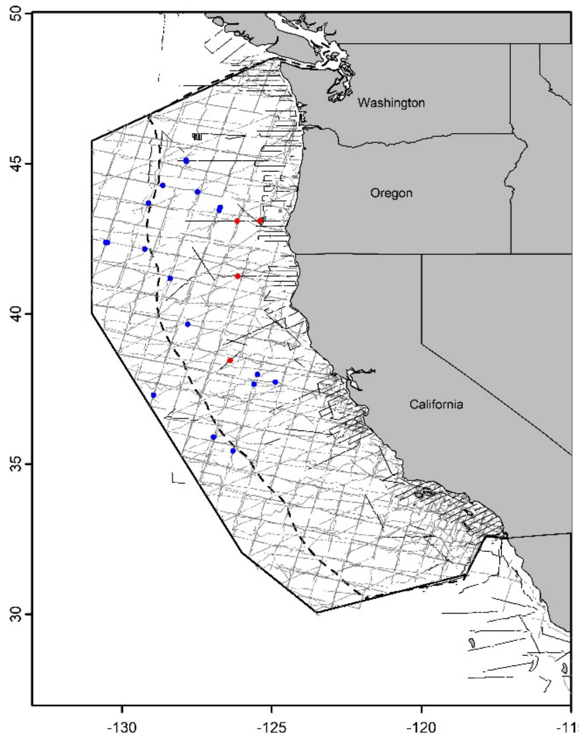


**Figure 1.** Sei whale sighting locations from shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents the U.S. EEZ; thin lines indicate completed transect effort of all surveys combined. Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Ohsumi and Wada (1974) estimated the pre-whaling abundance of sei whales to be 58,000-62,000 in the North Pacific. Tillman (1977) estimated sei whale abundance in the North Pacific and revised this pre-whaling estimate to 42,000. His estimates for the year 1974 ranged from 7,260 to 12,620. These previous studies depended on using the history of catches and trends in CPUE or sighting rates. Hakamada *et al*. (2017) estimated sei whale abundance at 29,632 sei whales (CV = 0.242, 95% CI 18,576–47,267) in the central and eastern North Pacific based on visual line-transect surveys between 2010 and 2012. This estimate corresponds with the first systematic sighting survey abundance estimate for this species over a pelagic high-seas region. However, while the study area of Hakamada *et al*. (2017) included waters north of 40ºN latitude and west of 135ºW longitude, it excluded waters of the California Current, where sightings are rare (Barlow 2016). The most-recent estimate of sei whale abundance in the California Current (864, CV=0.40) is based on a 2014 survey, however this estimate is now 10 years old. Although there was no formal assessment of an abundance trend for sei whales, estimates reported in Barlow (2016) showed an increasing trend from 1991-2014, with the 2014 estimate being the highest estimated.

### Minimum Population Estimate

Although the most-recent abundance estimate for this stock is from 2014, an estimate of minimum population size may be inferred by assuming the population size has at least been stable over the period 1991-2014 (Barlow 2016), based on increasing estimates of abundance. Thus, the minimum population size is calculated from the most-recent estimate (864, CV=0.40), resulting in a minimum population size of 625 whales.

### Current Population Trend

No data on trends in sei whale abundance exist for the eastern North Pacific. Although the population in the North Pacific is expected to have grown since being given protected status in 1976, the possible effects of continued unauthorized takes (Yablokov 1994), vessel strikes and gillnet mortality make this uncertain. Barlow (2016) noted that an increase in sei whale abundance observed in 2014 in the California Current is partly due to recovery of the population from commercial whaling, but may also involve distributional shifts in the population.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the growth rate of sei whale populations in the North Pacific (Best 1993).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (625) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.1 (for an endangered species), resulting in a PBR of 1.25 whales.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

The California swordfish drift gillnet fishery is the most likely U.S. fishery to interact with sei whales from this stock, but no entanglements have been observed from 9,246 observed fishing sets from 1990-2021 (Carretta 2022, Table 1). Mean annual takes for this fishery (Table 1) are based on 2017-2021 data and are zero whales annually. However, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net.

**Table 1.** Summary of available information on the incidental mortality and injury of sei whales (eastern North Pacific stock) for commercial fisheries that might take this species. n/a indicates that data are not available. Mean annual takes are based on 2017-2021 data unless noted otherwise.

| Fishery Name | Year(s) | Data Type | Observer Coverage | Observed mortality (and injury in parentheses) | Estimated mortality (CV in parentheses) | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | 2017<br>2018<br>2019<br>2020<br>2021 | observer | 0.186<br>0.251<br>0.226<br>0.222<br>0.228 | 0 | 0 | 0 (n/a) |

### Vessel Strikes

No vessel strikes of sei whales have been documented during the most-recent 5-yr period (2017-2021) (Carretta *et al*. 2023).

## STATUS OF STOCK

The NMFS sei whale recovery plan notes that basic data such as distribution, abundance, trends and stock structure is of poor quality or largely unknown, owing to the rarity of sightings of this species (NMFS 2011). Sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). The initial abundance has never been reported separately for the eastern North Pacific stock, but this stock was also depleted by whaling. Kanda *et al*. (2006) found a high level of genetic variation among sei whale samples in the western North Pacific and hypothesized that the population did not suffer from a genetic bottleneck due to commercial whaling. Sei whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the eastern North Pacific stock is automatically considered a "depleted" and "strategic" stock under the MMPA. Total observed fishery mortality is zero and therefore is considered to be approaching zero mortality and serious injury rate. Risks to sei whales include vessel strikes, though none were recorded in the most-recent 5-yr period (Carretta *et al*. 2023). Increasing levels of anthropogenic sound in the world's oceans is a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al.* 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al.* 2013), but it is unknown if sei whales respond in the same manner to such sounds.

## OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

The status review of sei whales under the ESA (NOAA 2021) includes threats to sei whales' prey base as the result of climate change. Specifically, the report notes that "<u>In the Northeast Pacific, harmful algal bloom (HAB) events have been increasing in strength, intensity and extension resulting in mortality events for other cetacean and marine mammal species (Cook et al. 2015; Lefebvre et al. 2016; Häussermann et al. 2017). This indicates a similar process may be occurring in both hemispheres. In addition, Sasaki et al. (2012) reported that seasonal shifts in sei whale habitat in the western North Pacific is linked with changing oceanographic conditions due to climate change.</u>" Other potential impacts noted in the report include increasing vessel noise, impacts from oil and gas exploration, military sonar, vessel strikes, and entanglement in fishing gear.

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Revised 05/31/2024

# MINKE WHALE (*Balaenoptera acutorostrata scammoni*): California/Oregon/Washington Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) recognizes 3 stocks of minke whales in the North Pacific: one in the Sea of Japan/East China Sea, one in the rest of the western Pacific west of 180W, and one in the "remainder" of the Pacific (Donovan 1991). The "remainder" stock only reflects the lack of exploitation in the eastern Pacific and does not imply that only one population exists in that area (Donovan 1991). In the "remainder" area, minke whales are relatively common in the Bering and Chukchi seas and in the Gulf of Alaska, but are not considered abundant in any other part of the eastern Pacific (Leatherwood *et al*. 1982; Brueggeman *et al*. 1990). In the Pacific, minke whales are usually seen over continental shelves (Brueggeman *et al*. 1990). In the extreme north, minke whales are believed to be migratory, but in inland waters of Washington and in central California they appear to establish home ranges (Dorsey *et al*. 1990). Minke whales occur year-round in California (Dohl *et al*. 1983; Forney *et al*. 1995; Barlow 1997) and in the Gulf of California (Tershy *et al*. 1990). Minke whales are present at least in summer/fall along the Baja California peninsula (Wade and Gerrodette 1993). Because the "resident" minke whales from California to Washington appear behaviorally distinct from migratory whales further north, minke whales in coastal waters of California, Oregon, and Washington (including Puget Sound) are considered as a separate stock. Minke whales in Alaskan waters are addressed in a separate stock assessment report.

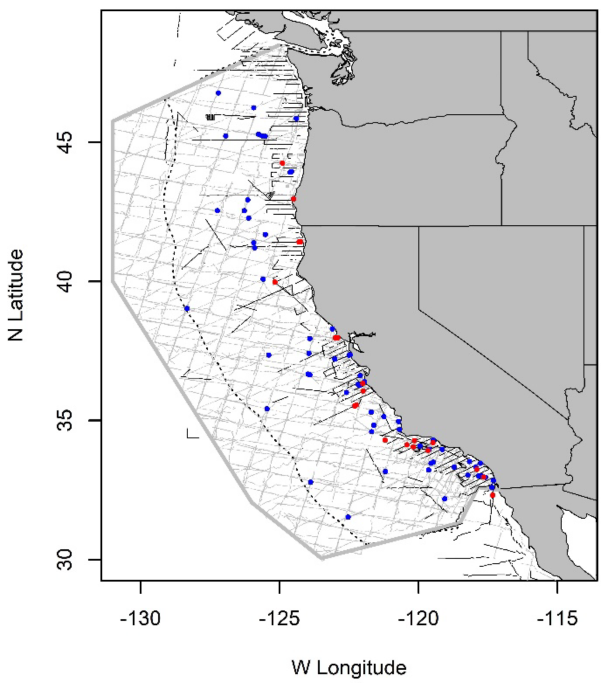


**Figure 1**. Minke whale sightings based on shipboard surveys off California, Oregon, and Washington, 1991-2018. Dashed line represents U.S. EEZ, thin lines indicate completed transect effort (gray = 1991-2014, black = 2018). Sightings from the 2018 survey are shown in red.

## POPULATION SIZE

Becker *et al*. (2020) generated species distribution models (SDMs) from fixed and dynamic ocean variables, using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the California Current Ecosystem (CCE). The use of SDMs for density estimation is well-established for this region and models incorporate changes in species abundance and habitat shifts over time (Becker *et al*. 2016, 2017, 2020, Redfern *et al*. 2017). Additionally, use of SDMs facilitates abundance estimation when survey coverage is limited, as was the case in 2018 when line-transect effort was largely limited to continental shelf waters (Henry *et al*. 2020). The best-estimate of abundance is taken as the estimate from 2018, or 915 (CV=0.792) animals (Becker *et al.* 2020).

### Minimum Population Estimate

The minimum population estimate for minke whales is taken as the lower 20th percentile of the log-normal distribution of the 2018 abundance estimate (Becker *et al*. 2020), or 509 whales.

**Current Population Trend**

No apparent trends in population size are evident from a series of abundance estimates generated from 1991-2018 vessel-based line-transect surveys and habitat-based species distribution models applied to these survey data (Barlow 2016, Becker *et al.* 2016, Figure 2).

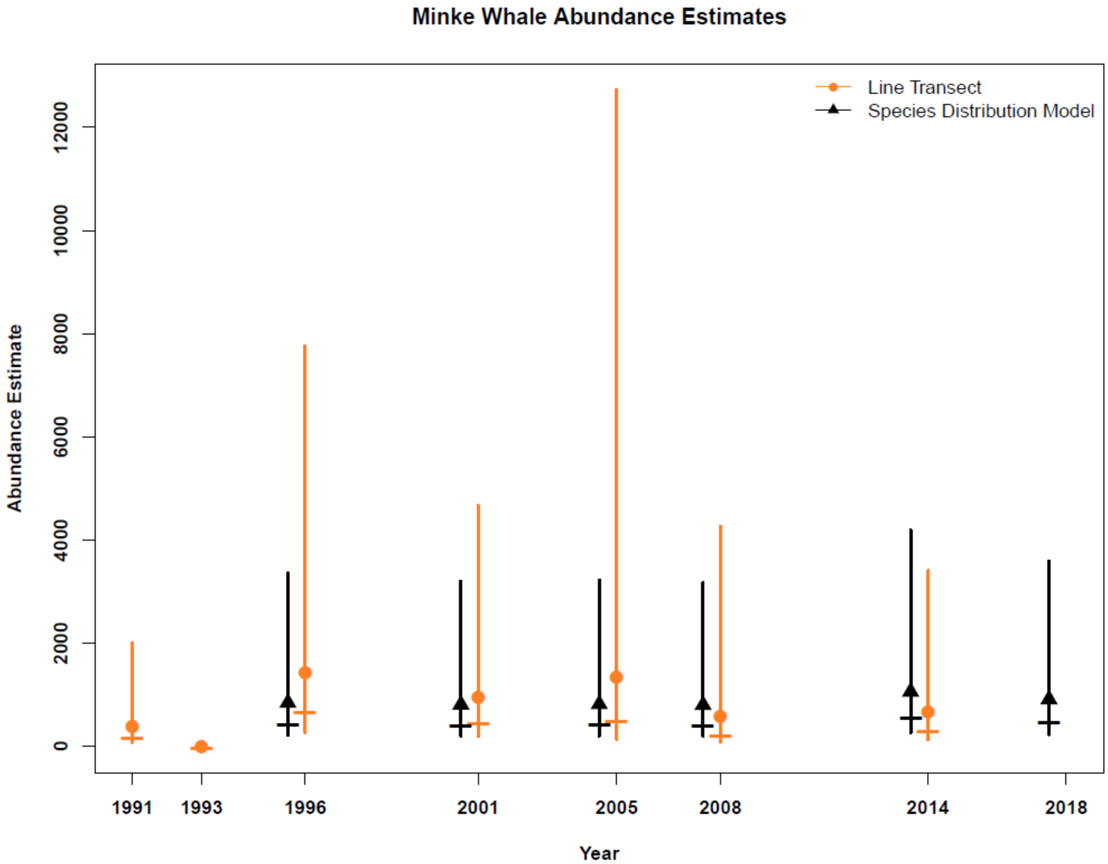


**Figure 2.** Minke whale abundance estimated from vessel-based line transect surveys (Barlow 2016) and habitat-based species distribution models based on 1991-2018 line-transect surveys (Becker *et al*. 2020). Vertical bars indicate approximate 95% log-normal confidence limits for line-transect estimates and 95% confidence limits reported from species distribution model estimates. Line-transect surveys in 1991 and 1993 exclude Oregon and Washington waters. Vertical bars indicate approximate 95% log-normal confidence limits for line-transect and species distribution model estimates. Horizontal hatch marks represent minimum population size estimates based on 20$^{th}$ percentiles of mean estimates.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the growth rate of minke whale populations in the North Pacific (Best 1993).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (509) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.40 (for a stock of unknown status with a mortality estimate CV > 0.80 ), resulting in a PBR of 4.1 whales.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Table 1.** Summary of available information on the incidental mortality and injury of minke whales (CA/OR/WA stock) for commercial fisheries that might take this species (Carretta 2022, Carretta *et al.* 2023). Mean annual takes are based on 2017-2021 data.

| Fishery Name | Years | Data Type | Observer Coverage | Observed mortality (and serious injury) | Estimated Mortality (CV) | Mean Annual Takes (CV) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish gillnet fishery** | 2017<br>2018<br>2019<br>2020<br>2021 | Observer | 0.186<br>0.251<br>0.226<br>0.222<br>0.228 | 0 | 0.1 (3.9) | 0.02 (3.9) |
| **CA halibut and other species large mesh (>3.5”) set gillnet fishery** | 2017 | Observer | ~10% | 0 | 0 | 0 (n/a) |
| **Dungeness Crab Pot Fishery (Oregon)** | 2021 | Sighting | n/a | 0 (0) | 0 | 0 (n/a) |
| **Unidentified fisheries** | 2017-2021 | Sightings and strandings | n/a | 0 (0.75) | 0.75 (n/a) | ≥ 0.15 (n/a) |
| **Total annual takes** | | | | | | ≥ 0.17 (n/a) |

## Fishery Information

Minke whales may occasionally be caught in coastal set gillnets off California, in salmon drift gillnet in Puget Sound, Washington, and in offshore drift gillnets off California. The most-recent estimate of bycatch in the California swordfish drift gillnet fishery is 0.1 (CV=3.9) whales for the 5-year period 2017-2021, or 0.02 whales annually (Carretta 2022, Table 1). This is a model-based estimate based on a total of four minke whales observed entangled (2 dead, 2 released alive) between 1990-2021 from 9,246 observed fishing sets (Carretta 2022). Two additional unidentified fishery interactions with minke whales were recorded during 2015-2019, totaling 0.75 serious injuries/deaths (Carretta *et al*. 2023). One minke whale was disentangled from commercial Dungeness crab pot gear (Oregon) in 2021; the initial and final injury status were non-serious (Carretta *et al*. 2023). The mean annual mortality and serious injury of minke whales from this stock during 2017-2021 is 0.17 animals (Table 1).

## Vessel Strikes

No vessel strikes of minke whales were reported during the most recent 5-years, 2017 to 2021, but most strikes are likely to go undetected compared to larger baleen whales where estimates of vessel strike detection are generally <10% (see blue and fin whale stock assessments).

## Other Mortality

One minke whale carcass attributed to a shooting related death was reported during 2017-2021 (report indicated tremendous hemorrhage associated with being shot through left portion of skull) (Carretta *et al.* 2023).

## STATUS OF STOCK

Minke whales are not listed as "endangered" under the Endangered Species Act and are not considered "depleted" under the MMPA. The annual mortality and serious injury due to fisheries (0.17/yr), shootings (0.2/yr) and vessel strikes (0.0/yr) is less than the calculated PBR for this stock (4.1), so they are not considered a "strategic" stock under the MMPA. Estimated fishery mortality is less than 10% of the PBR; therefore, total fishery mortality is approaching zero mortality and serious injury rate. Trends in the abundance of this stock are unknown. Harmful algal blooms are a habitat concern for minke whales and at least one death along the U.S. west coast has been attributed to domoic acid toxicity resulting from the consumption of northern anchovy prey (Fire *et al.* 2010). Increasing levels of anthropogenic sound in the world’s oceans has been suggested to be a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al*. 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al*. 2013), but it is unknown if minke whales respond in the same manner to such sounds.

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Revised 12/31/2015

# BRYDE'S WHALE (*Balaenoptera edeni*): Eastern Tropical Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) recognizes 3 stocks of Bryde's whales in the North Pacific (eastern, western, and East China Sea), 3 stocks in the South Pacific (eastern, western and Solomon Islands), and one cross-equatorial stock (Peruvian) (Donovan 1991). Bryde's whales are distributed widely across the tropical and warm-temperate Pacific (Leatherwood et al. 1982), and there is no justification for splitting stocks between the northern and southern hemispheres (Donovan 1991). Past surveys have shown them to be common and distributed throughout the eastern tropical Pacific with a concentration around the equator east of 110ºW (corresponding approximately to the IWC's "Peruvian stock") and a lower densities west of 140ºW (Lee 1993; Wade and Gerrodette 1993). They are also the most common baleen whale in the central Gulf of California (Tershy et al. 1990). Sightings and acoustic recordings of Bryde's whales in southern California waters have increased in the past decade (Kerosky *et al*. 2012, Smultea *et al*. 2012), possibly signaling a northward range expansion (Kerosky *et al*. 2012). Acoustic recordings indicate Bryde's whales are present in southern California waters from summer through early winter (Kerosky *et al*. 2012). At least seven sightings have been documented in southern / central California waters between 1991 and 2014 (Barlow and Forney 2007, Smultea *et al*. 2012, Barlow 2016). Bryde's whales in California waters likely belong to a larger population inhabiting at least the eastern part of the tropical Pacific. Acoustic call types of Bryde's whales in southern California waters match a type found along the west coast of Baja California (Kerosky *et al*. 2012). For the Marine Mammal Protection Act (MMPA) stock assessment reports, Bryde's whales within the Pacific U.S. Exclusive Economic Zone are divided into two areas: 1) the eastern tropical Pacific (east of 150ºW and including the Gulf of California and waters off California; this report), and 2) Hawaiian waters.

## POPULATION SIZE

In the western North Pacific, Bryde's whale abundance in the early 1980s was estimated independently by tag mark-recapture and ship survey methods to be 22,000 to 24,000 (Tillman and Mizroch 1982; Miyashita 1986). Bryde's whale abundance has never been estimated for the entire eastern Pacific; however, a portion of that stock in the eastern tropical Pacific was estimated as 13,000 (CV=0.20; 95% CI = 8,900-19,900) (Wade and Gerrodette 1993), and the minimum number in the Gulf of California was estimated at 160 based on individually-identified whales (Tershy et al. 1990). The most recent verified sighting in California waters occurred in 2014 during a systematic line-transect survey designed to estimate cetacean abundance (Barlow 2016). That sighting did not occur during standard search effort and thus, no estimate of abundance is available from the 2014 survey.

### Minimum Population Estimate

The only minimum estimate of Bryde's whale abundance for the eastern tropical Pacific (11,163; Wade and Gerrodette 1993) is over 8 years old and thus, no current estimate of minimum abundance is available.

### Current Population Trend

There are no data on trends in Bryde's whale abundance in the eastern tropical Pacific.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

There are no estimates of the growth rate of Bryde's whale populations in the Pacific (Best 1993).

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock cannot be calculated because a current abundance estimate is unavailable.

## HUMAN CAUSED MORTALITY

### Historic Whaling

The reported take of North Pacific Bryde's whales by commercial whalers totaled 15,076 in the western Pacific from 1946-1983 (Holt 1986) and 2,873 in the eastern Pacific from 1973-81 (Cooke 1983). In addition, 2,304 sei-or-Bryde's whales were taken in the eastern Pacific from 1968-72 (Cooke 1983) (based on subsequent catches, most of these were probably Bryde's whales). None were reported taken by shore-based whaling stations in central or northern California between 1919 and 1926 (Clapham et al. 1997) or 1958 and 1965 (Rice 1974). There has been a prohibition on taking Bryde's whales since 1988.

**Table 1.** Summary of available information on the incidental mortality and injury of Bryde’s whales (eastern tropical Pacific stock) for commercial fisheries that might take this species (Carretta *et al*. 2014a, 2012a, 2012b, Carretta and Enriquez 2009, 2010; Carretta *et al*. 2004). n/a indicates that data are not available. Mean annual takes are based on 2001-2013 data unless noted otherwise.

| Fishery Name | Year(s) | Data Type | Percent Observer Coverage | Observed mortality (and injury in parentheses) | Estimated mortality (CV in parentheses) | Mean annual takes (CV in parentheses) |
|---|---|---|---|---|---|---|
| **CA/OR thresher shark/swordfish drift gillnet fishery** | 2001-2013 | observer | 19% | 0 | 0 | 0 |
| | | | | | | |
| **Total annual takes** | | | | | | 0 |

## Fishery Information

The California swordfish drift gillnet fishery is the only fishery that is likely to take Bryde’s whales from this stock, but no entanglements have been observed (Table 1). Detailed information on this fishery is provided in Appendix 1. Mean annual takes for this fishery are zero (Table 1) and are based on 2001-2013 data, the period during which a season/area closure has limited most fishing to southern California waters. Although Bryde’s whales have not been observed entangled in California gillnets, some gillnet mortality of large whales may go unobserved because whales swim away with a portion of the net.

## Ship Strikes

One Bryde’s whale was documented to have been killed by a ship strike in 2010 (Carretta et al. 2014b, Carretta et al. 2015). The whale was initially sighted alive in Washington state waters with propeller marks and stranded dead about a week later. The mean annual serious injury and mortality rate of Bryde’s whales over the most recent 5-year period (2009-2013) is 0.2 whales annually.

## STATUS OF STOCK

Commercial whaling of Bryde's whales was largely limited to the western Pacific. Bryde's whales are not listed as "threatened" or "endangered" under the Endangered Species Act (ESA). Bryde's whales in the eastern tropical Pacific would not be considered a strategic stock under the MMPA. The total human-caused mortality rate is 0.2 whales annually. Current abundance of this stock is unknown and therefore PBR cannot be calculated for this stock. Likewise, human-caused mortality cannot be evaluated in the context of PBR. Increasing levels of anthropogenic sound in the world’s oceans has been suggested to be a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound.

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Revised 5/31/2024

# ROUGH-TOOTHED DOLPHIN (*Steno bredanensis*): Hawai‘i Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins are found throughout the world in tropical and warm-temperate waters (Perrin *et al*. 2009), with genetic analysis of samples from all ocean basins revealing divergence between rough-toothed dolphins in the Atlantic versus those in the Indian and Pacific Oceans, suggesting the occurrence of two subspecies (Albertson *et al*. 2022). They are present around all the main Hawaiian Islands, though are relatively uncommon within the Maui Nui region (Baird *et al*. 2013), and have been observed close to the islands and atolls at least as far northwest as Pearl and Hermes Reef (Bradford *et al*. 2017). Rough-toothed dolphins are occasionally seen offshore throughout the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands during periodic shipboard surveys (Figure 1). Rough-toothed dolphins have also been documented in American Samoan waters (Oleson 2009).

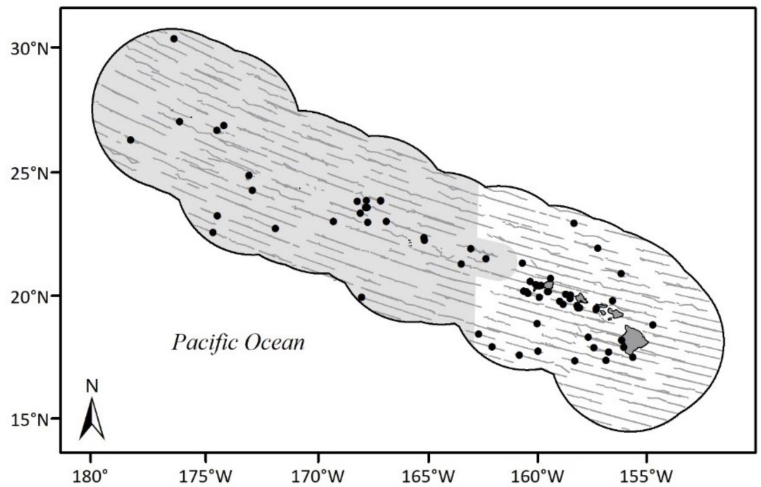


**Figure 1**. Rough-toothed dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al.* 2018) shipboard cetacean surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

Population structure in rough-toothed dolphins was recently examined using genetic samples from several tropical and sub-tropical island areas in the Pacific. Albertson *et al*. (2016) found significant differentiation in mtDNA and nuDNA from samples collected at Hawai‘i Island versus all other Hawaiian Island areas sampled. Estimates of differentiation among Kaua‘i, O‘ahu, and the northwestern Hawaiian Islands (NWHI) were lower and not statistically significant. Based on their result, Albertson *et al.* (2016) suggested that Hawai‘i Island warranted designation as a separate island-associated stock. Evaluation of individual rough-toothed dolphin encounters indicate differences in group sizes, habitat use, and behavior between groups seen near Hawai‘i Island and those seen near Kaua‘i and Ni‘ihau (Baird *et al*. 2008). Photographic identification studies suggested that dispersal rates between the islands of Kaua‘i/Ni‘ihau and Hawai‘i do not exceed 2% per year (Baird *et al.* 2008). Resighting rates off the island of Hawai‘i are high, with 75% of well-marked individuals resighted on two or more occasions, suggesting high site fidelity and low population size. Movement data from 17 individual rough-toothed dolphins tagged near Kaua‘i and Ni‘ihau show all individuals remained associated with Kaua‘i with exception of one individual that moved from Kaua‘i and O‘ahu and back (Baird 2016). The available genetics, movements, and social affiliation data suggest that there is at least one island-associated stock in the main Hawaiian Islands (MHI). Delineation of island-associated stocks of rough-toothed dolphins in Hawai‘i is under review (Martien *et al*. 2016).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are two Pacific management stocks: 1) the Hawai‘i Stock (this report), and 2) the American Samoa Stock. The Hawai‘i stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ of the Hawaiian Islands (NMFS 2023a).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of rough-toothed dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2022, Bradford *et al*. 2021; Table 1).

**Table 1**. Line-transect abundance estimates for rough-toothed dolphins in the Hawaiian Islands EEZ in 2002, 2010, 2017, and 2020, derived from NMFS surveys in the central Pacific since 1986 (Becker *et al.* 2022, Bradford *et al.* 2021).

| Year | Design-based Abundance | CV | 95% Confidence Limits | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|---|---|---|
| 2020 | - | - | - | 83,915 | 0.49 | 34,025-206,958 |
| 2017 | 76,375 | 0.41 | 35,286-165,309 | 86,068 | 0.49 | 34,857-212,519 |
| 2010 | 74,001 | 0.39 | 35,197-155,586 | | | |
| 2002 | 65,959 | 0.39 | 31,344-138,803 | | | |

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for the 2017 to 2020 period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for rough-toothed dolphins from Barlow *et al.* (2015). Although model-based estimates were previously derived for years 2002, 2010, and 2017 (Becker *et al.* 2021), those estimates did not include any dynamic environmental covariates, such that they were uninformative for individual survey years. Model-based estimates were derived only for the most recent years (2017-2020), such that direct comparison of model and design-based estimates for the full survey time series is not possible at this time. Bradford *et al.* (2021) produced design-based abundance estimates for rough-toothed dolphins for each full EEZ survey year, with the 2017 design-based and 2017 and 2020 model-based estimates largely similar in the mean estimate and confidence limits (Figure 2). Current model based-estimates are based on the implicit assumption that annual changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for the most recent survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of rough-toothed dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020 and found no significant difference, with no reliance on dynamic variables within the model-based approach, but roughly 30% higher density in summer-fall (though with broad and overlapping confidence intervals) based on the design-based estimates. Previously published design-based estimates for the Hawaiian Islands EEZ from 2002 and 2010 surveys (Barlow 2006, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 83,915 (CV=0.49) rough-toothed dolphins.

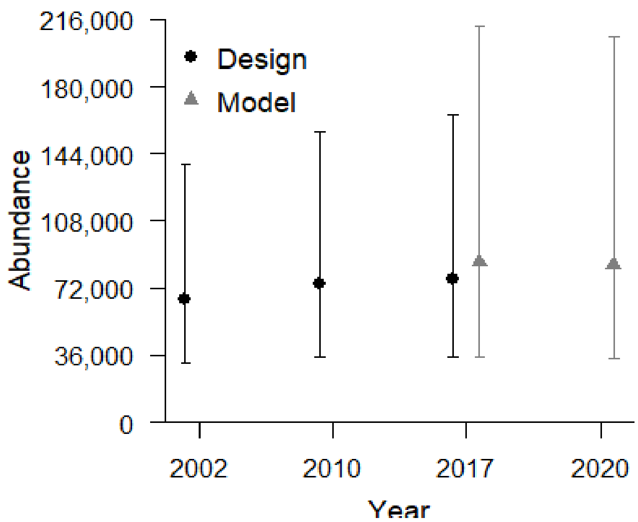


**Figure 2.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for rough-toothed dolphins for each survey year (2002, 2010, 2017, 2020).

A population estimate for this species has been made in the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands. Mark-recapture estimates for the islands of Kaua‘i/Ni‘ihau and Hawai‘i were derived from identification photographs obtained between 2003 and 2006, resulting in estimates of 1,665 (CV=0.33) around Kaua‘i/Ni‘ihau and 198 (CV=0.12) around the island of Hawai‘i (Baird *et al.* 2008). Such estimates may be representative of smaller island-associated populations at those island areas.

**Minimum Population Estimate**

The minimum population estimate is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) of the 2020 abundance estimate (from Becker *et al.* 2022) or 56,782 rough-toothed dolphins within the Hawaiian Islands EEZ.

**Current Population Trend**

The available abundance estimates for this stock have broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

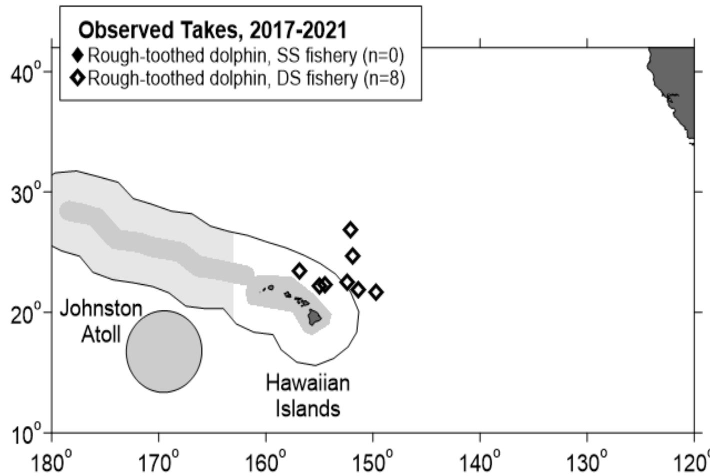


**Figure 3.** Locations of observed rough-toothed dolphin takes within the deep-set fishery (open diamonds) in the Hawaii-based longline fishery, 2017-2021. Solid lines represent the U.S. EEZ. Gray shading notes areas closed to longline fishing.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for the Hawaiʻi stock of rough-toothed dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (56,782) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.45 (for a stock of unknown status with Hawaiian Islands EEZ mortality and serious injury rate CV > 0.30; Wade and Angliss 1997), resulting in a PBR of 511 rough-toothed dolphins per year.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Fishery Information**

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Rough-toothed dolphins are known to take bait and catch from several Hawaiian sport and commercial fisheries operating near the main islands (Shallenberger 1981; Schlais 1984; Nitta & Henderson 1993). They have been specifically reported to interact with the day handline fishery for tuna (palu-ahi), the night handline fishery for tuna (ika-shibi), and the troll fishery for billfish and tuna (Schlais 1984; Nitta & Henderson 1993). Baird *et al*. (2008) reported increased vessel avoidance of boats by rough-toothed dolphins off the island of Hawaiʻi relative to those off Kauaʻi or Niʻihau and attributed this to possible shooting of dolphins that are stealing bait or catch from recreational fisherman off the island of Hawaiʻi (Kuljis 1983). Rough-toothed dolphins have been observed in nearshore waters with serious injuries resulting from fishing gear trailing from or wrapped around their bodies, though the source of the gear was not identified (Bradford and Lyman 2018). Photographs of 52 individuals with greater than 50% of the mouthline photographed showed evidence of injuries consistent with interactions with hook and line fisheries (Welch 2017). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017 and 2021, no rough-toothed dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage), but eight rough-toothed dolphins were observed taken in the DSLL fishery (15-21% observer coverage) (Figure 3, Bradford 2018, 2020, in press, in review,

**Table 2.** Summary of available information on incidental mortality and serious injury (MSI) of rough-toothed dolphins (McCracken & Cooper 2022b). Mean annual takes are based on 2017-2021 data unless indicated otherwise. Information on all observed takes (T) and MSI is included. Total takes were prorated to deaths, serious injuries, and

non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Outside U.S. EEZ | | Hawaiian Islands EEZ | |
|---|---|---|---|---|---|---|---|
| | | | | **Observed T/MSI** | **Estimated MSI (CV)** | **Observed T/MSI** | **Estimated MSI (CV)** |
| **Hawai'i-based deep-set longline fishery** | 2017 | Observer data | 20% | 0 | 0 (-) | 0 | 0 (-) |
| | 2018 | | 18% | 0 | 0 (-) | 0 | 0 (-) |
| | 2019 | | 21% | 1/0 | 3 (1.2) | 0 | 0 (-) |
| | 2020 | | 15% | 3/2 | 14 (0.5) | 2/2 | 10 (0.6) |
| | 2021 | | 18% | 1/1 | 5 (0.9) | 1/1 | 6 (0.9) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | **4.4 (0.5)** | | **3.2 (0.6)** |
| **Hawai'i-based shallow-set longline fishery** | 2017 | Observer data | 100% | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 0 | 0 | 0 | 0 |
| | 2019 | | 100% | 0 | 0 | 0 | 0 |
| | 2020 | | 100% | 0 | 0 | 0 | 0 |
| | 2021 | | 100% | 0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | | **0** | | **0** |
| **Minimum total annual takes within U.S. EEZ (2017-2021)** | | | | | | | **3.2 (0.6)** |

McCracken & Cooper 2022b). In the DSLL fishery, 5 rough-toothed dolphins were taken outside the U.S. EEZ, including 1 rough-toothed dolphin found dead, 2 considered seriously injured, and 1 considered non-seriously injured based on an evaluation of the observer's description of each interaction and following criteria for assessing serious injury in marine mammals (NMFS 2023b). Inside of the Hawaiian Islands EEZ, 2 were observed dead and 1 determined to be seriously injured.

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the U.S. EEZ), and the ratio of observed dead and seriously injured dolphins versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for rough-toothed dolphins during 2017-2021 are 4.4 (CV=0.5) rough-toothed dolphins outside of the Hawaiian Islands EEZ, and 3.2 (CV=0.6) rough-toothed dolphins within the Hawaiian Islands EEZ (Table 2, McCracken and Cooper 2022b).

## STATUS OF STOCK

The Hawai'i stock of rough-toothed dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of rough-toothed dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Rough-toothed dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries related to nearshore fisheries; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus total mean annual takes are undetermined. The total number of estimated rough-toothed dolphins killed or seriously injured by longline fisheries inside (3.2) and outside (4.4) of the Hawaiian Islands EEZ is less than 10% of PBR (51), such that the fishery-related mortality or serious injuries rate for the entire Hawai'i stock can be considered to be insignificant and approaching zero. Island-associated populations of rough-toothed dolphins may experience relatively greater rates of fisheries mortality and serious injury. One rough-toothed dolphin stranded in the main Hawaiian Islands tested positive for *Brucella* (Chernov 2010) and another for *Morbillivirus* (Jacob 2012). *Brucella* is a bacterial infection that, if common in the population, may limit recruitment by compromising male and female reproductive systems, and can also cause neurological disorders that may result in death (Van Bressem *et al.* 2009). Although *morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animal is not known as it was found in only a few tested tissues (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species (Jacob *et al*. 2016) and *Brucella* in 3 species (Chernov 2010) raises concerns about the history and prevalence of these diseases in Hawaii and the potential population impacts, including cumulative impacts of disease with other stressors.

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Revised 01/15/2011

# ROUGH-TOOTHED DOLPHIN *(Steno bredanensis)*: American Samoa Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Rough-toothed dolphins are found throughout the world in tropical and warm-temperate waters (Perrin et al. 2009). Rough-toothed dolphins are common in the South Pacific from the Solomon Islands, where they were taken by dolphin hunters, to French Polynesia and the Marquesas (reviewed by Reeves et al 1999). Rough-toothed dolphins have been observed during summer and winter surveys around the American Samoan island of Tutuila (Johnston et al. 2008) and are thought to be common throughout the Samoan archipelago (Craig 2005). Rough-toothed dolphins were among the most commonly-sighted cetaceans during small boat surveys conducted from 2003 to 2006 around Tutuila, though not observed during a 2006 survey of Swain's Island and the Manu'a Group (Johnston et al. 2008). Photo-identification data collected during the surveys suggest the presence of a resident population of rough-toothed dolphins in the waters surrounding Tutuila (Johnston et al. 2008). Approximately 1/3 of the individuals within the photo-id catalog were sighted in multiple years (Johnston et al. 2008). One rough-toothed dolphin was taken entangled near 40-fathom bank south of the islands by the American Samoa-based longline in 2008 (Oleson 2009), indicating some rough-toothed dolphins maintain a more pelagic distribution. Nothing is known about stock structure for this species in the South Pacific. For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are two Pacific management stocks: 1) The Hawaiian Stock, which includes animals found within the U.S. EEZ of the Hawaiian Islands, and 2) the American Samoa Stock, which include animals inhabiting the EEZ waters around American Samoa (this report).

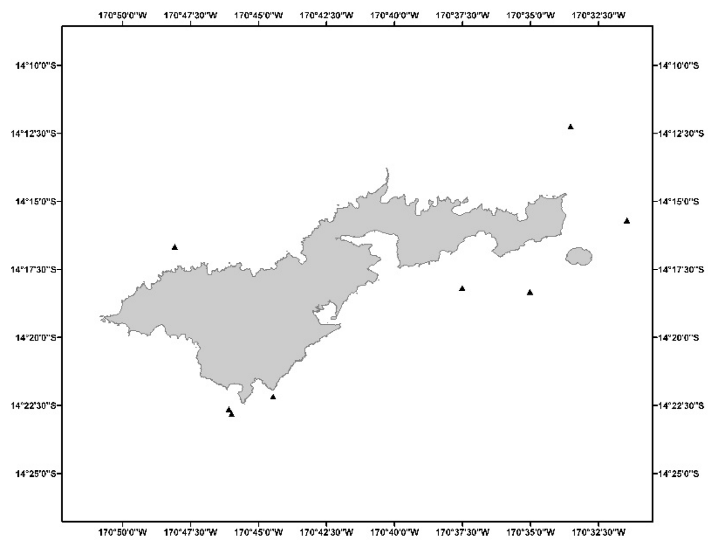


**Figure 1.** Rough-toothed dolphin sightings during cetacean sighting surveys around Tutuila, American Samoa, 2003-2006 (Johnston et al, 2008).

## POPULATION SIZE

No abundance estimates are currently available for rough-toothed dolphins in U.S. EEZ waters of American Samoa; however, density estimates for rough-toothed dolphins in other tropical Pacific regions can provide a range of likely abundance estimates in this unsurveyed region. Published estimates of rough-toothed dolphins (animals per km$^2$) in the Pacific are: 0.0035 (CV=0.45) for the U.S. EEZ of the Hawaiian Islands (Barlow 2006); 0.0017 (CV=0.63) for nearshore waters surrounding the main Hawaiian Islands (Mobley et al. 2000), 0.0076 (CV=0.32) and 0.0017 (CV=0.16) for the eastern tropical Pacific Ocean (Wade and Gerrodette 1993; Ferguson and Barlow 2003). Applying the lowest and highest of these density estimates to U.S. EEZ waters surrounding American Samoa (area size = 404,578 km$^2$) yields a range of plausible abundance estimates of 692 – 3,115 rough-toothed dolphins.

### Minimum Population Estimate

No minimum population estimate is currently available for waters surrounding American Samoa, but the rough-toothed dolphin density estimates from other tropical Pacific regions (Barlow 2003, Mobley et al. 2000, Wade and Gerrodette 1993, Ferguson and Barlow 2003, see above) can provide a range of likely values. The log-normal 20$^{th}$ percentiles of plausible abundance estimates for the American Samoa EEZ, based on the densities observed elsewhere, range from 426 – 2,731 rough-toothed dolphins.

## Current Population Trend

No data are available on current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

No PBR can presently be calculated for rough-toothed dolphins within the American Samoa EEZ, but based on the range of plausible minimum abundance estimates (426 – 2,731), a recovery factor of 0.40 (for a species of unknown status with a fishery mortality and serious injury rate CV > 0.50 within the American Samoa EEZ; Wade and Angliss 1997), and the default growth rate (½ of 4%), the PBR would likely fall between 3.4 and 22 rough-toothed dolphins per year.

## ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in American Samoan fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used, and float lines from lobster traps and longlines can be expected to occasionally entangle whales (Perrin et al. 1994). The primary fishery in American Samoa is the commercial pelagic longline fishery that targets tunas, which was introduced in 1995 (Levine and Allen 2009). In 2008, there were 28 federally permitted vessels within the longline fishery in American Samoa. The fishery has been monitored since March 2006 under a mandatory observer program, which records all interactions with protected species (Pacific Islands Regional Office 2009). One rough-toothed dolphin was seriously injured by the fishery in 2008 (Oleson 2009).

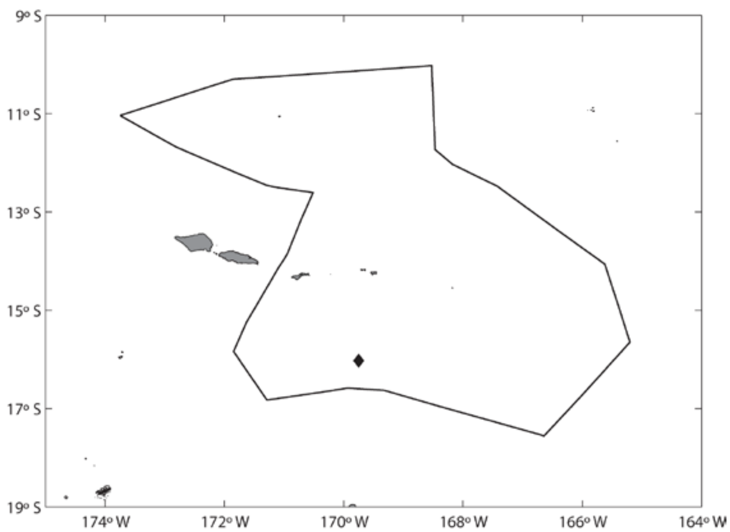


**Figure 2.** Locations of observed rough-toothed dolphin takes (filled diamonds) in the American Samoa longline fishery, 2006-2008. Solid lines represent the U.S. EEZ. Set locations in this fishery are summarized in Appendix 1.

**Table 1.** Summary of available information on incidental mortality and serious injury of rough-toothed dolphins (American Samoan stock) in commercial fisheries within the U.S. EEZ (Oleson 2009). Longline fishery take estimates represent only those trips with at least 10 sets/trip (Oleson 2009). Mean annual takes are based on 2006-2008 data unless otherwise indicated.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Observed and estimated mortality and serious injury of rough-toothed dolphins in the American Samoa EEZ | | |
|---|---|---|---|---|---|---|
| | | | | American Samoa EEZ | | |
| | | | | Obs. | Estimated (CV) | Mean Annual Takes (CV) |
| **American Samoa-based longline fishery** | 2006<br>2007<br>2008 | observer data | 9.0%<br>7.7%<br>8.5% | 0<br>0<br>1 | 0 (-)<br>0 (-)<br>10.9 (2.0) | 3.6 (0.6) |
| **Minimum total annual takes within U.S. EEZ waters** | | | | | | 3.6 (0.6) |

Prior to 1995, bottom fishing and trolling were the primary fisheries in American Samoa but they became less prominent after longlining was introduced (Levine and Allen 2009). Nearshore subsistence fisheries include spear fishing, rod and reel, collecting, gill netting, and throw netting (Craig 1993, Levine and Allen 2009). Information on fishery-related mortality of cetaceans in the nearshore fisheries is unknown, but the gear types used in American Samoan fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used. Although boat-based nearshore fisheries have been randomly monitored since 1991, by the American Samoa Department of Marine and Wildlife Sources (DMWR), no estimates of annual human-caused mortality and serious injury of cetaceans are available.

## STATUS OF STOCK

The status of rough-toothed dolphins in American Samoan waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this species. It is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The status of the American Samoan stock of rough-toothed dolphins under the 1994 amendments to the MMPA cannot be determined at this time because no abundance estimates are available and PBR cannot be calculated. However, the estimated rate of fisheries-related mortality or serious injury within the American Samoa EEZ (3.6 animals per year) is between the range of likely PBRs (3.4 – 22) for this region. Insufficient information is available to determine whether the total fishery mortality and serious injury for rough-toothed dolphins is insignificant and approaching zero mortality and serious injury rate.

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Revised 5/31/2024

# RISSO'S DOLPHIN (*Grampus griseus*): Hawai‘i Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Risso's dolphins are found in tropical to warm-temperate waters worldwide (Perrin *et al*. 2009). Risso’s dolphins represent less than 1% of all odontocete sightings in leeward surveys of the main Hawaiian Islands from 2000 to 2012 (Baird *et al*. 2013); however, they are regularly sighted during periodic shipboard surveys of the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands (Figure 1). Most sightings of Risso’s dolphins occur in deep waters offshore. A single satellite tagged animal moved broadly between offshore waters off Kona, Koho‘olawe, and Lāna‘i over a 2-week period (Baird 2016). Sighting, habitat, and limited movement data do not appear to support finer population structure in Hawaiian waters, though differences in the spectral characteristics of Risso’s dolphin echolocation clicks between Hawai‘i and the U.S. West Coast suggest there may be an indication of population differentiation within the ocean basin (Soldevilla *et al.* 2017).

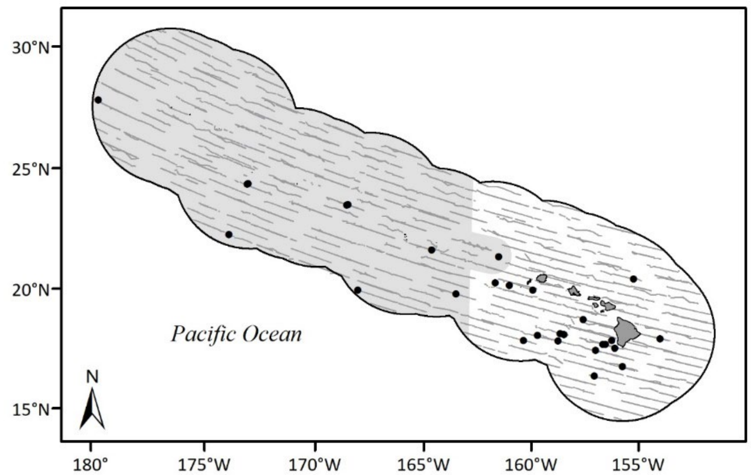


**Figure 1**. Risso's dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al.* 2018) shipboard cetacean surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, Risso's dolphins within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawai‘i stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ of the Hawaiian Islands (NMFS 2023).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in updated model-based abundance estimates of Risso’s dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al*. 2021, 2022; Table 1).

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates. The modeling framework incorporated Beaufort-specific trackline detection probabilities for Risso’s dolphins from Barlow *et al*. (2015). Models were used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al*. 2015, Becker *et al*. 2016). Bradford *et al*. (2021) produced design-based abundance estimates for Risso’s dolphins in 2002, 2010, and 2017 that can be used as a point of comparison to the model-based estimates for those years. While on average, the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation.

Table 1. Model-based line-transect abundance estimates for Risso's dolphins in the Hawaiian Islands EEZ in 2002 and 2010 (Becker *et al*. 2021) and 2017 and 2020 (Becker *et al.* 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

| Year | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2020 | 6,979 | 0.29 | 3,649-13,348 |
| 2017 | 7,437 | 0.30 | 4,027-13,736 |
| 2010 | 6,174 | 0.20 | 4,159-9,165 |
| 2002 | 6,916 | 0.21 | 4,623-10,346 |

Model based-estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of Risso's dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Both analyses showed slightly higher densities of Risso's dolphins in the MHI in winter, with the spatially-explicit model showing marked differences in winter and non-winter distribution driven by the relationship with mixed layer depth for this species. Previously published abundance estimates for the Hawaiian Islands EEZ (Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al*. 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al*. (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 6,979 (CV=0.29) Risso's dolphins.

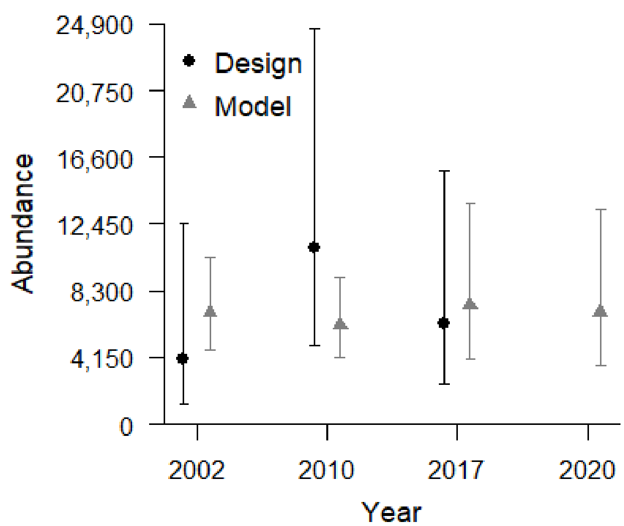


**Figure 2.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for Risso's dolphins for each survey year (2002, 2010, 2017, 2020).

Population estimates have been made off Japan (Miyashita 1993), in the eastern tropical Pacific (Wade and Gerrodette 1993), and off the U.S. West Coast (Barlow 2016), but it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands and in the central North Pacific.

### Minimum Population Estimate

The minimum population estimate is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) of the 2020 abundance estimate (from Becker *et al.* 2022), or 5,283 Risso's dolphins within the Hawaiian Islands EEZ.

### Current Population Trend

The model-based abundance estimates for Risso's dolphins provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Model-based examination of Risso's dolphin trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of Risso's dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (5,283) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with no known fishery mortality and serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 53 Risso's dolphins per year.

## HUMAN CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No interactions between nearshore fisheries and Risso's dolphins have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

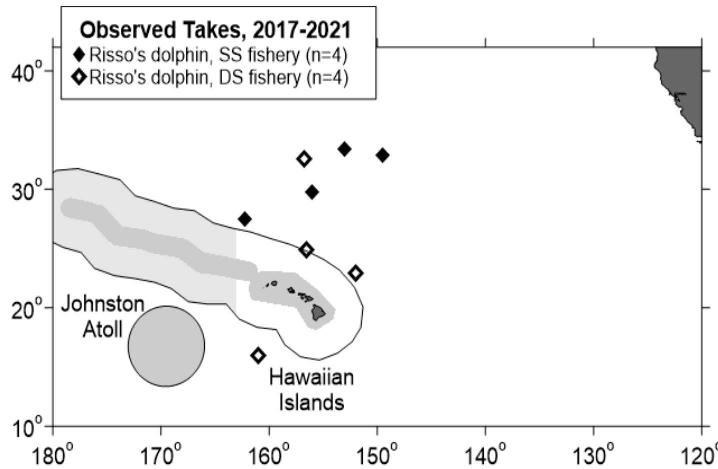


**Figure 3.** Locations of observed Risso's dolphin takes within the shallow-set fishery (filled diamonds) and deep-set fishery (open diamonds) in the Hawai'i-based longline fishery,2017-2021. Solid lines represent the U.S. EEZ. Gray shading notes areas closed to longline fishing.

There are currently two distinct longline fisheries based in Hawai'i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017 and 2021, 4 Risso's dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage), and 4 Risso's dolphins were observed taken in the DSLL fishery (15-21% observer coverage) (Figure 3, Bradford 2018, 2020, 2021, In press, In review, McCracken and Cooper 2022b). One Risso's dolphin in the DSLL fishery was killed, 4 in the SSLL fishery and 2 in the DSLL fishery were considered to have been seriously injured, all outside the U.S. EEZ.

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2021 are 5.0 (CV=0.4) Risso's dolphins outside of U.S. EEZ, and 0 within the Hawaiian Islands EEZ (Table 2, McCracken and Cooper 2022b).

**Table 2.** Summary of available information on incidental mortality and serious injury (MSI) of Risso's dolphin (Hawaii stock) in commercial longline fisheries, within and outside of U.S. EEZ (McCracken and Cooper, 2022b). Mean annual takes are based on 2017-2021 data unless indicated otherwise. Information on all observed takes (T) and MSI is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Outside U.S. EEZ | | Hawaiian Islands EEZ | |
|---|---|---|---|---|---|---|---|
| | | | | **Observed T/MSI** | **Estimated MSI (CV)** | **Observed T/MSI** | **Estimated MSI (CV)** |
| **Hawai‘i-based deep-set longline fishery** | 2017 | Observer data | 20% | 1/1 | 5 (0.9) | 0 | 0 (-) |
| | 2018 | | 18% | 0 | 0 (-) | 0 | 0 (-) |
| | 2019 | | 21% | 1/1 | 7 (0.9) | 0 | 0 (-) |
| | 2020 | | 15% | 2/1 | 16 (0.5) | 0 | 0 (-) |
| | 2021 | | 18% | 0/0 | 0 (-) | 0 | 0 (-) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | **5.0 (0.4)** | | **0 (-)** |
| **Hawai‘i-based shallow-set longline fishery** | 2017 | Observer data | 100% | 2/2 | 2 | 0 | 0 |
| | 2018 | | 100% | 2/2 | 2 | 0 | 0 |
| | 2019 | | 100% | 0/0 | 0 | 0 | 0 |
| | 2020 | | 100% | 0/0 | 0 | 0 | 0 |
| | 2021 | | 100% | 0/0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | | **0.8** | 0 | **0** |
| **Minimum total annual takes within U.S. EEZ (2017-2021)** | | | | | | | **0 (-)** |

## STATUS OF STOCK

The Hawai‘i stock of Risso's dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of Risso's dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Risso's dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within the Hawaiian Islands EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. One Risso's dolphin stranded on the MHI tested positive for *Morbillivirus* (Jacob *et al*. 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters, all identified as a unique strain of *morbillivirus* (Jacob *et al*. 2016), raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including cumulative impacts of disease with other stressors.

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Revised 5/31/2024

# COMMON BOTTLENOSE DOLPHIN (*Tursiops truncatus truncatus*): Hawaiian Islands Stock Complex – Kaua'i/Ni'ihau, O'ahu, Maui Nui, Hawai'i Island, and Hawai'i Pelagic Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Common bottlenose dolphins are widely distributed throughout the world in tropical and warm-temperate waters (Perrin *et al.* 2009). Bottlenose dolphins are common throughout the Hawaiian Islands, both in nearshore waters as well as at great distances from shore (Baird *et al*. 2013, Bradford *et al*. 2021; Figure 1).

Separate offshore and coastal forms of bottlenose dolphins have been identified along continental coasts (Ross and Cockcroft 1990; Van Waerebeek *et al.* 1990), and there is evidence that similar onshore-offshore forms may exist in Hawaiian waters (Baird 2016). In their analysis of sightings of bottlenose dolphins in the eastern tropical Pacific (ETP), Scott and Chivers (1990) noted a large hiatus between the westernmost sightings and the Hawaiian Islands. These data suggest that bottlenose dolphins in Hawaiian waters belong to a separate stock from those in the ETP. Furthermore, recent photo-identification and genetic studies of bottlenose dolphins sampled near each of the main Hawaiian Islands suggest limited movement of bottlenose dolphins between islands and offshore waters (Baird *et al.* 2009; Martien *et al.* 2012, Harnish 2021). These data support the existence of demographically independent resident populations at each of four main Hawaiian Island groupings – Kaua'i and Ni'ihau, O'ahu, the Maui Nui region (Moloka'i, Lāna'i, Maui, Kaho'olawe), and Hawai'i Island. Genetic data support inclusion of bottlenose dolphins in deeper waters surrounding the main Hawaiian Islands as part of the broadly distributed pelagic population (Martien *et al.* 2012). Over 99% of the bottlenose dolphins linked through photo-identification to one of the insular population around the main Hawaiian Islands (Baird *et al.* 2009) have been documented in waters of 1000 m or less (Martien and Baird 2009). Based on these data, Martien and Baird (2009) suggested that the boundaries between the insular stocks and the Hawai'i Pelagic stock be placed along the 1000 m isobath. Since that isobath does not

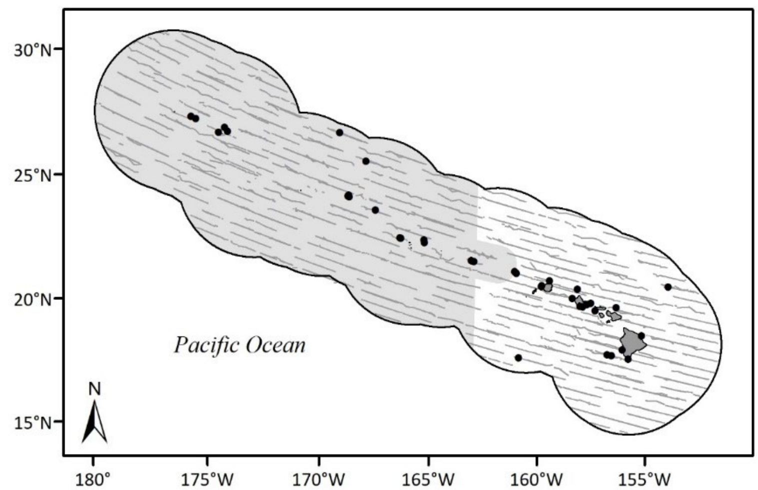



**Figure 1.** Bottlenose dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010(Bradford *et al*. 2017), and 2017 (Yano *et al.* 2018 shipboard cetacean surveys of the U.S. EEZ around the Hawaiian Islands outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray. Insular stock boundaries are shown in Figure 2.

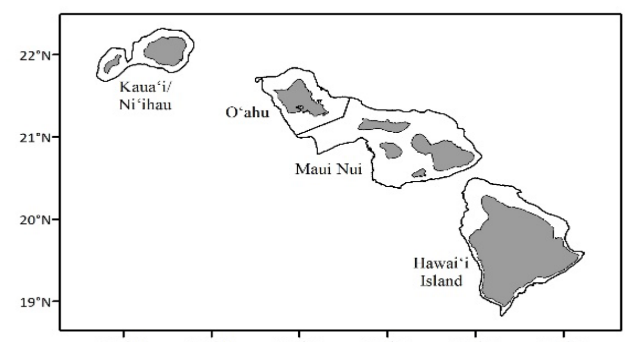



**Figure 2**. Main Hawaiian Islands insular bottlenose dolphin stock boundaries (gray lines).

separate Oahu from the Maui Nui Region, the boundary between those stocks runs approximately equidistant between the 500 m isobaths around Oʻahu and the Maui Nui Region, through the middle of Kaiwi Channel. These boundaries (Figure 2) are applied in this report to recognize separate insular and pelagic bottlenose dolphin stocks for management (NMFS 2023a). These boundaries may be revised in the future as additional information becomes available. To date, no data are available regarding population structure of bottlenose dolphins in the Northwestern Hawaiian Islands (NWHI), though sightings during a shipboard survey in 2010 indicate they are commonly found close to the islands and atolls there (Bradford *et al.* 2017). Given the evidence for island resident populations in the main Hawaiian Islands, the larger distances between islands in the NWHI, and the finding of population structure within the NWHI in other dolphin species (e.g., Andrews *et al.* 2010), it is likely that additional demographically independent populations of bottlenose dolphins exist in the NWHI. However, until data become available upon which to base stock designations in this area, bottlenose dolphins in the NWHI will remain part of the Hawaiʻi Pelagic Stock.

For the Marine Mammal Protection Act (MMPA) Pacific stock assessment reports, bottlenose dolphins within the Pacific U.S. Exclusive Economic Zone (EEZ) are divided into seven stocks: 1) California, Oregon and Washington offshore stock, 2) California coastal stock, and five Pacific Islands Region management stocks (this report): 3) Kauaʻi/Niihau, 4) Oʻahu, 5) Maui Nui (Molokaʻi, Lānaʻi, Maui, Kahoʻolawe), 6) Hawaiʻi Island and 7) the Hawaiʻi Pelagic Stock, including animals found both within the U.S. EEZ around the Hawaiian Islands and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaiʻi Pelagic stock is evaluated based on data from U.S. EEZ of the Hawaiian Islands (NMFS 2023a). Estimates of abundance, potential biological removals, and status determinations for the five Hawaiian stocks are presented separately below.

## HUMAN CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiʻi fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. There are at least two reports of entangled bottlenose dolphins dying in gillnets off Maui (Nitta and Henderson 1993, Maldini 2003, Bradford and Lyman 2013). Although gillnet fisheries are not observed or monitored through any State or Federal program, State regulations now ban gillnetting around Maui and much of Oʻahu and require gillnet fishermen to monitor their nets for bycatch every 30 minutes in those areas where gillnetting is permitted. In 2018, a bottlenose dolphin calf was observed with a gunshot wound through its melon, possibly as a result of a fisheries interaction (Harnish *et al.* 2019). Although the wound was initially judged to be serious, sightings of this animal since the injury was initially observed have indicated the wound is healing and the animal has survived (Harnish *et al.* 2019), such that the injury was ultimately determined to be non-serious (Bradford and Lyman 2020) under criteria for assessing serious injury in marine mammals (NMFS 2023b). In 2019, this same individual was observed hooked in the mouth and entangled around its pectoral fin by the trailing line, also initially judged to be a serious injury (Bradford and Lyman 2022). However, based on the observations in 2021 of the animal in good body condition, the injury is currently considered to be non-serious (Bradford and Lyman 2022). In 2020, an adult bottlenose dolphin was found dead as a result of an ingested circle hook piercing its esophagus, with the hook and attached monofilament line attributed to a nearshore fishery (Bradford and Lyman 2023). This recent mortality indicates that nearshore fisheries still pose a risk to bottlenose dolphins around the Hawaiian Islands. However, no estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

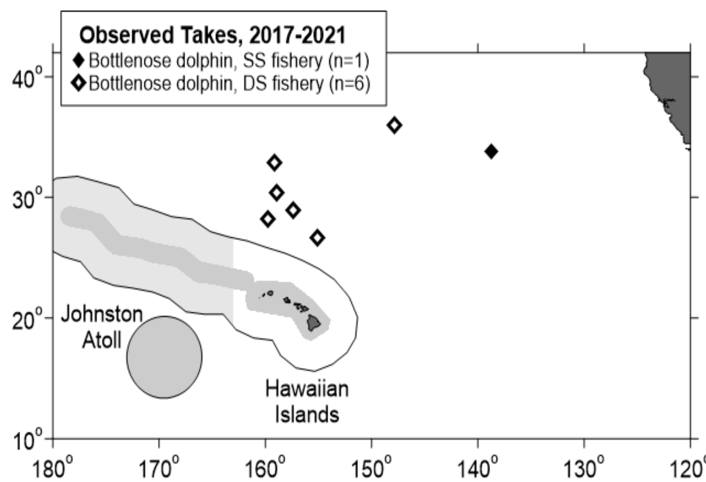



**Figure 3.** Locations of observed Pelagic Stock bottlenose dolphin takes within the shallow-set fishery (filled diamond) and deep-set fishery (open diamonds) in the Hawaii-based longline fishery, 2017-2021. Solid lines represent the U. S. EEZ. Gray shading notes areas closed to longline fishing.

Bottlenose dolphins are one of the species commonly reported to steal bait and catch from several Hawaiʻi sport and commercial fisheries (Nitta and Henderson 1993, Schlais 1984). Observations of bottlenose dolphins stealing bait or catch have been made in the day handline fishery for tuna (palu-ahi), the night handline fishery for tuna (ika-shibi), the handline fishery for mackerel scad, the troll fishery for billfish and tuna, and the inshore set gillnet fishery (Nitta and Henderson 1993). Nitta and Henderson (1993) indicated that bottlenose dolphins remove bait and catch from handlines used to catch bottomfish off the island of Hawaiʻi and Kaula Rock and formerly on several banks of the Northwestern Hawaiian Islands. Bottlenose dolphins were thought to interact with the bottomfish fishery in the NWHI (Kobayashi and Kawamoto 1995), though this fishery is no longer permitted for the NWHI. Fishermen around the main Hawaiian Islands claim interactions with dolphins that steal bait and catch are increasing, including anecdotal reports of bottlenose dolphins getting "snagged" (Rizzuto 2007). An assessment of the incidence of potential fishing gear-associated scarring on bottlenose dolphins near Maui Nui revealed 27% of non-calf well-marked individuals photographed between 1996 and 2020 had one or more scars that may be attributed to fishing gear (Machernis *et al.* 2021).

There are currently two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the NWHI. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163˚ W. Between 2017 and 2021, one bottlenose dolphin was observed hooked or entangled in the SSLL fishery (100% observer coverage), and six bottlenose dolphins were observed taken in the DSLL fishery (15-21% observer coverage) within the Hawaiian Islands EEZ or adjacent high-seas waters (Bradford 2018, 2020, 2021, 2023, in review). Based on the observed take locations (Figure 3), these takes are all considered to have been from the Pelagic Stock of bottlenose dolphins. All 7 dolphins were considered to have been seriously injured, based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2023b).

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the U.S. EEZ), and the ratio of observed dead and seriously injured dolphins versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for the Pelagic Stock during 2017-2021 are 6.6 (CV=0.4) bottlenose dolphins outside of the Hawaiian Islands EEZ, and 0 within the Hawaiian Islands EEZ (Table 1, McCracken and Cooper 2022b). One unidentified cetacean, likely to be a bottlenose dolphin based on the observer's description, was taken in the DSLL fishery in 2017 (Bradford 2018),

**Table 1.** Summary of available information on incidental mortality and serious injury (MSI) of bottlenose dolphins (Hawaiʻi Pelagic stock) in commercial longline fisheries, within and outside of the U.S. EEZ (McCracken and Cooper 2022b). Mean annual takes are based on 2017-2021 data unless otherwise indicated. Information on all observed takes (T) and MSI is included along with MSI estimates. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Outside U.S. EEZ | | Hawaiian Islands EEZ | |
|---|---|---|---|---|---|---|---|
| | | | | Observed T/MSI | Estimated MSI (CV) | Observed T/MSI | Estimated MSI (CV) |
| **Hawaiʻi-based deep-set longline fishery** | 2017 | Observer data | 20% | 1/1 | 6 (0.9) | 0 | 0 (-) |
| | 2018 | | 18% | 1/1 | 3 (0.9) | 0 | 0 (-) |
| | 2019 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2020 | | 15% | 1/1 | 10 (0.6) | 0 | 0 (-) |
| | 2021 | | 18% | 3/3 | 9 (0.6) | 0 | 0 (-) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | **5.6 (0.4)** | 0 | **0 (-)** |
| **Hawaiʻi-based shallow-set longline fishery** | 2017 | | 100% | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 1/1 | 1 | 0 | 0 |
| | 2019 | | 100% | 0 | 0 | 0 | 0 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| | 2020 | | 100% | 0 | 0 | 0 | 0 |
| | 2021 | | 100% | 0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | | **1** | | **0** |
| **Minimum total annual takes within U.S. EEZ (2017-2021)** | | | | | | | **0 (-)** |

## KAUA‘I / NI‘IHAU STOCK

### POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 were used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the Kaua‘i/Niihau stock of bottlenose dolphins were produced for 2003 through 2007 and 2011 through 2018. The 2018 abundance estimate for the Kaua‘i/Ni‘ihau stock was 112 (CV=0.24) bottlenose dolphins.

### Minimum Population Estimate

The minimum population estimate for the Kaua‘i/Ni‘ihau stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 92 bottlenose dolphins.

### Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) suggest that the Kaua‘i/Ni‘ihau stock of bottlenose dolphins may have declined over the nearly 20-year period of the study, with a high of 193 (CV=0.25) dolphins in 2003 to the low of 112 (CV=0.24) in 2018, representing an overall average annual decline of 2.6% (95% CI -6.9% to -1.7%). However, the annual estimates did not differ significantly throughout the study period and varied only by a few individuals between 2011 and 2018, such that the trends are not considered reliable (Van Cise *et al.* 2021). Further, while survey effort was most consistent for the Kaua‘i/Ni‘ihau stock, sampling variability was not fully accounted for in the estimates of abundance and trend.

### CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

### POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (92) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality or serious injury within the Kaua‘i/Ni‘ihau stock range; Wade and Angliss 1997), resulting in a PBR of 0.9 bottlenose dolphins per year.

### STATUS OF STOCK

The Kaua‘i/Ni‘ihau Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in the Kaua‘i/Ni‘ihau stock relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion. Bottlenose dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for Kaua‘i/Ni‘ihau bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. One stranded bottlenose dolphin from the Kaua‘i/Ni‘ihau stock tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob *et al.* 2016), raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

## O‘AHU STOCK

### POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the O‘ahu stock of bottlenose dolphins were produced for 2002 through 2018, except for 2005. The 2018 abundance estimate for the O‘ahu stock was 112 (CV=0.17) bottlenose dolphins.

#### Minimum Population Estimate

The minimum population estimate for the O‘ahu stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 97 bottlenose dolphins.

#### Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) suggest that the O‘ahu stock of bottlenose dolphins may have declined over the nearly 20-year period of the study, with a high of 193 (CV=0.31) dolphins in 2002 to the low of 112 (CV=0.17) in 2018, representing an overall average annual decline of 3% (95% CI -10.3% to +2.7%). However, the annual estimates did not differ significantly throughout the study period and varied by only a few individuals over the last half of the study period, such that the trends are not considered reliable (Van Cise *et al*. 2021). Similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend, but particularly for the O‘ahu stock, it is possible that the apparent decline is an artifact of increased citizen science contributions in one subarea and contracted survey effort over the study period.

### CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

### POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (97) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the stock range (Wade and Angliss 1997), resulting in a PBR of 1.0 bottlenose dolphins per year.

### STATUS OF STOCK

The O‘ahu stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in O‘ahu waters relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion. Bottlenose dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries, though there is evidence of a bottlenose dolphin that was shot in the head off O‘ahu (Harnish *et al.* 2019) that later became hooked and entangled in fishing gear (Bradford and Lyman 2022). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for O‘ahu bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

## MAUI NUI STOCK

### POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the Maui Nui stock of bottlenose dolphins were produced for all years except 2008.

The 2018 abundance estimate for the Maui Nui stock was 64 (CV=0.15) bottlenose dolphins.

### Minimum Population Estimate

The minimum population estimate for the Maui Nui stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 56 bottlenose dolphins.

### Current Population Trend

Annual abundance estimates derived in Van Cise *et al.* (2021) suggest that the Maui Nui stock of bottlenose dolphins has declined over the nearly 20-year period of the study, with a high of 288 (CV=0.17) dolphins in 2000 to the low of 64 (CV=0.15) in 2018, representing an overall average annual decline of 8.6% (95% CI -13% to -6%). While the analysis suggests a statistically significant decline in this stock (Van Cise *et al*. 2021), similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend. Particularly for the Maui Nui stock, it is possible that the apparent decline is an artifact of increased citizen science contributions in one subarea and contracted survey effort over the study period.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (56) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the Maui Nui stock area (Wade and Angliss 1997), resulting in a PBR of 0.6 bottlenose dolphins per year.

## STATUS OF STOCK

The Maui Nui Region Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in Maui Nui waters relative to OSP is unknown. Although recent analyses suggest this stock may be declining (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion. Bottlenose dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries of this stock; however, there is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for Maui Nui bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

# HAWAI‘I ISLAND STOCK

## POPULATION SIZE

Photographic data from multiple contributors spanning 2000 to 2018 was used to assess the annual abundance of each main Hawaiian Islands insular population of bottlenose dolphins using a POPAN model stratified within each stock area based on spatial gaps in sightings and significant bathymetric or geographic features (Van Cise *et al.* 2021). Annual abundance estimates for the Hawai‘i Island stock of bottlenose dolphins were produced for all years from 2002 to 2018. The 2018 abundance estimate for the Hawai‘i Island stock was 136 (CV=0.43) bottlenose dolphins.

### Minimum Population Estimate

The minimum population estimate for the Hawai‘i Island stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2018 abundance estimate (from Van Cise *et al.* 2021), or 96 bottlenose dolphins.

### Current Population Trend

Annual abundance estimates derived in Van Cise et al. (2021) suggest that the Hawai‘i Island stock of bottlenose dolphins has increased over the nearly 20-year period of the study, with a low of 10 (CV=0.17) in 2000 to the high of 136 (CV=0.43) in 2018, representing an overall average annual increase of 10.5% (95% CI 0.94% to 15.31%). This estimated annual growth rate is greater than the species maximum expected growth rate of 4% and was driven largely by influxes of new individuals during the study period. Similar to other stocks, sampling variability was not fully accounted for in the estimates of abundance and trend, but particularly for the Hawai‘i Island stock, the abundance estimates likely underestimate true stock size because sampling for this stock was entirely on the leeward side of Hawai‘i Island (Van Cias *et al*. 2021). Thus, the increasing trend may be an artifact of variability in sampling and individual habitat use.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size (96) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no reported fishery mortality in the Hawai‘i Islands stock area (Wade and Angliss 1997), resulting in a PBR of 1.0 bottlenose dolphins per year.

## STATUS OF STOCK

The Hawai‘i Island stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in waters around Hawai‘i Island relative to OSP is unknown. Although recent analyses suggest this stock may be increasing (Van Cise *et al.* 2021), sampling limitations increase uncertainty around this conclusion. Bottlenose dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. In the past 5 years, one bottlenose dolphin was found dead on Hawai‘i Island as a result of an ingested circle hook piercing its esophagus (Bradford and Lyman 2023). There is no systematic monitoring of takes in nearshore fisheries that may take this species, thus the single observed mortality may be an underestimate of the total fishery mortality for this stock. Total fishery mortality and serious injury for Hawai‘i Island bottlenose dolphins is not approaching zero mortality and serious injury rate. Hawai’i Island bottlenose dolphins are regularly seen near aquaculture pens off the Kona coast, and aquaculture workers have been observed feeding bottlenose dolphins. Bottlenose dolphins in this region are also known to interact with divers. Since 2007, about one quarter (36) of Hawai‘i Islands bottlenose dolphins have been observed associated with a pelagic mariculture operation for kanpachi off the Kona coast of Hawai‘i Island, with 22 of those individuals seen at the farm on more than one occasion (Harnish *et al.* 2023). Farm-associated dolphins are weakly linked to the rest of the Hawai‘i Island population, and are seen in smaller groups near the farm than those groups seen away from the farm, factors that have been linked to lower survival in other populations (Stanton and Mann, 2012). *Morbillivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

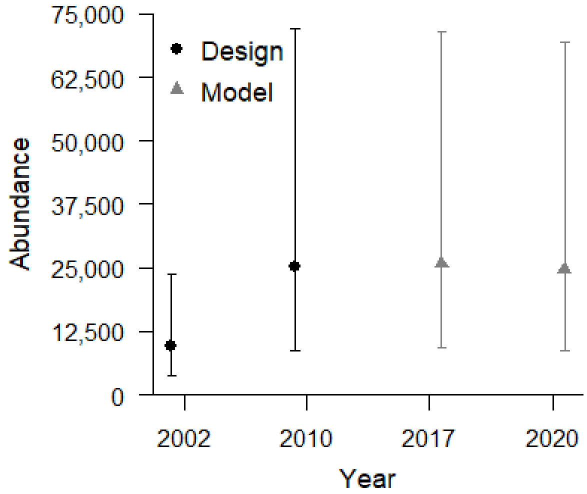
# HAWAI’I PELAGIC STOCK

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of bottlenose dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2022, Bradford *et al.* 2021; Table 2).

Table 2. Line-transect abundance estimates for bottlenose dolphins in the Hawaiian Islands EEZ in 2002, 2010, 2017, and 2020, derived from NMFS surveys in the central Pacific since 1986 (Becker *et al.* 2022, Bradford *et al.* 2021).

| Year | Design-based Abundance | CV | 95% Confidence Limits | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|---|---|---|
| 2020 | - | - | - | 24,669 | 0.57 | 8,774-69,361 |
| 2017 | - | - | - | 25,857 | 0.56 | 9,356-71,464 |
| 2010 | 25,188 | 0.58 | 8,791-72,168 | - | - | - |
| 2002 | 9,678 | 0.49 | 3,924-23,868 | - | - | - |




Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2022) estimates of abundance for Hawai‘i pelagic bottlenose dolphins for each survey year (2002, 2010, 2017, 2020).

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive stock-specific habitat-based models of animal density for the 2017 to 2020 period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al*. 2015, Becker *et al.* 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for bottlenose dolphins from Barlow *et al.* (2015). Although model-based estimates were previously derived for years 2002, 2010, and 2017 (Becker *et al.* 2021), those estimates were not specific to the Hawai‘i Pelagic stock and as such may have reflected both the habitat associations and abundance of the insular stocks within the main Hawaiian Islands. Stock-specific model-based estimates were derived only for the most recent years (2017-2020), such that direct comparison of model and design-based estimates for the full survey time series is not possible at this time. Bradford *et al.* (2021) produced design-based abundance estimates for bottlenose dolphins for each full EEZ survey year with bottlenose dolphin encounters, with the 2010 design-based and 2017 and 2020 model-based estimates largely similar in the mean estimate and confidence limits (Figure 4). Current model based-estimates are based on the implicit assumption that annual changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for the most recent survey year. Previously published design-based estimates for the Hawaiian Islands EEZ from 2002 and 2010 surveys (Barlow 2006, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 24,669 (CV=0.57) bottlenose dolphins.

### Minimum Population Estimate

The minimum population estimate for the Hawai‘i Pelagic stock of bottlenose dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate (from Becker *et al.* 2022), or 15,783 bottlenose dolphins.

### Current Population Trend

The available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (15,783) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with a Hawaiian Islands EEZ fishery mortality and serious injury rate CV of 0; Wade and Angliss 1997), resulting in a PBR of 158 bottlenose dolphins per year.

## STATUS OF STOCK

The Hawai‘i Pelagic Stock of bottlenose dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of bottlenose dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. It is not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. The estimated rate of fisheries related mortality or serious injury within the Hawaiian Islands EEZ is zero, such that the total fishery mortality and serious injury for Hawaiʻi Pelagic bottlenose dolphins is insignificant and approaching zero mortality and serious injury rate. *Morbilivirus* has been detected within insular stocks of bottlenose dolphins in Hawaiʻi (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawaiʻi and the potential population impacts, including the cumulative impacts of disease with other stressors.

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# PANTROPICAL SPOTTED DOLPHIN (*Stenella attenuata attenuata*): Hawaiian Islands Stock Complex – Oʻahu, Maui Nui, Hawaiʻi Island, and Hawaiʻi Pelagic Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Pantropical spotted dolphins are primarily found in tropical and subtropical waters worldwide (Perrin *et al.* 2009). Much of what is known about the species in the North Pacific has been learned from specimens obtained in the large directed fishery in Japan and in the eastern tropical Pacific (ETP) tuna purse-seine fishery (Perrin *et al*. 2009). Spotted dolphins are common and abundant throughout the Hawaiian Islands, including nearshore where they are the second most frequently sighted species during nearshore surveys (Baird *et al*. 2013) and offshore where they are frequently observed during periodic shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands (Figure 1). Morphological differences and distribution patterns indicate that the spotted dolphins around the Hawaiian Islands belong to a stock that is distinct from those in the ETP (Perrin 1975; Dizon *et al.* 1994; Perrin *et al*. 1994b).

Pantropical spotted dolphins have been observed in all months of the year around the main Hawaiian Islands, and in areas ranging from shallow nearshore waters to depths of 5,000 m, although they peak in sighting rates in depths from 1,500 to 3,500 m (Baird *et al.* 2013). Although they represent from 22.9 to 26.5% of the odontocete sightings from Oʻahu, the Maui Nui (Molokaʻi, Lānaʻi, Maui, Kahoʻolawe), and Hawaiʻi Island, they are largely absent from the nearshore waters around Kauaʻi and Niʻihau, representing only 3.9% of sightings in that area (Baird *et al.* 2013). Genetic analyses of 176 unique samples of pantropical spotted dolphins collected during nearshore surveys off each of the main Hawaiian Islands from 2002 to 2003, and near Hawaiʻi Island from 2005 to 2008, suggest three island-associated stocks

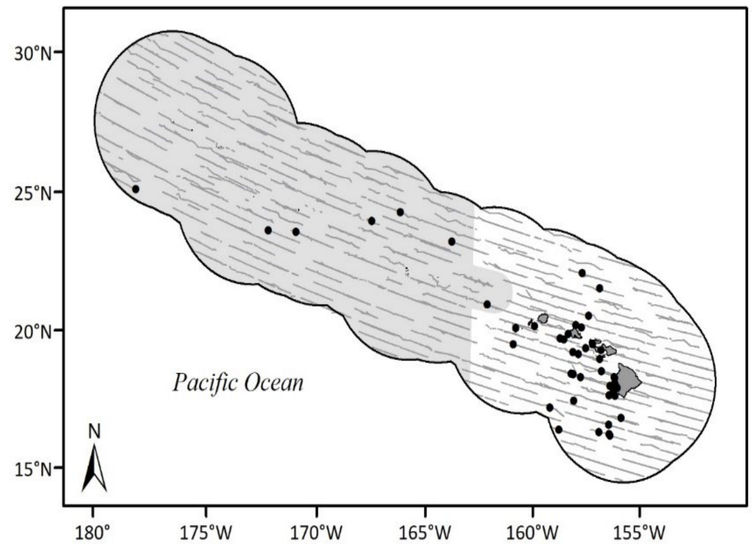


**Figure 1.** Pantropical spotted dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al.* 2018) shipboard surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray. Insular stock boundaries are shown in Figure 2.

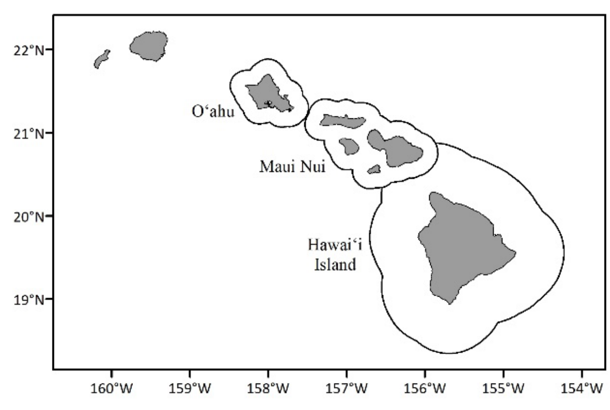


**Figure 2**. Main Hawaiian Islands insular stock boundaries (gray lines). Areas beyond the insular boundaries represent the pelagic stock range.

are evident (Courbis *et al*. 2014). The results of the Courbis *et al*. (2014) study indicate that pantropical spotted dolphins in Hawaiʻi's nearshore waters have low haplotypic diversity with haplotypes unique to each of the island areas. Courbis *et al*. (2014) conducted extensive tests on the relatedness of individuals among islands using the microsatellite dataset and found significant differences in haplotype frequencies between islands, suggesting genetic differentiation in spotted dolphins among islands. This suggestion is supported by the results of assignments tests, which indicate support for 3 island-associated populations: Hawaiʻi Island, the Mui Nui region, and Oʻahu. Samples from Kauaʻi and Niʻihau did not cluster together, but instead were spread among the Hawaiʻi and Oʻahu clusters. Analysis of migration rate further support the separation of pantropical spotted dolphins into three island-associated stocks, with migration between regions on the order of a few individuals per generation. Based on an overview of all available information on pantropical spotted dolphins in Hawaiian waters, and NMFS guidelines for assessing marine mammal stocks (NMFS 2023), Oleson *et al.* (2013) proposed designation of three new island associated stocks in Hawaiian waters, as well as recognition of a fourth broadly distributed spotted dolphin stock given the frequency of sightings in pelagic waters. Stock boundaries for main Hawaiian Islands spotted dolphin stocks are based on the furthest distance from shore of an insular sighting. Around Oʻahu and Maui Nui the stock extends to 20km form shore and around Hawaiʻi Island to 65km. Fishery interactions with pantropical spotted dolphins and sightings near Palmyra and Johnston Atolls (NMFS PIRO unpublished data) demonstrate that this species also occurs in the U.S. EEZ of those locations, but it is not known whether these animals are part of the Hawaiʻi Pelagic population or are a separate stock or stocks of pantropical spotted dolphins.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are four Pacific management stocks within the Hawaiian Islands EEZ (Oleson *et al.* 2013): 1) the Oʻahu stock, which includes spotted dolphins within 20 km of Oʻahu, 2) the Maui Nui stock, which includes spotted dolphins within 20 km of Maui, Molokaʻi, Lānaʻi, and Kahoʻolawe, collectively, 3) the Hawaiʻi Island stock, which includes spotted dolphins found within 65 km from Hawaiʻi Island, and 4) the Hawaiʻi Pelagic stock, which includes spotted dolphins inhabiting the waters throughout the Hawaiian Islands EEZ, outside of the insular stock areas, but including adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaiʻi pelagic stock is evaluated based on data from the U.S. EEZ around the Hawaiian Islands (NMFS 2023). Spotted dolphins involved in eastern tropical Pacific tuna purse-seine fisheries are managed separately under the MMPA.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaiʻi (Nitta and Henderson 1993). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch. Commercial and recreational troll fisherman have been observed "fishing" dolphins off the islands of Hawaiʻi, Lānaʻi, and Oʻahu, including spotted dolphins, in order to catch tuna associated with the animals (Courbis *et al*. 2009, Rizzuto 2007, Shallenberger 1981). Anecdotal reports from fisherman indicate that spotted dolphins are sometimes hooked (Rizzuto 1997). An assessment of the incidence of potential fishing gear-associated scarring on pantropical spotted dolphins near Maui Nui revealed 13% of non-calf well-marked individuals photographed between 1996 and 2020 had one or more scars that may be attributed to fishing gear (Machernis *et al.* 2021). A study of the incidence of fishing vessels associated with spotted dolphins revealed that hundreds of boats appear to be engaging in this fishing method, including a high incidence of trolling through the group of dolphins or maneuvering through the group to drop hook and line gear ahead of the dolphin group (Baird and Webster 2020). In 2017, a spotted dolphin (Maui Nui stock) was seen near Lānaʻi with a band of debris around its rostrum preventing it from opening its mouth, which was determined to be a serious injury (Bradford and Lyman 2019). Serious injuries from nearshore fishing gear have previously been observed in other insular stocks (Bradford and Lyman 2018).

There are currently two distinct longline fisheries based in Hawaiʻi: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017 and 2021, no pantropical spotted dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage) or in the DSLL fishery (15-21% observer coverage) (McCracken and Cooper 2022).

## O‘AHU STOCK

## POPULATION SIZE

The population size of the O‘ahu stock of pantropical spotted dolphins has not been estimated. Model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

### Minimum Population Estimate

There is no information on which to base a minimum population estimate of the O‘ahu stock of pantropical spotted dolphins.

### Current Population Trend

No data are available on current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the O‘ahu stock of pantropical spotted dolphins is calculated as the minimum population estimate <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the O‘ahu stock area; Wade and Angliss 1997). Because there is no minimum population estimate available, the PBR for O‘ahu stock of pantropical spotted dolphins is undetermined.

## STATUS OF STOCK

The O‘ahu stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of O‘ahu spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries, though fishermen may target groups of spotted dolphins around O‘ahu in order to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. *Morbillivirus* has been detected within other insular stocks of pantropical spotted dolphins in Hawai‘i (Jacob *et al*. 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

## MAUI NUI STOCK

## POPULATION SIZE

The population size of Maui Nui stock of pantropical spotted dolphins has not been estimated. Model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

### Minimum Population Estimate

There is no information on which to base a minimum population estimate of the Maui Nui stock of pantropical spotted dolphins.

### Current Population Trend

No data are available on current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Mui Nui stock of pantropical spotted dolphins is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the Mui Nui stock area; Wade and Angliss 1997). Because there is no minimum population estimate available for this stock, the PBR for 4-Islands stock of pantropical spotted dolphins is undetermined.

## STATUS OF STOCK

The Maui Nui stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of Maui Nui spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Aside from the spotted dolphin entangled in marine debris in 2017 (Bradford and Lyman 2018), there are no other reports of recent mortality or serious injuries, though fishermen may target groups of spotted dolphins around the Maui Nui region to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020), with injuries potentially associated with fishing line observed in a portion of well-marked animals (Machernis *et al.* 2021). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. *Morbillivirus* has been detected within other insular stocks of pantropical spotted dolphins in Hawaiʻi (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawaiʻi and the potential population impacts, including the cumulative impacts of disease with other stressors.

## HAWAIʻI ISLAND STOCK

## POPULATION SIZE

The population size of the Hawaiʻi Island stock of pantropical spotted dolphins has not been estimated. Design and model-based estimates using line-transect datasets have been explored for this stock (Becker *et al.* 2022, Bradford *et al.* 2022), though the small sample size and an uneven distribution of survey effort resulted in unreliable estimates.

### Minimum Population Estimate

There is no information on which to base a minimum population estimate of the Hawaiʻi Island stock of pantropical spotted dolphins.

### Current Population Trend

No data are available on current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii Island stock of pantropical spotted dolphins is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the Hawaii Island stock area; Wade and Angliss 1997). Because there is no minimum population estimate available for this stock, the PBR for Hawaii Island stock of pantropical spotted dolphins is undetermined.

## STATUS OF STOCK

The Hawaiʻi Island stock of pantropical spotted dolphins is not considered a strategic stock under the MMPA. The status of Hawaiʻi Island spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spotted dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries, though fishermen target groups of spotted dolphins around Hawaiʻi Island to catch associated tuna, increasing the likelihood of dolphins being hooked or entangled (Baird and Webster 2020). There is no systematic monitoring for interactions with protected species within nearshore fisheries that may take this species, thus mean annual takes are undetermined. Insufficient information is available to determine whether the total fishery mortality and serious injury for this stock is insignificant and approaching zero mortality and serious injury rate. One spotted dolphin found stranded on Hawaiʻi Island has tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012) raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including the cumulative impacts of disease with other stressors.

## HAWAIʻI PELAGIC STOCK

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of pantropical spotted dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2022, Bradford *et al.* 2021; Table 1).

**Table 1.** Line-transect abundance estimates for Hawaiʻi pelagic pantropical spotted dolphins in the Hawaiian Islands EEZ in 2002, 2010, 2017, and 2020, derived from NMFS surveys in the central Pacific since 1986 (Becker *et al.* 2022, Bradford *et al.* 2021).

| Year | Design-based Abundance | CV | 95% Confidence Limits | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|---|---|---|
| 2020 | | | | 67,313 | 0.27 | 40,096-113,005 |
| 2017 | 39,798 | 0.51 | 15,432-102,637 | 73,667 | 0.28 | 42,769-126,886 |
| 2010 | 49,488 | 0.39 | 23,551-103,992 | | | |
| 2002 | 16,931 | 0.65 | 5,289-54,202 | | | |

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive stock-specific habitat-based models of animal density for the 2017 to 2020 period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for spotted dolphins from Barlow *et al.* (2015). Although model-based estimates were previously derived for years 2002, 2010, and 2017 (Becker *et al.* 2021), those estimates were not pelagic stock specific and as such may have reflected both the habitat associations and abundance of the insular stocks within the main Hawaiian Islands. Stock-specific model-based estimates were derived only for the most recent years (2017-2020), such that direct comparison of model and design-based estimates for the full survey time series is not possible at this time. Bradford *et al.* (2021) produced design-based abundance estimates for spotted dolphins for each full EEZ survey year. Design-based estimates for spotted dolphins are generally lower than model-based estimates, though the confidence limits broadly overlap with those produced from the model-based approach (Figure 3). Current model based-estimates are based on the implicit assumption that annual changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for the most recent survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of spotted dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020 and found no significant difference, with no reliance on season within the model-based approach and largely similar design-based estimates for summer-fall 2017 versus winter 2020. Previously published design-based estimates for the Hawaiian Islands EEZ from 2002 and 2010 surveys (Barlow 2006, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021, 2022) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 67,313 (CV=0.27) pantropical spotted dolphins.

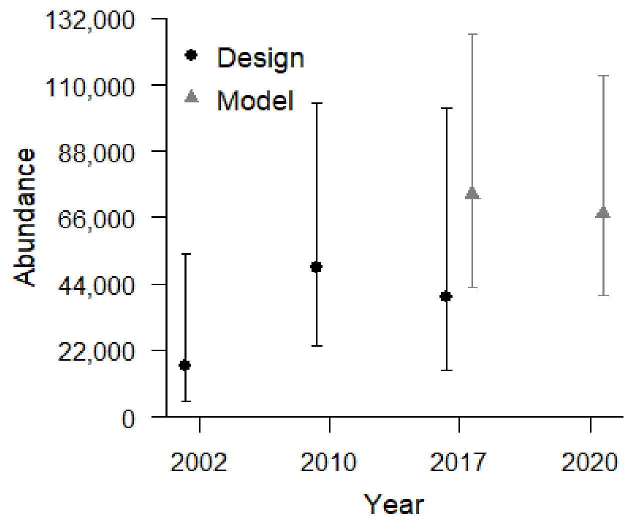


**Figure 3.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for hawai‘i pelagic pantropical spotted dolphins for each survey year (2002, 2010, 2017, 2020).

Population estimates are available for Japanese waters (Miyashita 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

### Minimum Population Estimate

The minimum population estimate for the Hawai‘i pelagic stock of pantropical spotted dolphins is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate for the pelagic stock area (from Becker *et al.* 2022), or 53,839 pantropical spotted dolphins.

### Current Population Trend

The available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawai‘i pelagic stock of pantropical spotted dolphins is calculated as the minimum population estimate within the U.S. EEZ of the Hawaiian Islands (53,839) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997), resulting in a PBR of 538 Hawai‘i pelagic pantropical spotted dolphins per year.

## STATUS OF STOCK

The Hawai‘i pelagic stock of spotted dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of Hawai‘i pelagic pantropical spotted dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Pantropical spotted dolphins are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within U.S. EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. *Morbillivirus* has been detected within other insular stocks of bottlenose dolphins in Hawai‘i (Jacob *et al.* 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters raises concerns about the history and prevalence of this disease in Hawai‘i and the potential population impacts, including the cumulative impacts of disease with other stressors.

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Revised 3/1/2019

**SPINNER DOLPHIN (Stenella longirostris longirostris):
Hawaiian Islands Stock Complex- Hawaii Island, Oahu/4-islands,
Kauai/Niihau, Pearl & Hermes Reef, Midway Atoll/Kure, Hawaii Pelagic**

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Six morphotypes within four subspecies of spinner dolphins have been described worldwide in tropical and warm-temperate waters (Perrin *et al.* 2009). The Gray's (or pantropical) spinner dolphin (*Stenella longirostris longirostris*) is the most widely distributed subspecies and is found in the Atlantic, Indian, central and western Pacific Oceans (Perrin *et al.* 1991). Spinner dolphins in Hawaii belong to this sub-species. Unlike Gray's spinner dolphins in the eastern tropical Pacific (ETP), which are commonly found in pelagic waters, spinner dolphins in Hawaii are island-associated and use shallow protected bays to rest and socialize during the day then move offshore at night to feed (Norris and Dohl 1980; Norris *et al.* 1994). Spinner dolphins in Hawaii are considered separate stocks from those in the ETP (Perrin 1975; Dizon *et al.* 1994). Andrews *et al.* (2010) found that mtDNA control region haplotype and nucleotide diversities of Hawaiian spinner dolphins are low compared with those from other geographic regions and suggested the existence of strong barriers to gene flow, both geographic and ecological. These analyses also reveal significant genetic distinction, at both mtDNA and microsatellite loci, between spinner dolphins sampled in American Sāmoa and those sampled in the Hawaiian Islands (Johnston *et al.* 2008, Andrews *et al.* 2010).

Most spinner dolphin research in Hawaii occurs in nearshore waters surrounding the main Hawaiian Islands and at Midway and Kure Atoll in the northwestern Hawaiian Islands (e.g. Norris *et al.* 1994, Karczmarski *et al.* 2005, Tyne *et al.* 2017). Spinner dolphins are rare in pelagic waters in the Hawaiian Archipelago, and have been infrequently seen during large-scale line-transect surveys. Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in 8 sightings in 2002 and 2 sightings in 2010, though none of the 2010 sightings occurred during on-effort survey (Barlow 2006, Bradford *et al.* 2017; Fig. 1).

The population structure of spinner dolphins in Hawaii has been assessed using

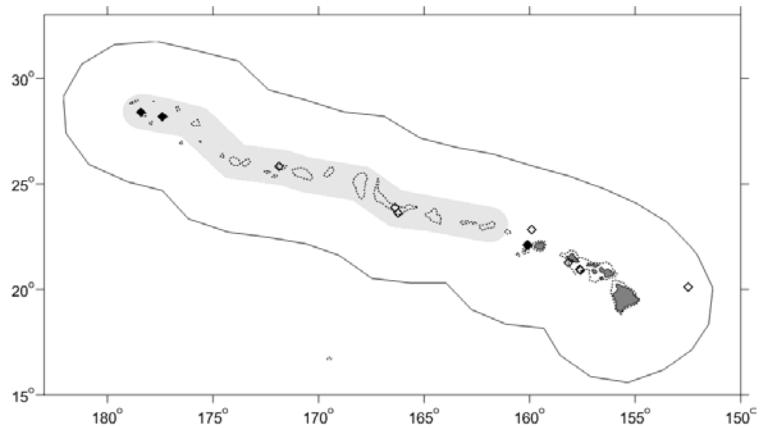


**Figure 1.** Spinner dolphin sighting locations during the 2002 (open diamonds) and 2010 (black diamonds) shipboard cetacean surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017; see Appendix 2 for details on timing and location of survey effort). Outer line indicates approximate boundary of survey area and U.S. EEZ. Gray shading indicates area of Papahanaumokuakea Marine National Monument. Dotted line represents the 1000 m isobath. Insular stock boundaries are shown in Figure 2.

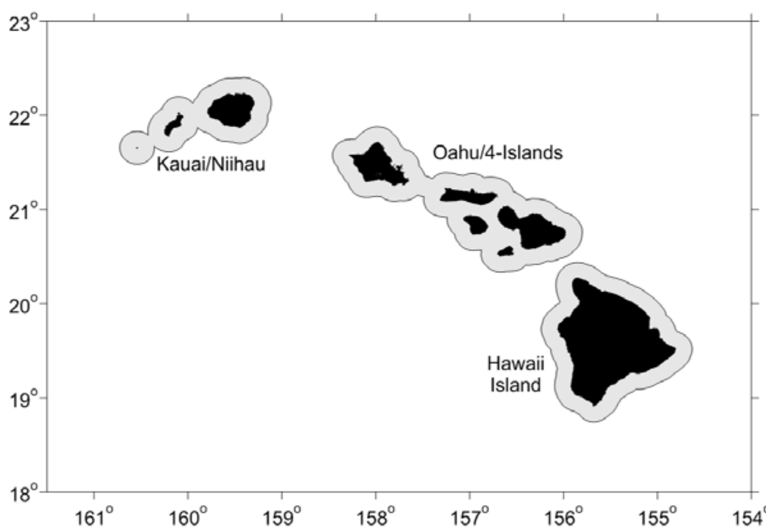


**Figure 2**. Spinner dolphin stock boundaries in the main Hawaiian Islands (Midway/Kure and Pearl and Hermes stock ranges not shown). Animals outside of the defined island areas are considered to be part of the Hawaii pelagic stock.

genetic and movement data. Andrews *et al.* (2010) found significant genetic distinctions between spinner dolphins sampled at different islands within the Hawaiian Archipelago. Most significant was differentiation between animals sampled off the Kona Coast of Hawaii Island and animals sampled at all other Hawaiian Islands. Similarly, in the Northwestern Hawaiian Islands, spinner dolphins sampled at Midway and Kure were shown not to be genetically distinct from each other, but are distinct from those sampled at all other islands. Andrews (2009) found that none of the pairwise comparisons between French Frigate Shoals, Niihau, Kauai, and Oahu were statistically significant and Oahu was not significantly differentiated from Maui/Lanai. Assignment tests, which may provide information about recent gene flow, show that for most islands and atolls within the Hawaiian Archipelago, more samples assigned to the island/atoll at which they were collected than to any other island. These patterns are supported by available photo-ID and animal movement data (Karczmarski *et al.* 2005). Spinner dolphin genetic data are lacking from some islands and atolls within the Hawaiian Archipelago (e.g., Molokai, Kahoolawe, Nihoa, Mokumanamana (Necker), Gardner Pinnacles, Laysan, and Lisianski). Sighting data confirms spinner dolphin presence at some locations (e.g., Molokai, Kahoolawe, Mokumanamana, and Gardner Pinnacles; PIFSC unpublished data), however, without genetic or photo-identification data it is difficult to evaluate connectivity between these dolphins and those at other islands.

Hill *et al.* (2010) proposed designation of island-associated stocks of spinner dolphins at Midway/Kure, Pearl and Hermes Reef, Kauai/Niihau, Oahu/4-Islands, and Hawaii Island based on microsatellite and mtDNA genetic data, movement patterns, and the geographic distances between the Hawaiian Islands (Karczmarski 2005, Andrews *et al.* 2010). They suggested an offshore boundary for each island-associated stock at 10 nmi from shore based on anecdotal accounts of spinner dolphin distribution. Analysis of individual spinner dolphin movements suggests that few individuals move long distances (from one main Hawaiian Island to another) and no dolphins have been seen farther than 10 nmi from shore (Hill *et al.* 2011). Based on the maximum distance from shore observed for island-associated animals, a 10 nmi stock boundary is assumed for management under the MMPA. Norris *et al.* (1994) suggested that spinner dolphins may move seasonally between leeward and windward shores of the main Hawaiian Islands.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, six spinner dolphin stocks are recognized within the U.S. EEZ of the Hawaiian Islands: (1) Hawaii Island, (2) Oahu/4-Islands, (3) Kauai/Niihau, (4) Pearl & Hermes Reef, (5) Kure/Midway, and (6) Hawaii Pelagic. This includes animals found within the Hawaiian Islands EEZ (outside of island-associated boundaries) and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaii pelagic stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005). Spinner dolphins in the eastern tropical Pacific that may interact with tuna purse-seine fisheries are managed separately under the MMPA.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaii-based fisheries cause marine mammal mortality and serious injury in other U.S. waters. Seven spinner dolphins were reported hooked or entangled by fishing gear or marine debris in the main Hawaiian Islands from 2012 through 2016, five from the Hawaii Island stock, and two from the Oahu/4-Islands stock (Bradford and Lyman 2015, 2018). All cases were reviewed following the criteria for assessing serious injury in marine mammals (NMFS 2012). In two cases off Kailua-Kona in 2012 and 2014, individual spinner dolphins were observed with line, net, or other debris entangled around the rostrum preventing the dolphin from opening its mouth, and in one case with additional trailing gear (Bradford and Lyman 2015, 2017). Both cases were considered serious injuries given the potential of the line to impact the animal's ability to feed. In April 2013, a spinner dolphin was observed off Mahaiula Beach, Hawaii entangled in fishing gear (300+ ft. of fishing line, float, glow stick and hook). A swimmer cut the line close to the body, removing much of the trailing line and associated gear, but leaving several wraps of line around the dolphin's tail. This animal was considered seriously injured despite the gear removal because it is unclear whether the mitigation improved the animal's status. In June 2016, a spinner dolphin was observed off Kailua-Kona, Hawaii with a single wrap of small gauge fishing line around and cutting into its tail stock and trailing 40-50 feet behind. A diver removed most of the trailing line, reducing the length to about 6 feet. The animal was considered seriously injured because the constricting wrap remained and was constriction was possibly worsened by the attempt to remove the gear. In March 2014, a male spinner dolphin stranded off Keahole Pt, Hawaii with twine netting wrapped around its rostrum and peduncle. Examination revealed hemorrhage at the rostrum and peduncle and suggested the animal drowned due to the entanglement. In March 2013 a spinner dolphin was observed off Waikiki, Oahu with a bag through its mouth and wrapped behind its head. This entanglement was considered a serious injury given the bag was unlikely to degrade causing an adverse health response. In January 2014 a spinner dolphin was observed at the entrance of Manele Bay, Lanai with red line/net wrapped around its rostrum and trailing down part of the body. This entanglement was considered a serious injury as the placement of the wrap could impact the animal's ability to feed. It is not possible to attribute any of these interactions to specific fisheries given the generic nature of the gear. There are eight additional reports between 1991 and 2011 of spinner dolphins found entangled, hooked, or shot (Bradford and Lyman 2013). No estimates of annual human-caused mortality and serious injury are available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species interactions.

Between 2012 and 2016, no spinner dolphins were observed hooked or entangled in either the deep-set (20-22% observer coverage) or shallow-set (100% observer coverage) longline fisheries operating in pelagic waters of the Hawaii EEZ and surrounding high-seas (Bradford and Forney 2017, Bradford 2018).

## HAWAII ISLAND STOCK

## POPULATION SIZE

Over the past few decades several abundance estimates were generated from studies along the Kona coast of Hawaii Island. Norris *et al.* (1994) photo-identified 192 individuals primarily within Kealekekua Bay along the west coast of Hawaii and estimated 960 animals for this area in 1979-1980. Östman (1994) photo-identified 677 individual spinner dolphins from a broader region, extending north to the Kohala Coast, from 1989 to 1992 and using the same estimation procedures as Norris *et al.* (1994), estimated a population size of 2,334 spinner dolphins. From 2010 to 2012, intensive year-round photo-identification surveys for spinner dolphins were carried out in Kauhako Bay, Kealakekua Bay, Honaunau Bay, and Makako Bay along the Kona Coast of Hawaii Island (Tyne *et al.* 2013). These surveys represent the most systematic and geographically extensive surveys for spinner dolphins in this region. Several mark-recapture models were evaluated with available data to examine the impact of sampling design. Models that used the most complete dataset yielded abundance estimates of 617 (CV=0.09) in 2011 and 665 (CV=0.09) in 2012 (Tyne *et al.* 2016). These are the best available and most recent abundance estimates for this stock. Considerable seasonal variation in spinner dolphin occurrence on the leeward versus south and east sides of the island may occur, with lower abundance off the leeward Kona coast in the winter, potentially due to increased wind and swell in that region (Norris *et al.* 1994). Because the most recent abundance estimate is based on year-round surveys, some of the animals seasonally present on the leeward side have likely been seen. However, because only four bays were surveyed, some portion of the population is likely not included in this abundance estimate and the new estimate is an underestimate of total population size.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution of the 2012 abundance estimate for Hawaii Island, or 617 spinner dolphins (Barlow *et al.* 1995).

### Current Population Trend

Quantitative trend analyses have not been conducted with available data, as estimates from the 1970s and 1980s did not include year-round surveys and occurred in a different study area than the 2010-2012 surveys. Tyne *et al.* (2016) evaluated the impact of sampling intensity and frequency on the ability to detect trends within this population and estimated that 6 annual estimates resulting from 7 years of monthly surveys at all four monitored bays would be required to detect a 5% change in population size with 80% power. Abundance estimates resulting from surveys at 3-year intervals would detect change with fewer surveys, over a longer time period (9-12 years).

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii Island stock is calculated as the minimum population estimate (617) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a stock of unknown status; Wade and Angliss 1997) resulting in a PBR of 6.2 spinner dolphins per year.

## STATUS OF STOCK

The Hawaii Island stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Hawaii Island spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Insufficient information is available to determine whether the total fishery mortality and serious injury for this Hawaii Island spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis and Timmel 2009). A two year study including collection of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their behavioral activities on a daily basis (Tyne *et al.* 2017), with rest most common midday and travel and socializing in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests they are less resilient to human disturbance than other cetaceans. Further, Tyne *et al*. (2015) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, such that displacement from resting bays by tourist or other activities, would reduce rest time, with potential for long-term health consequences in the population. Heenehan *et al*. (2017a) measured acoustic response of spinner dolphins to human activities and found that the dolphins increased vocal activity in two bays with predominantly dolphin-directed activities, but less so in bays with noise attributed to a broader range of human activities, suggesting greater behavioral disruption by human activities directed at the dolphins.

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. Naval sonar has been detected within spinner dolphin resting bays, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 (Heenehan *et al.* 2017b). Detection of the same sonar event at multiple spinner dolphin resting bays suggests a single sonar event may expose the entire spinner dolphin stock.

One spinner dolphin found stranded on Oahu tested positive for *Morbillivirus* (Jacob 2012). Although *morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animal is not known (Jacob 2012). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans. A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015. A retrospective analysis of previously stranded and archived spinner dolphins from Hawaii is now underway to determine if others may have died from the disease.

## OAHU/4-ISLANDS STOCK
## POPULATION SIZE

As part of the Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study, a total of twelve aerial surveys were conducted within 25 nmi of the main Hawaiian Islands in 1993, 1995 and 1998. An abundance estimate of 3,184 (CV=0.37) spinner dolphins resulted from the combined survey data (Mobley *et al.* 2000), now representing the Kauai/Niihau, Oahu/4-Islands, and Hawaii Island stocks. It is not feasible to partition this estimate into island-specific abundance estimates given the available data. New photo-ID mark-recapture estimates have resulted in seasonal abundance estimates for the Oahu/4-Islands stock. Closed capture models provide two separate estimates for the leeward coast of Oahu representing different time periods: 160 (CV = 0.14) for June to July, 2002; and 355 (CV = 0.09) for July to September 2007 (Hill *et al.* 2011). Both the 2002 and 2007 estimates likely underestimate true stock abundance as they include only dolphins found off the leeward coast of Oahu, and do not account for individuals that may spend most of their time along other parts of Oahu or somewhere in the 4-Islands area. The 2007 estimate is >8 years old and therefore is no longer used for stock assessment, based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005).

### Minimum Population Estimate

No minimum population estimate is available for this stock, as the most recent estimate of abundance is >8 years old.

### Current Population Trend

There are insufficient data to evaluate trends in abundance for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters. A default level of 4% is assumed for maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Oahu/4-Islands stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status; Wade and Angliss 1997). Because there is no minimum population estimate for Oahu/4-Islands spinner dolphins, the potential biological removal (PBR) is undetermined.

**STATUS OF STOCK**

The Oahu/4-Islands stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Oahu/4-Islands spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Insufficient information is available to determine whether the total fishery mortality and serious injury for this Oahu/4-Islands spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis and Timmel 2009). A two year study including collection of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their behavioral activities on a daily basis (Tyne *et al.* 2017), with rest most common midday and travel and socializing in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests they are less resilient to human disturbance than other cetaceans. Further, Tyne et al (2015) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, such displacement from resting bays by tourist or other activities, would reduce rest time, with potential for long-term health consequences for the population. Heenehan *et al*. (2017) measured acoustic response of spinner dolphins to human activities and found that the dolphins increased vocal activity in two bays with predominantly dolphin-directed activities, but less so in bays with noise attributed to a broader range of human activities, suggesting greater behavioral disruption by human activities directed at the dolphins.

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. Naval sonar has been detected within monitored spinner dolphin resting bays on Hawaii Island, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 (Heenehan *et al.* 2017b). Detection of the same sonar event at multiple spinner dolphin resting bays suggests a single sonar event may expose the entire spinner dolphin stock. Naval training also occurs near the other main Hawaiian Islands, suggesting the Hawaii Islands observations are not unique.

One spinner dolphin found stranded on Oahu has tested positive for *Morbillivirus* (Jacob 2012). Although *morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animal is not known (Jacob 2012). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans. A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015. A retrospective analysis of all previously stranded and archived spinner dolphins from Hawaii is now underway to determine if others may have died from the disease.

**KAUAI/NIIHAU STOCK**
**POPULATION SIZE**

As part of the Marine Mammal Research Program of the Acoustic Thermometry of Ocean Climate (ATOC) study, a total of twelve aerial surveys were conducted within 25 nmi of the main Hawaiian Islands in 1993, 1995 and 1998. An abundance estimate of 3,184 (CV=0.37) spinner dolphins was calculated from the combined survey data (Mobley *et al.* 2000), now representing the Kauai/Niihau, Oahu/4-Islands, and Hawaii Island stocks. More recent mark-recapture estimates based on photo-identification studies resulted in an estimate of 601 (CV = 0.20) spinner dolphins for the leeward coast of Kauai for the period October to November 2005. This estimate is likely an underestimate as it includes only dolphins found off the leeward coast of Kauai, and does not account for individuals that may spend most of their time along other parts of Kauai, Niihau, or Kaula Rock. The 2005 estimate is now >8 years old and therefore will no longer be used based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005).

**Minimum Population Estimate**

No minimum population estimate is available for this stock, as the most recent estimate of abundance is >8 years old.

### Current Population Trend

There is only one abundance estimate available for the stock area of Kauai/Niihau from 2005 and thus, no trend analysis is possible.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters. A default level of 4% is assumed for maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Kauai/Niihau stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status; Wade and Angliss 1997). Because there is no minimum population estimate for Kauai/Niihau spinner dolphins, the potential biological removal (PBR) is undetermined.

## STATUS OF STOCK

The Kauai/Niihau stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Kauai/Niihau spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate abundance trends. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Insufficient data are available to determine whether the total fishery mortality and serious injury for this Kauai/Niihau spinner dolphin stock is insignificant and approaching zero mortality and serious injury rate.

A habitat issue of increasing concern is the potential effect of swim-with-dolphin programs and other tourism activities on spinner dolphins around the main Hawaiian Islands (Danil *et al.* 2005, Courbis and Timmel 2009). A two year study including collection of behavioral time-series data indicates that spinner dolphins off the leeward coast of Hawaii Island spatially and temporally partition their behavioral activities on a daily basis (Tyne *et al.* 2017), with rest most common midday and travel and socializing in early morning and late afternoon. Foraging was not observed during the daytime. This behavior pattern suggests they are less resilient to human disturbance than other cetaceans. Further, Tyne et al (2015) observed that spinner dolphins do not engage in rest behavior outside of sheltered bays, such displacement from resting bays by tourist or other activities, would reduce rest time, with potential for long-term health consequences for the population. Heenehan *et al.* (2017) measured acoustic response of spinner dolphins to human activities and found that the dolphins increased vocal activity in two bays with predominantly dolphin-directed activities, but less so in bays with noise attributed to a broader range of human activities, suggesting greater behavioral disruption by human activities directed at the dolphins.

All Hawaiian spinner dolphin stocks are potentially exposed to high levels of Navy sonar and frequent detonations during training exercises. The sensitivity of spinner dolphins to these sound levels is unknown and therefore the impact of these exercises on spinner dolphin stocks is unknown. Naval sonar has been detected within monitored spinner dolphin resting bays on Hawaii Island, with median sonar exposure levels between 24.7 and 45.8 dB above median sound levels on one occasion in 2011 (Heenehan *et al.* 2017b). Detection of the same sonar event at multiple spinner dolphin resting bays suggests a single sonar event may expose the entire spinner dolphin stock. Naval training also occurs near the other main Hawaiian Islands, suggesting the Hawaii Islands observations are not unique.

One spinner dolphin found stranded on Oahu has tested positive for *Morbillivirus* (Jacob 2012). Although *morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animal is not known (Jacob 2012). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans. A spinner dolphin stranded off Hawaii Island was also determined to have died from infection with toxoplasmosis in 2015. A retrospective analysis of all previously stranded and archived spinner dolphins from Hawaii is now underway to determine if others may have died from the disease.

## PEARL & HERMES REEF STOCK
## POPULATION SIZE

There is no information on the abundance of the Pearl & Hermes Reef stock of spinner dolphins. A photo-identification catalog of individual spinner dolphins from this stock is available, though inadequate survey effort and low re-sighting rates prevent robust estimation of abundance.

### Minimum Population Estimate

There is no information on which to base a minimum population estimate for the Pearl & Hermes Reef stock of spinner dolphins.

### Current Population Trend

Insufficient data exists to assess population trends.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Pearl & Hermes Reef stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status; Wade and Angliss 1997). Because there is no minimum population estimate available for this stock the PBR for Pearl & Hermes Reef stock of spinner dolphins is undetermined.

## STATUS OF STOCK

The Pearl & Hermes Reef stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Pearl & Hermes Reef spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance for this stock. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Because the stock resides entirely within the Paphanaumokuakea Marine National Monument, where fishing is not permitted, it is assumed that the rate of mortality and serious injury within the stock area is zero. It is unlikely that habitat issues facing spinner dolphin in the main Hawaiian Islands impact those in the northwestern Hawaiian Islands to the same magnitude given their relative isolation from tourism, military sonar activities, and urban water input to the environment. Pearl and Hermes stock spinner dolphins may still be vulnerable to infection with *morbillivirus* or *Brucella* given transmission through wild populations is not well understood and not necessarily related to coastal proximity.

## MIDWAY ATOLL/KURE STOCK
## POPULATION SIZE

In the Northwestern Hawaiian Islands, a multi-year photo-identification study at Midway Atoll resulted in a population estimate of 260 spinner dolphins based on 139 identified individuals (Karczmarski *et al.* 1998). This abundance estimate for the Midway Atoll/Kure stock of spinner dolphins is > 8 years old and therefore will no longer be used, based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005). A 2010 shipboard line-transect survey within the Hawaiian EEZ resulted in a single off-effort sighting of spinner dolphins at Kure Atoll. This sighting cannot be used within a line-transect framework; however, photographs of individuals may be used in the future to estimate the abundance of spinner dolphin at Midway Atoll/Kure using mark-recapture methods.

### Minimum Population Estimate

The minimum population estimate for the Midway Atoll/Kure stock is > 8 years old and therefore will no longer be used (NMFS 2005). There is no current minimum population estimate available for this stock.

### Current Population Trend

Insufficient data exists to assess population trends.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Midway Atoll/Kure stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status; Wade and Angliss 1997). The PBR for the Midway Atoll/Kure stock of spinner dolphins is undetermined because no minimum population estimate is available for this stock.

## STATUS OF STOCK

The Midway Atoll/Kure stock of spinner dolphins is not considered strategic under the MMPA. The status of Midway Atoll/Kure spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Because the stock resides entirely within the Paphanaumokuakea Marine National Monument, where fishing is not permitted, it is assumed that the rate of mortality and serious injury within the stock area is zero. It is unlikely that habitat issues facing spinner dolphin in the main Hawaiian Islands impact those in the northwestern Hawaiian Islands to the same magnitude given their relative isolation from tourism, military sonar activities, and urban water input to the environment. The Midway Atoll/Kure stock of spinner dolphins may still be vulnerable to infection with *morbillivirus* or *Brucella* given transmission through wild populations is not well understood and not necessarily related to coastal proximity.

## HAWAII PELAGIC STOCK
## POPULATION SIZE

A 2002 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in an abundance estimate of 3,351 (CV=0.74) spinner dolphins (Barlow 2006); however, this estimate assumed a single Hawaiian Islands stock. Two of the 8 sightings during the 2002 survey did occur in pelagic waters far outside of the current island-associated stock boundaries, suggesting at least some spinner dolphins do occur in pelagic archipelago waters. This estimate for the Hawaiian EEZ is > 8 years old and therefore will no longer be used based on NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2005). A 2010 shipboard line-transect survey within the Hawaiian EEZ did not result in any sightings of pelagic spinner dolphins.

### Minimum Population Estimate

No minimum population estimate is available for this stock, as there were no sightings of pelagic spinner dolphins during a 2010 shipboard line-transect survey of the Hawaiian EEZ.

### Current Population Trend

Insufficient data exists to assess population trends.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii pelagic stock is calculated as the minimum population estimate <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status; Wade and Angliss 1997). Because there is no minimum population estimate for Hawaii pelagic spinner dolphins, the potential biological removal (PBR) is undetermined.

## STATUS OF STOCK

The Hawaii pelagic stock of spinner dolphins is not considered a strategic stock under the MMPA. The status of Hawaii pelagic spinner dolphins relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The estimated rate of fishery mortality and serious injury for this stock is zero in observed U.S. fisheries. This stock likely extends outside of U.S. EEZ waters, where international high seas fisheries may interact with and take animals from this stock. Exposure of pelagic spinner dolphins to habitat stressors common for island-associated spinner stocks in the main Hawaiian Islands is unknown. The Hawaii pelagic stock of spinner dolphins may be vulnerable to infection with *morbillivirus* or *Brucella* given transmission through wild populations is not well understood and not necessarily related to coastal proximity.

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Revised 01/15/2011

# SPINNER DOLPHIN (*Stenella longirostris longirostris*): American Samoa Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Gray's spinner dolphins (*Stenella longirostris longirostris*) are the most widely distributed subspecies of spinner dolphins and are found in tropical and subtropical waters of the Atlantic, Indian, central and western Pacific Oceans (Perrin et al. 1991, Norris et al. 1994, Oremus *et al.* 2007, Johnston et al. 2008). Spinner dolphins are considered common in American Samoa (Reeves et al. 1999). During small-boat surveys from 2003 to 2006 in the waters surrounding the island of Tutuila, the spinner dolphin was the most frequently encountered species (i.e., 34 of 52 sightings) and was found in waters with a mean depth of 44m (Johnston et al. 2008). Photo-identification data collected during the surveys indicate the presence of a resident population of spinner dolphins in the waters surrounding Tutuila (Johnston et al. 2008). Approximately 1/3 of the individuals within the photo-id catalog were sighted in multiple years (Johnston et al. 2008). In addition, some of these individuals demonstrated strong site fidelity and were encountered within only a few kilometers from one year to the next (Johnston et al. 2008). During a shipboard survey in 2006 spinner dolphins were also encountered just south of the island of Ta‘u, American Samoa (Johnston et al. 2008).

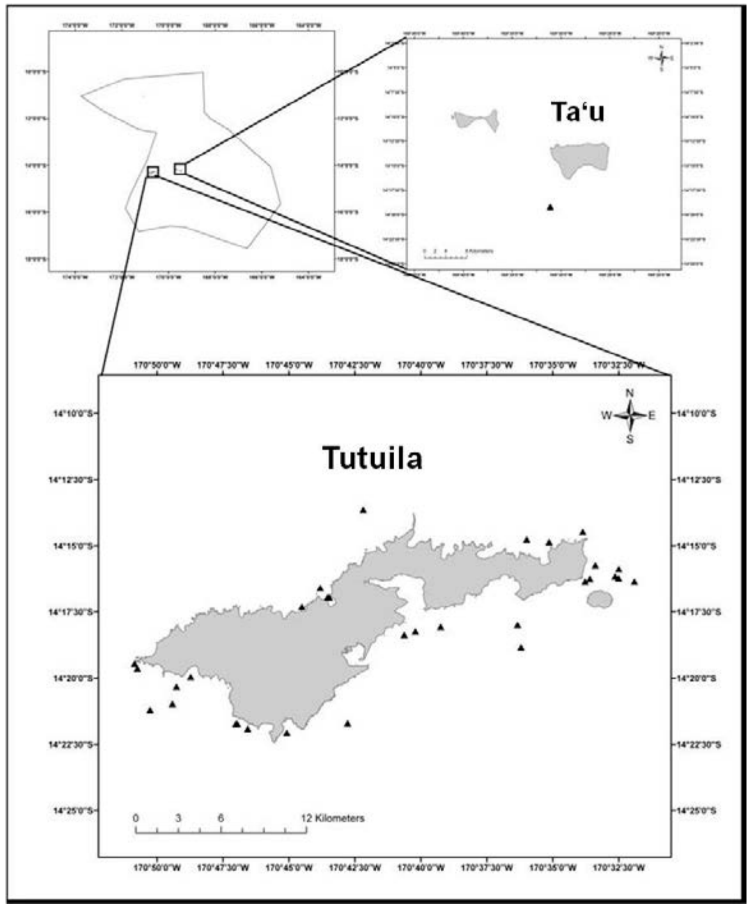


Figure 1. Spinner dolphin sightings from visual sighting surveys, 2003-2006 (Johnston et al 2008).

Genetic analyses of biopsy samples collected during the 2003-2006 small boat surveys around Tutuila indicate that spinner dolphins in American Samoa are distinct from those of the Hawaiian Archipelago. Pairwise F-statistical analyses revealed significant ($p<0.001$) genetic distinction, at both mtDNA and microsatellite loci, between spinner dolphins sampled in American Samoa and those sampled in the Hawaiian Islands (Johnston et al. 2008, Andrews 2009). For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are eight Pacific management stocks, six of these extend from the Hawaiian archipelago to 10 nmi offshore: 1) Kure/Midway, 2) Pearl and Hermes Reef, 3) French Frigate Shoals, 4) Kauai/Niihau, 5) Oahu/4-Islands, and 6) Hawaii Island, The Hawaii Pelagic Stock, which includes animals within the U.S. EEZ of the Hawaiian Islands, but more than 10 nmi from the shore where insular populations exist, and 8) the American Samoa Stock, which include animals inhabiting the EEZ waters around American Samoa (this report). Spinner dolphins involved in eastern tropical Pacific tuna purse-seine fisheries are managed separately under the MMPA.

## POPULATION SIZE

No abundance estimates are currently available for spinner dolphins in U.S. EEZ waters of American Samoa; however, density estimates for spinner dolphins in other tropical Pacific regions can provide a range of likely abundance estimates in this unsurveyed region. Published estimates of spinner dolphins (animals per $km^2$) in the Pacific are: 0.0014 (CV=0.74) for the U.S. EEZ of the Hawaiian Islands (Barlow 2006); 0.0443 (CV=0.37) for nearshore waters surrounding the main Hawaiian Islands (Mobley et al. 2000), 0.0532 (CV=0.19) and 0.0473 (CV=0.15) for the eastern tropical Pacific Ocean (Wade and Gerrodette 1993; Ferguson and Barlow 2003), and 0.1280 (CV=0.27) for eastern tropical Pacific waters west of 120$^{o}$W and north or south of 10$^{o}$, a region with similar oceanographic conditions to those around American Samoa. Applying the lowest and highest of these density estimates to U.S. EEZ waters surrounding American Samoa (area size = 404,578 km$^2$) yields a range of plausible abundance estimates of 553 – 51,773 spinner dolphins.

## Minimum Population Estimate

No minimum population estimate is currently available for waters surrounding American Samoa, but the spinner dolphin density estimates from other tropical Pacific regions (Barlow 2003, Mobley et al. 2000, Wade and Gerrodette 1993, Ferguson and Barlow 2003, see above) can provide a range of likely values. The lognormal 20$^{th}$ percentiles of plausible abundance estimates for the American Samoa EEZ, based on the densities observed elsewhere, range from 317 – 41,483 spinner dolphins.

## Current Population Trend

No data are available on current population trend.

# CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current and maximum net productivity rate in American Samoan waters.

# POTENTIAL BIOLOGICAL REMOVAL

No PBR can presently be calculated for spinner dolphins within the American Samoa EEZ, but based on the range of plausible minimum abundance estimates (317 – 41,483), a recovery factor of 0.50 (for a species of unknown status with no fishery mortality and serious injury within the American Samoa EEZ; Wade and Angliss 1997), and the default growth rate (½ of 4%), the PBR would likely fall between 3.2 and 415 spinner dolphins per year.

# HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in American Samoan waters is limited, but the gear types used in American Samoa's fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used, and float lines from lobster traps and longlines can be expected to occasionally entangle whales (Perrin et al. 1994). The primary fishery in American Samoa is the commercial pelagic longline fishery that targets tunas, which was introduced in 1995 (Levine and Allen 2009). In 2008, there were 28 federally permitted vessels within the longline fishery in American Samoa (Levine and Allen 2009). The fishery has been monitored since March 2006 under a mandatory observer program, which records all interactions with protected species (Pacific Islands Regional Office 2009). No interactions with spinner dolphins have been recorded. Prior to 1995, bottomfishing and trolling were the primary fisheries in American Samoa but became less prominent after longlining was introduced (Levine and Allen 2009). Nearshore subsistence fisheries include spear fishing, rod and reel, collecting, gill netting, and throw netting (Craig 1993, Levine and Allen 2009). Information on fishery-related mortality of cetaceans in the nearshore fisheries is unknown, but the gear types used in American Samoan fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used. Although boat-based nearshore fisheries have been randomly monitored since 1991, by the American Samoa Department of Marine and Wildlife Sources (DMWR), no estimates of annual human-caused mortality and serious injury of cetaceans are available.

# STATUS OF STOCK

The status of spinner dolphins in American Samoan waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known for spinner dolphins in American Samoa. Spinner dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The American Samoan stock of spinner dolphins is not considered a strategic stock under the 1994 amendments to the MMPA because the estimated rate of mortality and serious injury within the American Samoa EEZ is zero. Insufficient information is available to determine whether the total fishery mortality and serious injury for spinner dolphins is insignificant and approaching zero mortality and serious injury rate.

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# STRIPED DOLPHIN (*Stenella coeruleoalba*): Hawai‘i Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Striped dolphins are found in tropical to warm-temperate waters throughout the world (Perrin *et al.* 2009). Sightings have historically been infrequent in shallow waters (Shallenberger 1981, Mobley *et al.* 2000), though they are common, even nearshore, in waters greater than 3500m (Baird 2016). Striped dolphins are often seen offshore throughout the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands during periodic shipboard surveys (Figure 1).

Striped dolphins have been intensively exploited in the western North Pacific, where three migratory stocks are provisionally recognized (Kishiro and Kasuya 1993). In the eastern tropical Pacific, all striped dolphins are provisionally considered to belong to a single stock (Dizon *et al.* 1994). There is insufficient data to examine finer stock structure within Hawaiian waters, though available data do not suggest island-associated populations (Baird 2016).

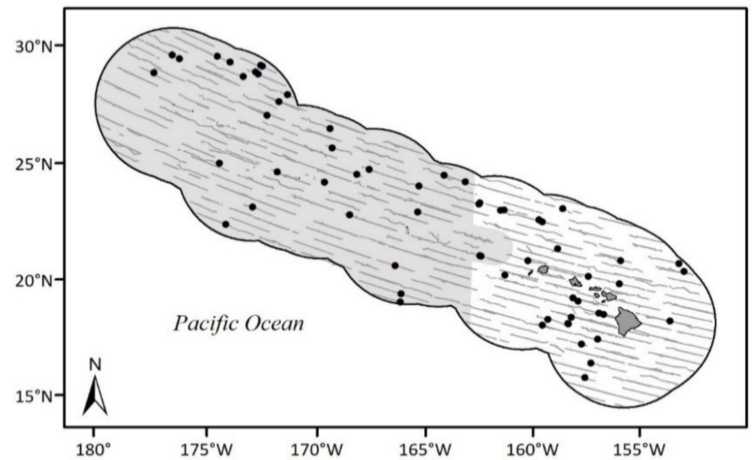


**Figure 1.** Striped dolphin sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al*. 2018) shipboard surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, striped dolphins within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) waters off California, Oregon and Washington, and 2) waters around Hawai‘i (this report), including animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawai‘i stock is evaluated based on data from the U.S. EEZ around the Hawaiian Islands (NMFS 2023). Striped dolphins involved in eastern tropical Pacific tuna purse-seine fisheries are managed separately under the MMPA.

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in updated model-based abundance estimates of striped dolphins in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2021, 2022; Table 1).

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates. The modeling framework incorporated Beaufort-specific trackline detection probabilities for striped dolphins from Barlow *et al.* (2015). Models were used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016). When model-based estimates are available for 2017 from both analyses, the results are largely similar for most species; however, striped dolphins are a notable exception, with 2017 estimates from Becker *et al,* (2022) nearly double those from Becker *et al.* (2021). Although Becker *et al.* (2022) attribute this change to the use of new calibrated group size, detailed review of the functional form of the model predictors reveals a shift from a linear decline in density with depth in Becker *et al.* (2021) to a thresholded form in Becker *et al.* (2022), with density constant at depths less than 3000m, leading to higher densities in shallow depths than the previous models. Bradford *et al.* (2021) produced design-based abundance estimates for striped dolphins in 2002, 2010, and 2017 that can be used as a point of comparison to the model-based estimates for those years.

**Table 1.** Model-based line-transect abundance estimates for striped dolphins in the Hawaiian Islands EEZ in 2002 and 2010 (Becker *et al.* 2021) and 2017 and 2020 (Becker *et al.* 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

| Year | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2020 | 64,343 | 0.28 | 37,822-109,462 |
| 2017 | 59,493 | 0.28 | 35,050-100,981 |
| 2010 | 36,886 | 0.22 | 24,004-56,681 |
| 2002 | 35,817 | 0.22 | 23,384-54,861 |

There is substantial variability within and between the design and model-based estimates across the time series (Figure 2), suggesting additional survey data are needed to develop a well-parameterized model for this species. Despite the substantial variability in the abundance estimates for this species, the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of striped dolphins within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Seasonal predictions using the model showed no reliance on dynamic variables, and design-based estimates were broadly similar (with broad and overlapping confidence intervals). Previously published abundance estimates for the Hawaiian Islands EEZ (e.g. Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 64,343 (CV=0.28) striped dolphins.

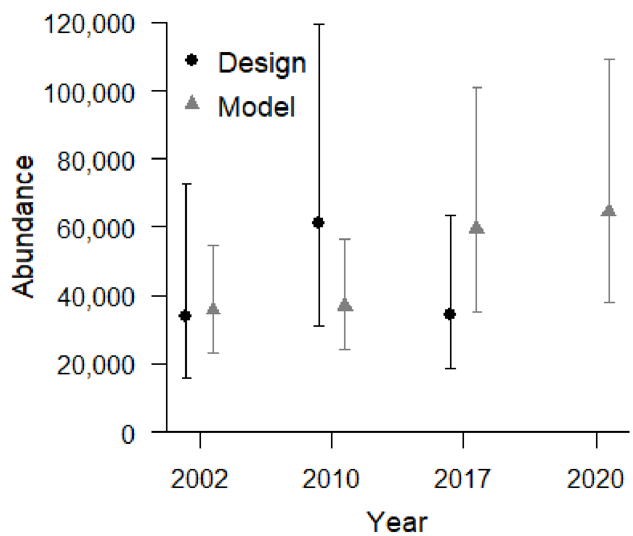


**Figure 2.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for striped dolphins for each survey year (2002, 2010, 2017, 2020).

Population estimates are available for Japanese waters (Miyashita 1993) and the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

**Minimum Population Estimate**

The minimum population estimate is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate (from Becker *et al.* 2022), or 51,055 striped dolphins.

**Current Population Trend**

The model-based abundance estimates for striped dolphins provided by Becker *et al.* (2021, 2022) are highly variable and do not explicitly allow for examination of population trend. Model-based examination of striped dolphin population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawai‘i stock of striped dolphins is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (51,055) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a stock of unknown status with no known fishery mortality and serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 511 striped dolphins per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaii (Nitta and Henderson, 1993). In 2021, a striped dolphin stranded on Maui with scarring on its rostrum consistent with a previous hooking and scarring on its peduncle consistent with a previous entanglement, although these findings were not considered to be related to the cause of death (Bradford and Lyman in press). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

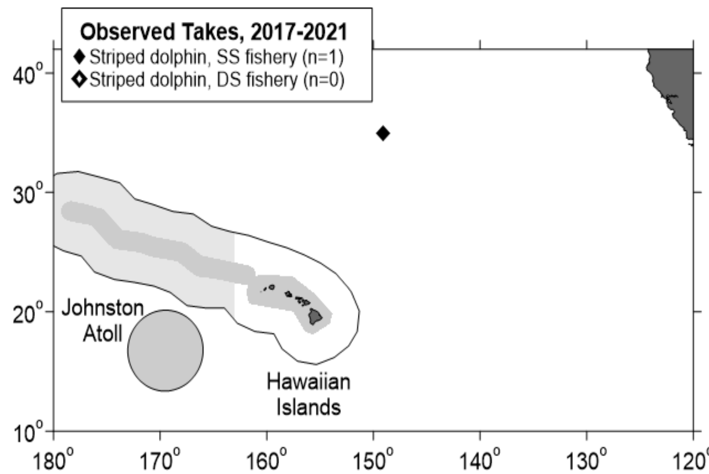


**Figure 3.** Location of a striped dolphin take within the shallow-set fishery (filled diamond) in Hawaii-based longline fisheries, 2017-2021. Solid lines represent the U.S. EEZs. Gray shading notes areas closed to longline fishing.

There are currently two distinct longline fisheries based in Hawai‘i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2017 and 2021, one striped dolphin was observed hooked or entangled in the SSLL fishery (100% coverage) outside of the U.S. EEZ, and no striped dolphins were observed hooked or entangled in the DSLL fishery (15-21% observer coverage) (Figure 3, Bradford 2018, 2020, 2021, 2023, in review). The striped dolphin was considered not seriously injured based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2023b).

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken and Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2022 are 0.2 dolphins outside of the U.S. EEZ, and 0 within the Hawaiian Islands EEZ (Table 2).

**Table 2.** Summary of available information on incidental mortality and serious injury (MSI) of striped dolphin (Hawai‘i stock) in commercial longline fisheries, within and outside of the U.S. EEZ (McCracken and Cooper 2022b). Mean annual takes are based on 2017-2021 data unless otherwise indicated. Information on all observed takes (T) and

MSI is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Outside U.S. EEZ | | Hawaiian Islands EEZ | |
|---|---|---|---|---|---|---|---|
| | | | | **Observed T/MSI** | **Estimated MSI (CV)** | **Observed T/MSI** | **Estimated MSI (CV)** |
| **Hawai'i-based deep-set longline fishery** | 2017 | Observer data | 20% | 0 | 0 (-) | 0 | 0 (-) |
| | 2018 | | 18% | 0 | 0 (-) | 0 | 0 (-) |
| | 2019 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2020 | | 15% | 0 | 0 (-) | 0 | 0 (-) |
| | 2021 | | 18% | 0 | 0 (-) | 0 | 0 (-) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | **0 (-)** | | **0 (-)** |
| | 2017 | Observer data | 100% | 1/0 | 1 | 0 | 0 |
| | 2018 | | 100% | 0 | 0 | 0 | 0 |
| | 2019 | | 100% | 0 | 0 (-) | 0 | 0 (-) |
| | 2020 | | 100% | 0 | 0 (-) | 0 | 0 (-) |
| | 2021 | | 100% | 0 | 0 (-) | 0 | 0 (-) |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | | **0.2** | | **0** |
| **Minimum total annual takes within U.S. EEZ (2017-2021)** | | | | | | | **0 (-)** |

## STATUS OF STOCK

The Hawai'i stock of striped dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of striped dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Striped dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries in the U.S. EEZ, total fishery mortality and serious injury for striped dolphins can be considered insignificant and approaching zero. Several serious diseases have been found in stranded striped dolphins in Hawai'i. One striped dolphin stranded in the main Hawaiian Islands tested positive for *Brucella* (Chernov 2010), two for *Morbillivirus* (Jacob *et al.* 2016), and one for beaked whale circovirus (Clifton *et al.* 2023). *Brucella* is a bacterial infection that if common in the population may limit recruitment by compromising male and female reproductive systems, and can also cause neurological disorders that may result in death (Van Bressem *et al.* 2009). Although M*orbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al.* 2009), its impact on the health of the stranded animals is not known as it was found in only one tested tissue within each animal (Jacob *et al.* 2016). Beaked whale circovirus has been only recently described in cetaceans, with effects on the brain, lungs, and lymph system that may result in immunosuppression. Its role in the death of the striped dolphin was not clear, although all 6 tested tissues were positive for the disease. The presence of beaked whale circovirus and *Morbillivirus* each in 10 species (Clifton *et al.* 2023, Jacob *et al.* 2016) and *Brucella* in 3 species (Cherbov 2010, West unpublished data) raises concerns about the history and prevalence of these diseases in Hawai'i and the potential population impacts on Hawaiian cetaceans. It is not known if any of these diseases are common in the Hawai'i stock of striped dolphins.

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Revised 5/5/2021

# FRASER'S DOLPHIN (*Lagenodelphis hosei*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Fraser's dolphins are distributed worldwide in tropical waters (Dolar 2009 in Perrin *et al*. 2009). The species was first documented within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands during a 2002 cetacean survey (Barlow 2006) and have been occasionally observed during surveys of Hawaiian waters since that time (Bradford *et al*. 2017, Yano *et al*. 2018, Figure 1). There have been only 4 sightings of Fraser's dolphins during nearshore surveys in the leeward main Hawaii Islands since the early 2000s (Baird *et al*. 2013).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there is a single Pacific management stock including animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

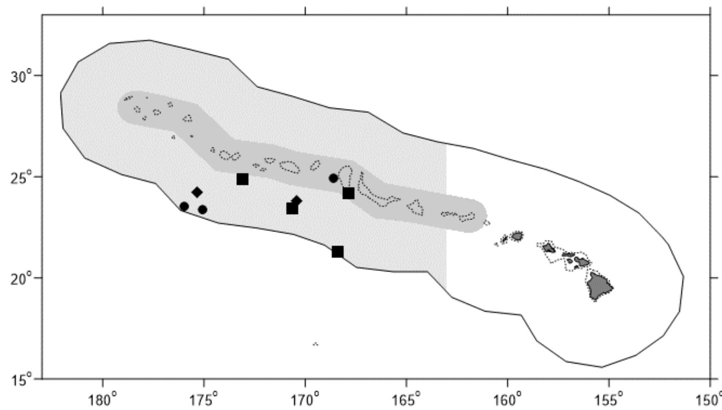


**Figure 1**. Fraser's dolphin sighting locations during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard cetacean surveys of U.S. waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ was recently reevaluated for each survey year, resulting in the following abundance estimates of Fraser's dolphins in the Hawaii EEZ (Bradford *et al*. 2021; Table 1).

Table 1. Line-transect abundance estimates for Fraser's dolphins derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 40,960 | 0.70 | 11,887-141,143 |
| 2010 | 56,688 | 0.70 | 16,391-196,056 |
| 2002 | 28,980 | 1.02 | 5,518-152,195 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for Fraser's dolphins from Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al*. (2021) uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available. The best estimate of abundance is based on a 2017 survey, or 40,960 (CV=0.70). Population estimates for Fraser's dolphins have been made in the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands and in the central North Pacific.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) of the 2017 abundance estimate or 24,068 Fraser's dolphins.

**Current Population Trend**

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for Fraser's dolphins.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of Fraser's dolphin is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (24,068) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 241 Fraser's dolphins per year.

## HUMAN CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No interactions between nearshore fisheries and Fraser's dolphins have been reported in Hawaiian waters. There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no Fraser's dolphins were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (20-21% observer coverage) (Bradford 2018a, 2018b, 2020*,* Bradford and Forney 2017, McCracken 2019). There were two other unidentified cetaceans taken in the DSLL fishery during this period, which may have been Fraser's dolphins.

## STATUS OF STOCK

The Hawaii stock of Fraser's dolphins is not considered strategic under the 1994 amendments to the MMPA. The status of Fraser's dolphins in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this stock. Fraser's dolphins are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries the total fishery mortality and serious injury can be considered to be insignificant and approaching zero.

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Revised 5/5/2021

# MELON-HEADED WHALE (*Peponocephala electra*): Hawaiian Islands Stock Complex: Hawaiian Islands & Kohala Resident Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Melon-headed whales are found in tropical and warm-temperate waters throughout the world. Although largely considered oceanic, photo-identification studies and other lines of evidence indicate the presence of island-associated populations in several locations (Brownell 2009). Small numbers have been taken in the tuna purse-seine fishery in the eastern tropical Pacific, and they are occasionally killed in direct fisheries in Japan and elsewhere in the western Pacific. Melon-headed whales in Hawaiian waters appear to prey primarily upon a large diversity of cephalopods, and fish species (West *et al*. 2018). Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in one sighting in 2002, one in 2010, and seven in 2017 (Figure 1; Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018).

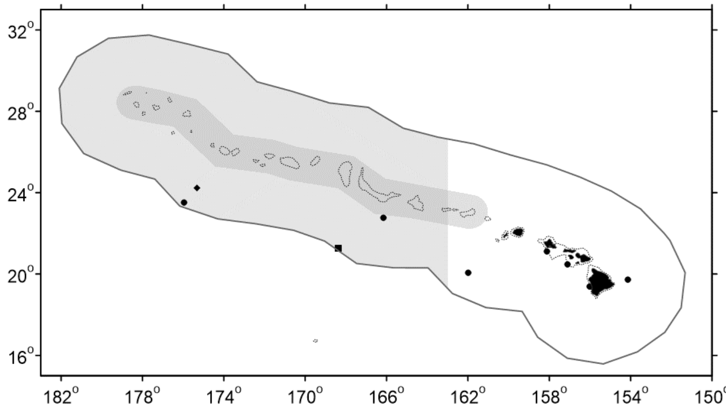


**Figure 1**. Melon-headed whale sighting location during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Photo-identification and telemetry studies suggest there are two demographically-independent populations of melon-headed whales in Hawaiian waters, the Hawaiian Islands stock and the Kohala resident stock. Resighting data and social network analyses of photographed individuals indicate very low rates of interchange between these populations (0.0009/yr) (Aschettino *et al.* 2012). This finding is supported by genetic analyses that indicate significant differentiation between the Kohala residents and other melon-headed whales sampled in Hawaiian waters, despite overall high levels of interchange among most populations sampled (Martien *et al*. 2017), suggesting differences in social organization and foraging behavior may drive the observed structure. Some individuals in each population have been seen repeatedly for more than a decade, implying high site-fidelity for both populations. Individuals in the larger Hawaiian Islands stock have been resighted throughout the main Hawaiian Islands. Satellite telemetry data revealed distant offshore movements, nearly to the edge of the U.S. EEZ around the Hawaiian Islands (Figure 2), with apparent foraging near cold and warm-core eddies (Woodworth *et al.* 2012). Individuals in the smaller Kohala resident stock have a range restricted to shallower waters of the Kohala shelf and west side of Hawaii Island (Aschettino *et al.* 2012, Oleson *et al.* 2013). Satellite telemetry data indicate they occur in waters less than 2500m depth around the northwest and west shores of Hawaii Island, west of $156^0$ 45’ W and north of $19^0$ 15’N (Oleson *et al.* 2013). The northern boundary between the two stocks provisionally runs through the Alenuihaha Channel between Hawaii Island and Maui, bisecting the distance between the 1000 m depth contours (Oleson *et al.* 2013). Genetic analysis showed the strongest differentiation between animals sampled at Palmyra Atoll and other locations (Martien *et al*. 2017)

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are two Pacific management stocks within the Hawaiian Islands EEZ (Oleson *et al.* 2013): 1) the Kohala resident stock, which includes melon-headed whales off the Kohala Peninsula and west coast of Hawaii Island and in less than 2500m of water, and 2) the Hawaiian Islands stock, which includes melon-headed whales inhabiting waters throughout the U.S. EEZ of the Hawaiian Islands, including the area of the Kohala resident stock, and adjacent high seas waters. At this time, assignment of individual melon-headed whales within the overlap area to either stock requires photographic-identification of the animal. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaiian Islands stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

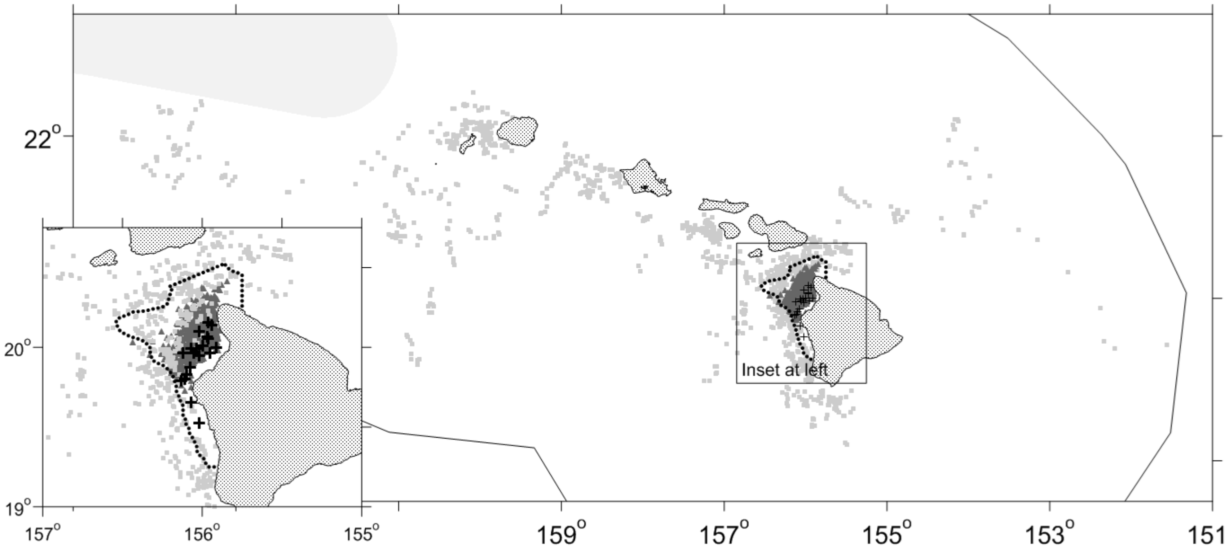


**Figure 2.** Sighting locations of melon-headed whales identified as being part of the Kohala resident stock (crosses) and telemetry records of Kohala resident (dark gray triangles) and Hawaiian Islands (light gray squares) melon-headed whale stocks (Oleson *et al.* 2013). The dotted line around waters adjacent to the northwest and west shores of Hawaii Island represents the provisional stock boundary for the Kohala resident stock (Oleson *et al.* 2013). The Kohala resident stock and the Hawaiian Islands stocks overlap throughout the range of the Kohala resident stock**.** Outer line represents U.S. EEZ. Gray shading indicates area of Papahanaumokuakea Marine National Monument.

Information on fishery-related mortality and serious injury of cetaceans in U.S. EEZ of the Hawaiian Islands waters is limited, but the gear types used in Hawaii fisheries are responsible for marine mammal mortality and serious injury in other U.S. fisheries. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaii (Nitta and Henderson, 1993). No interactions between nearshore fisheries and melon-headed whales have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch. Long-term photo-identification studies have noted individuals from both the Kohala Resident and Hawaiian Islands stocks with bullet holes in their dorsal fin or with linear scars on their fins or bodies (Aschettino 2010) which may be consistent fisheries interactions.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no melon-headed whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017,). However, three unidentified delphinids were taken in the DSLL fishery, some of which may have been melon-headed whales.

### Other Mortality

In recent years, there has been increasing concern that loud underwater sounds, such as active sonar and seismic operations, may be harmful to beaked whales (Cox *et al.* 2006) and other cetaceans, including melon-headed whales (Southall *et al.* 2006, 2013) and pygmy killer whales (*Feresa attenuata*) (Wang and Yang 2006). The use of active sonar from military vessels has been implicated in mass strandings of beaked whales and recent mass-stranding reports suggest some delphinids may be impacted as well. A 2004 mass-stranding of 150-200 melon-headed whales in Hanalei Bay, Kauai occurred during a multi-national sonar training event around Hawaii (Southall *et al.* 2006).

Although data limitations regarding the position of the whales prior to their arrival in the Bay, the magnitude of sonar exposure, behavioral responses of melon-headed whales to acoustic stimuli, and other possible relevant factors preclude a conclusive finding regarding the role of Navy sonar in triggering this event, sonar transmissions were considered a plausible cause of the mass stranding based on the spatiotemporal link between the sonar exercises and the stranding, the direction of movement of the transmitting vessels near Hanalei Bay, and propagation modeling suggesting the sonar transmissions would have been audible at the mouth of Hanalei Bay (Southall *et al.* 2006; Brownell *et al.* 2009). In 2008 approximately 100 melon-headed whales stranded within a lagoon off Madagascar during high-frequency multi-beam sonar use by oil and gas companies surveying offshore. Although the multi-beam sonar cannot be conclusively deemed the cause of the stranding event, the very close temporal and spatial association and directed movement of the sonar use with the stranding event, the unusual nature of the stranding event, and that all other potential causal factors were considered unlikely to have contributed, an Independent Scientific Review panel found that multi-beam sonar transmissions were a "plausible, if not likely" contributing factor (Southall *et al*. 2013) in this mass stranding event. This examination together with that of Brownell *et al.* (2009) suggests melon-headed whale may be particularly sensitive to impacts from anthropogenic sounds. No estimates of potential mortality or serious injury are available for U.S. waters.

## KOHALA RESIDENT STOCK

## POPULATION SIZE

Using the photo-ID catalog of individuals encountered between 2002 and 2009, Achettino (2010) used a POPAN open-population model to produce a mark-recapture abundance estimate of 447 (CV=0.12) individuals. The dataset used in this analysis is more than 8 years old, and there is no current estimate of abundance for this stock.

### Minimum Population Estimate

There is no current minimum population estimate for the Kohala resident stock of melon-headed whales. The data used in the 2002-2009 mark-recapture estimate (Aschettino 2010) are considered outdated, and therefore are not suitable for deriving a minimum abundance estimate.

### Current Population Trend

Only one abundance estimate is available for this stock, such that there is insufficient information to assess population trends.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population estimate times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997). Because there is no minimum population size estimate for this stock, the PBR is undetermined.

## STATUS OF STOCK

The Kohala resident stock of melon-headed whales is not considered strategic under the MMPA. The status of this stock relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Melon-headed whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reports of recent mortality or serious injuries; however, there is no systematic monitoring of takes in near-shore fisheries that may take this species. Given noted bullet holes and potential line injuries on individuals from this stock, insufficient information is available to determine whether the total fishery mortality and serious injury for Kohala Resident melon-headed whales is insignificant and approaching zero mortality and serious injury rate. The very restricted range and small population size of Hawaii Island resident melon-headed whales suggests this population may be at risk due to its proximity to U.S. Navy training, including sonar transmissions, in the Alenuihaha Channel between Hawaii Island and Maui (Forney *et al*. 2017). Although a 2004 mass-stranding in Hanalei Bay, Kauai could not be conclusively linked to Naval training events in the region (Southall *et al.* 2006), the spatiotemporal link between sonar exercises and the stranding does raise concern on the potential impact on the Kohala Resident population due to of sonar training nearby.

## HAWAIIAN ISLANDS STOCK

## POPULATION SIZE

Encounter data from shipboard line-transect survey of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of melon-headed whales in the Hawaii EEZ (Bradford *et al.* 2021; Table 1).

Table 1. Line-transect abundance estimates for melon-headed whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al.* 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 40,647 | 0.74 | 11,097-148,890 |
| 2010 | 8,743 | 1.01 | 1,685-45,375 |
| 2002 | 9,024 | 1.08 | 1,602-50,821 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for bottlenose dolphins from Barlow *et al.* (2015), as there is insufficient sample size to estimate $g(0)$ values for melon-headed whales. Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021), uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available estimates for each survey year. The best estimate of abundance for this stock is from the 2017 survey, or 40,647 (CV=0.74) whales. An abundance estimate of melon-headed whales is available for the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 line-transect abundance estimate (Bradford *et al.* 2021) or 23,301 melon-headed whales.

### Current Population Trend

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock. Because of the relatively large group sizes observed for melon-headed whales (average 150-200 animals), a substantial increase in abundance can be realized with very few additional sightings (one each in 2002 and 2010 versus three in 2017). Alternative approaches will be required to examine population trend in melon-headed whales.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (23,301) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality; Wade and Angliss 1997), resulting in a PBR of 233 melon-headed whales per year.

## STATUS OF STOCK

The Hawaiian Islands stock of melon-headed whales is not considered strategic under the 1994 amendments to the MMPA. The status of this stock relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Melon-headed whales are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. There have been no reports of recent mortality or serious injuries; however, there is no systematic monitoring of takes in near-shore fisheries that may take this species. Given noted bullet holes and potential line injuries on individuals from this stock, insufficient information is available to determine whether the total fishery mortality and serious injury for Hawaiian Islands melon-headed whales is insignificant and approaching zero mortality and serious injury rate. A 2004 mass-stranding of melon-headed whales in Hanalei Bay, Kauai occurred during a multi-national sonar training event around Hawaii (Southall *et al.* 2006). Although the event could not be conclusively linked to Naval training events in the region (Southall *et al.* 2006), the spatiotemporal link between sonar exercises and the stranding does raise concern on the potential impact on the Hawaiian Islands population due to its frequent use of nearshore areas within the main Hawaiian Islands.

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Revised 5/5/2021

# PYGMY KILLER WHALE (*Feresa attenuata*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

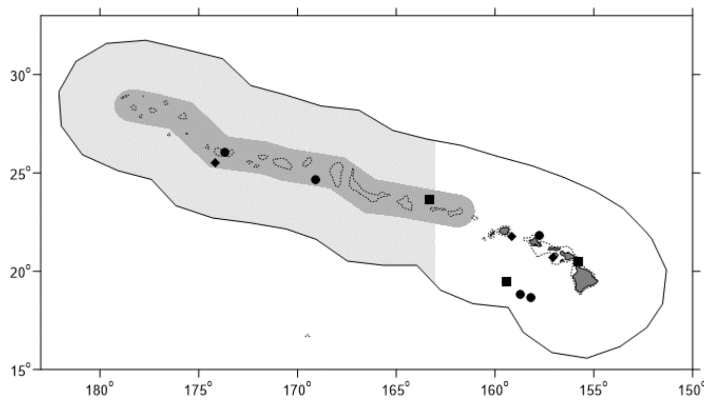


**Figure 1**. Pygmy killer whale sighting locations during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Pygmy killer whales are found in tropical and subtropical waters throughout the world (Ross and Leatherwood 1994). They are poorly known in most parts of their range. Small numbers have been taken directly and incidentally in both the western and eastern Pacific. Pryor *et al*. (1965) noted that pygmy killer whales appeared to be resident off Oahu. Resightings of several individuals over several decades indicate resident groups off Kona and leeward Oahu (McSweeney *et al*. 2009, Baird 2016) Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in three sightings of pygmy killer whales in 2002, five in 2010, and three in 2017 (Figure 1; Barlow 2006, Bradford *et al* 2017, Yano *et al*. 2018).

Pygmy killer whales in Hawaiian waters may comprise more than one demographically independent population. A 22-year study off the Hawaii Island indicates that pygmy killer whales occur there year-round and in stable social groups. Over 80% of pygmy killer whales seen off Hawaii Island have been resighted and 92% have been linked into a single social network (McSweeney *et al*. 2009). Movements have also been documented between Hawaii Island and Oahu and between Oahu and Lanai (Baird *et al*. 2011a). Satellite telemetry data from four tagged pygmy killer whales suggest this resident group remains within 20km of shore (Baird *et al* 2011a, 2011b). Encounter rates for pygmy killer whales during near shore surveys are rare, representing less only 1.7% of all cetacean encounters to since 2000 (Baird *et al*. 2013). Division of this population into a separate island-associated stock may be warranted in the future.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there is a single Pacific management stock including animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of pygmy killer whales in the Hawaii EEZ (Bradford *et al*. 2021; Table 1).

Table 1. Line-transect abundance estimates for pygmy killer whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al.* 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 10,328 | 0.75 | 2,771-38,491 |
| 2010 | 27,833 | 0.50 | 10,950-70,747 |
| 2002 | 3,854 | 0.77 | 1,015-14,640 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities derived for pygmy killer whales following the methods of Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al*. (2021*),* uses a consistent approach for estimating all abundance parameters and are considered the best available estimates for each survey year. The best estimate of abundance for this stock is based on the 2017 survey, or 10,328 (CV=0.75). A population estimate has been made for this species in the eastern tropical Pacific (Wade and Gerrodette 1993), but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al* 1995) of the 2017 abundance estimate or 5,885 pygmy killer whales within the Hawaiian EEZ.

### Current Population Trend

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for pygmy killer whales.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for pygmy killer whales stock is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (5,885) times one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.5 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 59 pygmy killer whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaii fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaii (Nitta & Henderson, 1993). A stranded pygmy killer whale from Oahu showed signs of hooking injury (Schofield 2007) and mouthline injuries have also been noted in some individuals (Baird unpublished data), though it is not known if these interactions result in serious injury or mortality. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no pygmy killer whales were observed hooked or entangled in the SSLL fishery (100% observer coverage), or in the DSLL fishery (20-21% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019). There were four additional unidentified cetaceans taken in the DSLL fishery during this period, some of which may have been pygmy killer whales.

### Other Mortality

Loud underwater sounds, such as active sonar and seismic operations, may be harmful to beaked whales (Cox *et al*. 2006) and other cetaceans, including melon-headed whales (Southall *et al*. 2006, 2013, Brownell *et al*. 2009) and pygmy killer whales (Wang and Yang 2006). The use of active sonar from military vessels has been implicated in mass strandings of beaked whales, and recent mass-stranding reports suggest some delphinids may be impacted as well. Two mass-strandings of pygmy killer whales occurred in the coastal areas of southwest Taiwan in February 2005, possibly associated with offshore naval training exercises (Wang and Yang 2006). A necropsy of one of the pygmy killer whales revealed hemorrhaging in the cranial tissues of the animal. Additional research on the behavioral response of delphinids in the presence of sonar transmissions is needed in order to understand the level of impact. No estimates of potential mortality or serious injury are available for U.S. waters.

## STATUS OF STOCK

The Hawaii stock of pygmy killer whales is not considered strategic under the 1994 amendments to the MMPA. The status of pygmy killer whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Pygmy killer whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. One pygmy killer whale stranded in the MHI has tested positive for *Morbillivirus* (Jacob *et al*. 2016). The presence of *morbillivirus* in 10 species of cetacean in Hawaiian waters, all identified as a unique strain of *morbillivirus*, (Jacob *et al*. 2016), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including cumulative impacts of disease with other stressors.

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# FALSE KILLER WHALE (*Pseudorca crassidens*): Hawaiian Islands Stock Complex – Main Hawaiian Islands Insular, Northwestern Hawaiian Islands, and Hawai'i Pelagic Stocks

## STOCK DEFINITION AND GEOGRAPHIC RANGE

False killer whales are found worldwide in tropical and warm-temperate waters (Stacey *et al.* 1994). In the North Pacific, this species is well known from southern Japan, Hawai'i, and the eastern tropical Pacific. False killer whales have been encountered during periodic shipboard line-transect surveys of the U.S. Exclusive Economic Zone (EEZ) around the Hawaiian Islands (Figure 1), and focused studies near the main and Northwestern Hawaiian Islands (NWHI) indicate that false killer whales occur in nearshore waters throughout the Hawaiian archipelago (Baird *et al.* 2008, 2013). This species also occurs in the U.S. EEZ around Palmyra and Johnston Atolls (Barlow *et al.* 2008) and American Samoa (Johnston *et al.* 2008, Oleson 2009). Genetic, photo-identification, and telemetry studies indicate there are several demographically independent populations of false killer whales throughout the Pacific and three in Hawaiian waters. Genetic analyses indicate restricted gene flow between island-associated populations of false killer whales sampled near the main Hawaiian Islands (MHI) and the NWHI, versus those in pelagic waters of the Eastern (ENP) and Central North Pacific (CNP) (Chivers *et al.* 2010; Martien 2014). The mtDNA analysis reveals strong phylogeographic patterns consistent with local evolution of haplotypes unique to false killer whales occurring nearshore within the Hawaiian Archipelago, while the nuDNA analysis suggests NWHI false killer whales are at least as differentiated from MHI animals as they are from offshore animals. Photo-ID and social network analyses of individuals seen near the MHI indicate a tight social network with no connections to false killer whales seen near the NWHI or offshore waters, and satellite telemetry collected from 27 tagged MHI false killer whales shows movements restricted to the MHI (Baird *et al.* 2010, 2012). Further analysis of photographic and genetic data from individuals seen near the MHI suggests the occurrence of 4 separate social clusters (Mahaffy *et al.* 2023). Parentage analysis of sampled individuals reveals natal group fidelity of males and females and mating within the natal group 36-64% of the time (Martien *et al*. 2019). Additional evidence for the separation of false killer whales in Hawaiian waters into three separate stocks is summarized by Oleson *et al.* (2010, 2012).

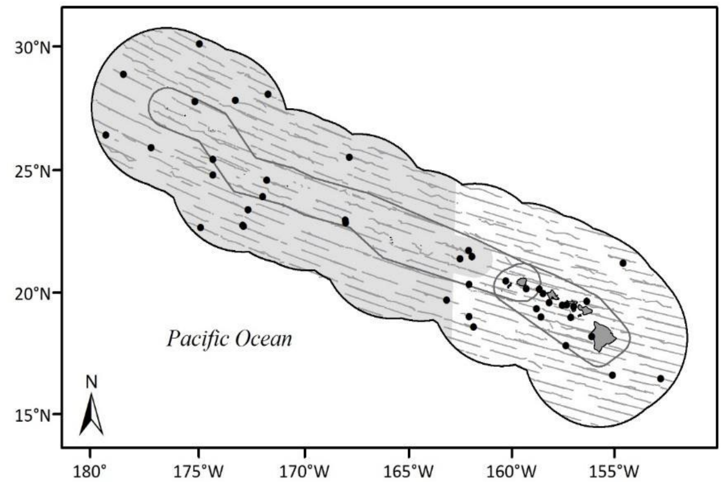


**Figure 1.** False killer whale sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al.* 2018) shipboard surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The MHI insular and NWHI false killer whale stock areas are marked by dark gray lines. Detail of stock boundaries shown in Figure 2. The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

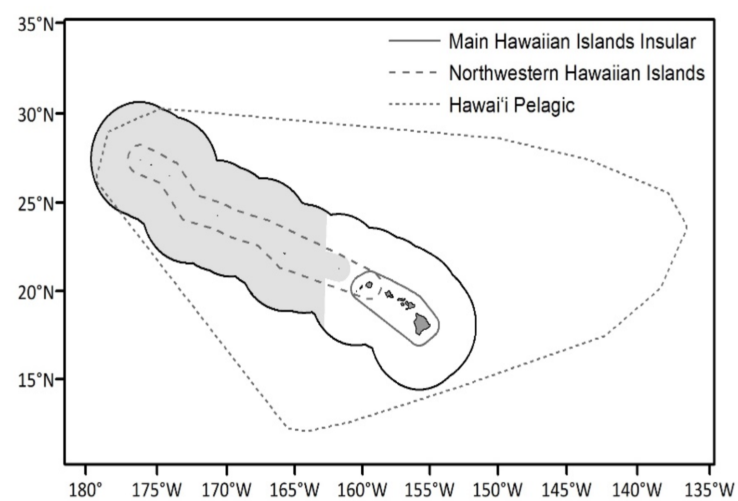


**Figure 2**. Boundaries for each of the false killer whale stock in Hawai'i: main Hawaiian Islands insular (solid gray), Northwestern Hawaiian Islands (dashed), and Hawai'i pelagic (dotted). Black line represents the U.S. EEZ. The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

Outside of the Hawaiian insular waters, population structure is also evident. Significant differences in both mtDNA and nuDNA are evident between pelagic false killer whales in the ENP and CNP (Chivers *et al.* 2010, Martien *et al*. 2014) and telemetry data from 10 pelagic false killer whales tagged within the Hawaiian Islands EEZ indicates pelagic animals there also use waters to the east of the EEZ (Oleson *et al.* 2023). Large gaps in the genetic sample distribution throughout the tropical Pacific preclude finer delineation of population structure and boundaries for pelagic populations.

The stock range and boundaries for Hawaiʻi insular stocks of false killer whales are reviewed in Bradford *et al.* (2015), and the area used for assessing the Hawaiʻi pelagic false killer whale stock (assessment area) is reviewed in Oleson *et al.* (2023) (Figure 2, there referred to as management area). The three stocks have partially overlapping ranges within the Hawaiian Islands EEZ. MHI insular false killer whales have been satellite tracked as far as 115 km from the MHI. NWHI false killer whales have been seen up to 93 km from the NWHI and near shore around Kauaʻi and Oʻahu (Baird *et al.* 2012, Bradford *et al.* 2015). Hawaiʻi pelagic stock animals have been satellite tracked to within 5.6 km of the MHI, throughout the NWHI, and east outside of the EEZ to $138^0$ W. Stock boundary descriptions are complex, but can be summarized as follows. The MHI insular stock boundary is derived from a Minimum Convex Polygon (MCP) bounded around a 72 km radius of the MHI, resulting in a boundary shape that reflects greater offshore use in the leeward portion of the MHI. The NWHI stock boundary is defined by a 93 km radius around the NWHI, with this radial boundary extended to the southeast to encompass Kauaʻi and Niʻihau. The NWHI boundary is latitudinally expanded at the eastern end of the NWHI to encompass animal movements observed outside of the 93 km radius (Figure 2). The Hawaiʻi pelagic stock has no inner boundary within the EEZ. The assessment area for the Hawaiʻi pelagic stock is defined by an MCP around all genetic, telemetry, sighting, and bycatch location data known or assumed to be of Hawaiʻi pelagic stock animals with a 35 km buffer around the points (Oleson *et al.* 2023). This assessment area extends throughout most of the Hawaiian Islands EEZ, east to $132^0$ W and south to $12^0$ N with a complex shape (Figure 2). The construction of these stock boundaries results in multiple stock overlap zones. The entirety of the MHI insular stock area is an overlap zone between the MHI insular and Hawaiʻi pelagic stocks. The entirety of the NWHI stock range is an overlap zone between NWHI and Hawaiʻi pelagic false killer whales. All three stocks overlap out to the MHI insular stock boundary between Kauaʻi and Nihoa and to the NWHI stock boundary between Kauaʻi and Oʻahu (see Figure 2).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are five Pacific Islands Region management stocks: 1) the Main Hawaiian Islands insular stock, which includes animals inhabiting waters within a modified 72 km radius around the MHI, 2) the Northwestern Hawaiian Islands stock, which includes animals inhabiting waters within a 93 km radius around the NWHI and Kauaʻi, with a latitudinal expansion of this area at the eastern end of the range, 3) the Hawaiʻi pelagic stock, which includes false killer whales inhabiting waters of the U.S. EEZ around Hawaiʻi and adjacent high seas waters, as defined by the Hawaiʻi pelagic false killer whale assessment area (Oleson *et al*. 2023), 4) the Palmyra Atoll stock, which includes animals found within the U.S. EEZ of Palmyra Atoll, and 5) the American Samoa stock, which includes animals found within the U.S. EEZ of American Samoa. Estimates of abundance, potential biological removal, and status determinations for the first three stocks are presented below. Palmyra Atoll and American Samoa stocks appear in separate reports. False killer whales are also known to occur in other areas of U.S. jurisdiction in the Pacific Islands, though stock assessments have not yet been developed for those regions.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Interactions with false killer whales, including depredation of pelagic fish catch, are identified in logbooks and NMFS observer records from Hawaiʻi pelagic longline fishing trips (Nitta and Henderson 1993, Oleson *et al.* 2010, PIRO 2015). False killer whales have been observed feeding on a variety of large pelagic fish, including mahi mahi (*Coryphaena hippurus)*, yellowfin tuna (*Thunnus albacares),* big eye tuna (*T. obesus*), albacore *(T. alalunga),* wahoo (*Acanthocybium solandri*), skipjack (*Katsuwonus pelamis*), and broadbill swordfish (*Xiphias gladius*) (Baird 2016), and they are reported to take large fish from troll lines of commercial and recreational fishermen (Shallenberger 1981). There are anecdotal reports of marine mammal interactions in the commercial Hawaiʻi shortline fishery that sets gear at Cross Seamount and possibly around the MHI. The commercial shortline fishery is licensed to sell catch through the State of Hawai'i Commercial Marine License program, and until recently, no reporting systems existed to document marine mammal interactions. Baird and Gorgone (2005) documented high rates of dorsal fin disfigurements consistent with injuries from unidentified fishing line for MHI insular stock false killer whales. Evaluation of additional individuals with dorsal fin injuries and disfigurements suggests that the interaction rate between false killer whales and various forms of hook and line gear may vary by population and social cluster, with the highest rates in the MHI insular stock (Baird *et al.* 2014). The commercial or recreational fishery or fisheries responsible for these injuries is unknown, though through examination of satellite telemetry dataset and commercial logbook effort data, it is clear that there are regions where such interactions are far more likely, including the Kohala coast of Hawaiʻi Island

and the waters extending from the southeast end of O'ahu around to the north side of Maui and to the southwest side of Lāna'i (Baird *et al.* 2021). A stranded MHI insular false killer whale in October 2013 had five fishing hooks and fishing line in its stomach, and another stranded animal in September 2016 had one fishing hook in its stomach (Bradford and Lyman 2018). Although the fishing gear is not believed to have caused the death of either whale, examinations confirm that MHI insular false killer whales consume previously hooked fish or are interacting with MHI hook and line fisheries. Many of the hooks within the whale's stomach were not consistent with those currently allowed for use within the commercial longline fisheries and could originate from a variety of nearshore fisheries. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or other fisheries because these fisheries are not monitored for protected species bycatch.

Because of high rates of false killer whale mortality and serious injury in Hawai'i-based longline fisheries, a Take Reduction Team was established in January 2010 (75 FR 2853, 19 January, 2010). The Team was charged with developing recommendations to reduce incidental mortality and serious injury of the Hawai'i pelagic, MHI insular, and Palmyra stocks of false killer whales in Hawai'i-based longline fisheries. The Team submitted a draft Take Reduction Plan (TRP) to NMFS, and NMFS published a final TRP based on the Team's recommendations (77 FR 71260, 29 November, 2012). Take reduction measures include gear requirements, time-area closures (the Southern Exclusion Zone, or SEZ), and measures to improve captain and crew response to hooked and entangled false killer whales. The seasonal contraction of the Longline Exclusion Zone (LLEZ) around the MHI was also eliminated. The TRP became effective December 31, 2012, with gear requirements effective February 27, 2013. Adjustments to bycatch estimation methods were implemented for 2013 to account for changes in fishing gear and captain training intended to reduce the false killer whale serious injury rate (McCracken 2015).

There are two distinct longline fisheries based in Hawai'i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the LLEZ around the MHI and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50-nmi radius around the NWHI. In August, 2016 the PMNM area was expanded to extend to the 200-nmi EEZ boundary west of 163° W.

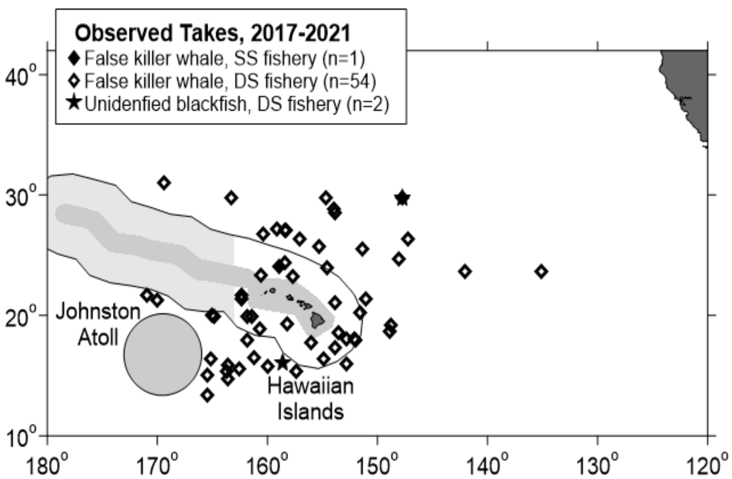


**Figure 3.** Locations of observed false killer whale takes within the shallow-set fishery (filled diamond), deep-set fishery (open diamonds), and possible takes (blackfish) of this species (closed stars) in the Hawaii-based longline fisheries, 2017-2021. Some take locations overlap. Stock boundaries for false killer whales are not shown. Solid lines represent the U.S. EEZ. Gray shading notes areas closed to longline fishing**.**

Between 2017 and 2021, one false killer whale was observed hooked or entangled in the SSLL fishery (100% observer coverage), and 54 false killer whales were observed taken in the DSLL fishery (15-21% observer coverage) (Bradford 2018, 2020, 2021, 2023, in review) (Figure 3). The severity of injuries resulting from interactions with longline gear is based on an evaluation of the observer's description of each interaction and follows the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2023b). In the DSLL fishery, 1 false killer whale was taken within the overlap area of the pelagic and NWHI stocks. Stock identity is not known for any bycaught whales, though those outside of the stock overlap areas are assumed to be Hawai'i pelagic stock animals. Of the 54 bycaught whales, 7 were found dead, 37 were considered seriously injured, 8 non-seriously injured, and 2 had injuries with a severity that could not be determined based on the information provided by the observer. Two additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) were also taken in the DSLL fishery outside of the Hawaiian Islands EEZ, with one considered seriously injured and one not seriously injured.

The SEZ, a large triggered closure area south of the MHI implemented under the TRP, was closed following the trigger of 2 serious injuries within the Hawaiian Islands EEZ in November 2018. This closure remained in effect through the remainder of calendar year 2018. Following re-opening of the SEZ on January 1, 2019, the SEZ was again closed in February 2019 following a serious injury and a mortality within the Hawaiian Islands EEZ. Following the closure there were 3 additional serious injuries within the Hawaiian Islands EEZ in 2019. The SEZ remained closed until August 2020. Following an increase in the trigger to 4 whales in 2021, an SEZ closure was triggered after a fourth and fifth serious injuries within the EEZ were reported in December 2021, though the SEZ was not closed given the closure would not have been effective until after the automatic reopening date at the start of the new calendar year.

**Table 1.** Summary of available information on incidental mortality and serious injury (MSI) of false killer whales (FKW) and unidentified blackfish (UB, false killer whale or short-finned pilot whale) in commercial longline fisheries, by stock and EEZ area, as applicable (McCracken and Cooper 2022b, 2023). Mean annual takes are presented for 2017-2021. Information on observed takes (T) and MSI is included. UB are prorated as either FKW or short-finned pilot whales based on distance from shore (McCracken 2010). CVs are estimated based on the combined variances of annual FKW and UB take estimates and the relative density estimates for each stock within the overlap zones. Values of '0' presented with no further precision are based on observation at 100% coverage and are not estimates.

| | | | | Observed takes | | Estimated MSI (CV) | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | FKW T/MSI<br>UB T/MSI | | Hawai'i Pelagic Stock | | | |
| **Fishery Name** | **Year** | **Data Type** | **Percent Observer Coverage** | **Outside U.S EEZ** | **Within Hawaiian Islands EEZ** | **Assessment Area Outside U.S EEZ** | **Assessment Area Within Hawaiian Islands EEZ** | **MHI insular Stock** | **NWHI Stock** |
| Hawaii-based deep-set longline fishery | 2017 | Observer data | 20% | 6/5†<br>0 | 2/1<br>0 | 29.7 (0.4) | 8.4 (0.7) | 0.1 (0.8) | 0 (2.1) |
| | 2018 | | 18% | 8/5<br>1/1 | 4/4<br>0 | 30.6 (0.4) | 12.3 (0.5) | 0.1 (0.6) | 0 (2.0) |
| | 2019 | | 21% | 9/7<br>1/0 | 6/5<br>0 | 38.7 (0.3) | 26.0 (0.4) | 0.0 (0.5) | 0.6 (2.0) |
| | 2020 | | 15% | 3/2†<br>0 | 1/1<br>0 | 14.2 (0.5) | 5.1 (0.9) | 0.0 (0.9) | 0 (2.2) |
| | 2021 | | 18% | 10/9†<br>0 | 5/5<br>0 | 37.0 (0.4) | 32.1 (0.4) | 0.2 (0.5) | 0.1 (2.0) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | | **30.0 (0.2)** | **16.8 (0.2)** | **0.1 (0.3)** | **0.2 (1.6)** |
| Hawaii-based shallow-set longline fishery | 2017 | Observer data | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| | 2019 | | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| | 2020 | | 100% | 0<br>0 | 1/1<br>0 | 0 | 1 | 0 | 0 |
| | 2021 | | 100% | 0<br>0 | 0<br>0 | 0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | | | **0** | **0.2** | **0** | **0** |
| **Minimum Total annual takes by stock (2017-2021)** | | | | | | **47.0 (0.2)** | | **0.1 (0.3)** | **0.2 (1.6)** |

† Injury severity could not be determined based on information collected by the observer. Injury severity is prorated (see text).

The total estimated number of dead or seriously injured whales is calculated based on observer coverage rate, the location of the observed take (*i.e.* within or outside of the Hawai'i pelagic stock management area), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process was subset into several periods, as described in McCracken and Cooper (2022a). Prior to the implementation of the FKW TRP, for the period 2008 to 2012, the rate of dead and seriously injured false killer whales was 93% (McCracken 2014). The implementation of weak hooks under the TRP was intended to reduce the serious injury rate in the deep-set fishery, and as such, the proportion of dead and seriously injured whales versus non-serious injuries is calculated annually based on the injury status of observed takes since the implementation of the TRP in 2013 (McCracken 2019).

Complete assessment of human-caused mortality within the full Hawai‘i pelagic false killer whale assessment area requires information on bycatch in foreign fleets. Foreign longline fleets operate within the tropical Pacific, including immediately outside of the Hawaiian Islands EEZ. Although the magnitude of foreign longline effort near the Hawaiian Islands EEZ is thought to be relatively low compared to that of the Hawai‘i-based fleet, there is considerable effort to the southwest of the EEZ and north of $30^0$ N on the east side of the islands (based on Global Fishing Watch data). The Western and Central Pacific Fisheries Commission (WCPFC) has collated 76 interactions with false killer whales in the western and central Pacific across the member fleets, including reports from the Hawai‘i-based vessels from 2015 to 2020 (Williams *et al*. 2021). However, the WCPFC has not developed estimates of total bycatch for any segment of the fleet "given the low levels and imbalanced nature of observer coverage" (Peatman and Nicols 2020). Commercial fishing within the eastern portion of the Hawai‘i pelagic false killer whale assessment area is managed by the Inter-American Tropical Tuna Commission, though similar concerns about observer coverage have so far precluded any bycatch estimates for false killer whales in this region. The mortality rate of bycaught animals in foreign longline fleets may also be higher than in the U.S. fleet given the bycatch mitigation measures in place for the Hawai‘i-based fleet, leading to additional uncertainty in the magnitude of the impact on the stock (Oleson *et al.* 2023).

Biological samples or individual animal photographs are required to assign a take to a specific stock. Very few observed takes are identified to stock, as collection of such information is very rare. The pelagic stock is known to interact with longline fisheries based on a small number of genetic samples obtained by fishery observers (Chivers *et al.* 2010) both inside and outside of the Hawaiian Islands EEZ. No samples or photographs have been collected that can conclusively assign takes to stock within the MHI insular-pelagic overlap zone or the NWHI-pelagic stock overlap zone. However, MHI insular and NWHI false killer whales have been documented via telemetry to move far enough offshore to reach longline fishing areas (Bradford *et al.* 2015), and MHI insular stock animals have high rates of dorsal fin disfigurements consistent with injuries from unidentified fishing line (Baird and Gorgone 2005, Baird *et al.* 2014). When takes cannot be assigned to stock, annual bycatch estimates are prorated to stock using the following process. Takes of unidentified blackfish are prorated to false killer whale and short-finned pilot whale based on distance from shore (McCracken 2010), given patterns of previous bycatch for each species. Following proration of unidentified blackfish takes to species, Hawaiian Islands EEZ and high-seas estimates of false killer whale take are calculated by summing the annual false killer whale take and the annual blackfish take prorated as false killer whale within each region (McCracken 2020). Takes within the shallow-set longline fishery are assigned to the stock area in which they were observed. Estimated takes in the deep-set fishery within the Hawaiian Islands EEZ are apportioned to each stock area by first allocating take to each area based on relative annual fishing effort (by set) in that area. If an observed take occurred within the MHI-pelagic or NWHI-pelagic overlap zones, the take was assigned to that zone and the remaining estimated bycatch was assigned to stock areas as previously described. For both the shallow-set and deep-set fisheries, stock area bycatch estimates are then multiplied by the relative density of each stock within the stock area to estimate stock-specific bycatch for each year. Uncertainty in stock-specific bycatch estimates combines variances of total annual false killer whale bycatch and the fractional variance of false killer whale density according to which stock is being estimated. Enumeration of fishing effort within stock overlap zones is assumed to be known without error. Proration of unidentified blackfish takes and of false killer whale takes within the stock overlap zones introduces unquantified uncertainty into the bycatch estimates, but until methods of determining stock identity for animals observed taken within the overlap zone are available, and all animals taken can be identified to species and stock (*e.g.,* with photos or tissue samples), these proration approaches are needed to ensure that potential impacts to all stocks are assessed in the overlap zones. Based on this approach, estimates of annual mortality and serious injury of false killer whales, by stock and EEZ area, are shown in Table 1.

## MAIN HAWAIIAN ISLANDS INSULAR STOCK

## POPULATION SIZE

Bradford *et al.* (2018) used encounter data from dedicated and opportunistic surveys for MHI insular false killer whales from 2000 to 2015 to generate annual mark-recapture estimates of abundance. Due to spatiotemporal biases imposed by sampling constraints, annual estimates reflected the abundance of MHI insular false killer whales within the surveyed area in that year, and therefore could not be considered indicative of total population size every year. The abundance estimate for 2015 was 167 (CV = 0.14). Annual estimates over the 16-year survey period ranged from 144 to 187 animals and are similar to multi-year aggregated estimates published previously (Oleson *et al.* 2010).

### Minimum Population Estimate

The minimum population estimate for the MHI insular stock of false killer whales is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2015 abundance estimate (from Bradford *et al.* 2018), or 149 false killer whales.

**Current Population Trend**

Reeves *et al.* (2009) suggested that the MHI insular stock of false killer whales may have declined between 1989 and 2007, based on sightings data collected near Hawaii using various methods. Baird (2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the main Hawaiian Islands between 1994 and 2003 (Mobley *et al.* 2000). Sighting rates during these surveys showed a statistically significant decline that could not be attributed to any weather or methodological changes. The Status Review of MHI insular false killer whales (Oleson *et al.* 2010) presented a quantitative analysis of extinction risk using a Population Viability Analysis (PVA). The modeling exercise was conducted to evaluate the probability of actual or near extinction, defined as a population reduced to fewer than 20 animals, given measured, estimated, or inferred information on population size and trends, and varying impacts of catastrophes, environmental stochasticity and Allee effects. All plausible models indicated the probability of decline to fewer than 20 animals within 75 years was greater than 20%. Though causation was not evaluated, all plausible models indicated the population had declined since 1989, at an average rate of -9% per year (95% probability intervals -5% to -12.5%), though some two-stage models suggested a lower rate of decline (Oleson *et al.* 2010). Annual abundance estimates in Bradford *et al.* (2018) are not appropriate for evaluating population trends, as the study area varied by year, and each annual estimate represents only animals present in the study area within each year.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the MHI insular false killer whale stock is calculated as the minimum population estimate (149) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.1 (for a stock listed as Endangered under the ESA and with minimum population size less than 1500 individuals; Taylor *et al.* 2000) resulting in a PBR of 0.3 false killer whales per year, or approximately one animal every 3.3 years.

## STATUS OF STOCK

The status of the MHI insular stock of false killer whales relative to OSP is unknown, although this stock appears to have declined during the past two decades (Oleson *et al.* 2010, Reeves *et al.* 2009; Baird 2009). MHI insular false killer whales are listed as "endangered" under the Endangered Species Act (1973) (77 FR 70915, 28 November, 2012). Because MHI insular false killer whales are formally listed as "endangered" under the ESA, they are automatically considered as a "depleted" and "strategic" stock under the MMPA. For the 5-yr period prior to the implementation of the TRP, the average estimated mortality and serious injury to MHI insular stock false killer whales (0.21 animals per year) exceeded the PBR (0.18 animals per year). Prior to the TRP, a seasonal contraction to the LLEZ potentially exposed a significant portion of the offshore range of the stock to longline fishing. Following implementation of the TRP, a significant portion of the recognized stock range is inside of the expanded year-round LLEZ around the MHI, providing significant protection for this stock from longline fishing. For the most recent 5-yr period, the estimate of mortality and serious injury (0.10) is below the PBR (0.30). The total fishery mortality and serious injury for the MHI insular stock of false killer whales cannot be considered to be insignificant and approaching zero, as it is ≥ 10% of PBR. The stock appears to be declining (Oleson *et al.*2010), though the cause of that decline has not been thoroughly assessed.

## OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

The Status Review report produced by the Biological Review Team (BRT) (Oleson *et al.* 2010, amended in Oleson *et al.* 2012) found of the 29 identified threats to the population, the effects of small population size, including inbreeding depression and Allee effects; exposure to environmental contaminants (Ylitalo *et al.* 2009); competition for food with commercial fisheries (Boggs and Ito, 1993, Reeves *et al.* 2009); and hooking, entanglement, or intentional harm by fishermen to be the most substantial threats to the population. There is significant geographic overlap between various nearshore fisheries and evidence of interactions with hook-and-line gear (Baird *et al.* 2015, 2021), such that these fisheries may pose a threat to the stock. Six MHI insular false killer whales stranded between 2010 and 2021, including 4 from cluster 3 (Baird *et al.* 2023), a high rate for a single social cluster. High concentrations of polychlorinated biphenyls (PCBs) exceeding those proposed to cause adverse health effects (Kannan *et al.* 2020) were measured from 29 of 41 sampled individuals in the MHI insular stock (Kratofil *et al.* 2020). PCB concentrations from four stranded individuals within this population all revealed levels more than twice the highest suggested health threshold for PCBs, and had the highest levels of any sampled whales in the study. Differences in contaminant loads for various contaminant classes are also evident among social clusters, suggesting differences in exposure or consumption of contaminated prey based on preferred foraging regions (Kratofil *et al.* 2020).

## HAWAII PELAGIC STOCK

### POPULATION SIZE

Encounter data from shipboard line-transect surveys conducted throughout the central Pacific were used to estimate the density and abundance of all pelagic false killer whales in the central Pacific (Bradford *et al*. 2020). Density data from the central Pacific modeled area were used to extract the abundance of the Hawai‘i pelagic false killer whale stock based on the assessment area (Oleson *et al.* 2023, there referred to as management area). The best estimate of abundance for the Hawai‘i pelagic stock of false killer whales is 5,528 (CV = 0.35). Previous stock assessment reports for this stock provided abundance within the Hawaiian Islands EEZ. For comparison to prior reports, the same process was used to estimate abundance for the portion of the assessment area that is within the EEZ, resulting in an estimate of 2,038 (CV = 0.35).

Bradford *et al*. (2020) produced design- and model-based abundance estimates for false killer whales within each survey year for the full Hawaiian Islands EEZ, which can be compared to the abundance estimate within the EEZ portion of the assessment area above. While on average, the estimates are broadly similar between the two approaches, annual design-based estimates show much greater variability between years than the model-based estimates (Figure 4). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Bradford *et al*. (2020) found through simulation that the low sighting rate in 2002 and high sighting rate in 2017 could be explained by encounter rate variation. Although a ‘year’ covariate was tested during model development, it was not selected as a significant variable. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best approach. Previous abundance estimates from the Hawaiian Islands EEZ and central Pacific (e.g., Barlow 2006, Barlow and Rankin 2007, Becker *et al.* 2012, Bradford *et al.* 2014, Forney *et al.* 2015) used subsets of the dataset and different line-transect parameters than those used by Bradford *et al.* (2020), such that these estimates have been superseded by the estimates presented here.

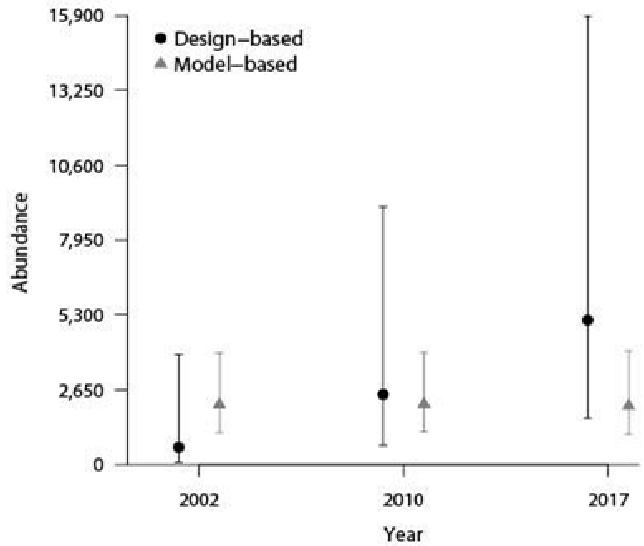


**Figure 4.** Comparison of design-based (circles) and model-based (triangles) estimates of abundance for false killer whales for each survey year (2002, 2010, 2017) (Bradford *et al*. 2020).

The reanalysis may still be subject to potential bias due to vessel attraction as described by Bradford *et al.* (2014), who reported that most (64%) false killer whale groups seen during the 2010 survey were moving toward the vessel when detected by the visual observers. Together with an increase in sightings close to the trackline, these behavioral data suggest vessel attraction is likely occurring and may be significant. Similar to the treatment of the detection function in Bradford *et al.* (2014, 2015), new model-based estimates use Beaufort-specific effective strip width estimates (following Barlow *et al*. 2015) derived from an analysis that used a half-normal model to minimize the effect of vessel attraction. The abundance estimate may still be positively biased due to vessel attraction because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. The acoustic and visual data suggests vessel attraction (Bradford *et al.* 2014), though the extent of any bias created by this movement is unknown.

### Minimum Population Estimate

The minimum population estimate for the Hawai‘i pelagic stock of false killer whales is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 model-based abundance estimate. Using the full assessment area, the minimum population estimate is 4,152. As noted in Oleson *et al.* (2023), gaps in survey coverage and in the distribution of genetic samples preclude placement of a precise stock boundary, such that the Nmin for the assessment area may not represent the entire stock range. However, given the available data, the assessment area represents what is currently known about the stock range, and as such, the Nmin derived from it is the best available minimum population estimate for Hawai‘i pelagic false killer whales. For comparison to previous EEZ-based assessments of this stock, it is useful to calculate the minimum population estimate based only on the EEZ portion of the assessment area, or 1,531 Hawai‘i pelagic false killer whales.

**Current Population Trend**

Although a ‘year’ covariate was evaluated during model development and not included during the model selection process, the final model-based abundance estimates for false killer whales provided by Bradford *et al.* (2020) do not explicitly examine population trend other than that driven by environmental factors. In contrast, annual design-based estimates suggest an increase in population size within the Hawaiian Islands EEZ; however, these changes can be largely explained by random variability in encounter rate common for species like false killer whales with low density and patchy distribution. Examination of population trend for the Hawai‘i pelagic stock of false killer whales requires additional data inside and outside of the Hawaiian Islands EEZ.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level is calculated as the minimum population estimate (4,152) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.44, resulting in a PBR of 36 false killer whales per year. An intermediate value of the recovery factor was chosen to reflect different levels of uncertainty around mortality and serious injury estimates across the assessment area. Outside of the EEZ, there is significant uncertainty in the magnitude of mortality and serious injury in foreign longline fleets operating within the assessment area, as there are presently no mortality and serious injury estimates for any foreign fleet. Therefore, a recovery factor of 0.4 is appropriate in this portion of the assessment area given the unknown status of the stock and the unknown total mortality and serious injury rate (NMFS 2023a). However, within the EEZ portion of the assessment area, the mortality and serious injury CV is less than 0.3 so a recovery factor of 0.5 is appropriate (NMFS 2023a). Taken together, a final prorated recovery factor was calculated based on the proportion of total stock abundance inside and outside the EEZ, resulting in a final recovery factor of 0.44 for the stock. For comparison to PBR values reported in previous assessments for this stock, an EEZ-only-PBR was also calculated using the minimum population estimate for the portion of the assessment area within the EEZ (1,531), and a recovery factor of 0.5, resulting in an EEZ-only-PBR of 15 Hawai‘i pelagic false killer whales per year.

## STATUS OF STOCK

The status of the Hawai‘i pelagic stock of false killer whales relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. This stock is not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Following the NMFS Guidelines for Assessing Marine Mammal Stocks (NMFS 2023a), the status of this transboundary stock of false killer whales is provided based on the assessment area, defined using all available biological data for this population. Total 5-year mean mortality and serious injury of the stock for 2017-2021 (47) is more than PBR (36); therefore, this stock is considered a “strategic stock” under the MMPA. Total fishery mortality and serious injury for the Hawaii pelagic stock of false killer whales cannot be considered insignificant and approaching zero (*i.e.*, less than 10% of PBR, or 3.6 animals per year). The estimated mortality and serious injury of Hawai‘i pelagic false killer whales in 2019 and 2021 were the highest recorded since before the TRP was implemented, with annual mortality and serious injury rates exceeding 60 animals per year. Take rates of false killer whales by the deep-set longline fishery outside of the EEZ continue to remain significantly higher since the TRP was implemented. Given survey and sampling limitations described in Oleson *et al*. (2023), the assessment area may not account for all animals in this stock, but does represent the best information available at this time. Although the assessment area may not represent the entire stock range, the abundance, PBR, and mortality and serious injury estimated for this area are subject to similar bias.

## OTHER FACTORS THAT MAY BE CAUSING A DECLINE OR IMPEDING RECOVERY

The mean concentration of polychlorinated biphenyls (PCBs) in all Hawai‘i false killer whale populations, including individuals from the pelagic stock (Kratofil *et al.* 2020), has been shown to exceed the level proposed to cause adverse health effects in other cetaceans (Kannan *et al.* 2020).

## NORTHWESTERN HAWAIIAN ISLANDS STOCK

### POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were reevaluated for each survey year, resulting in the following abundance estimates of NWHI false killer whales (Bradford *et al*. 2020; Table 3).

**Table 2.** Design-based line-transect abundance estimates for Northwestern Hawaiian Islands false killer whales derived from surveys of the entire Hawaiian Islands EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2020).

| Year | Design-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 477 | 1.71 | 48 - 4,712 |
| 2010 | 878 | 1.15 | 145 - 5,329 |
| 2002 | - | | |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for false killer whales using the methods of Barlow *et al*. (2015). Although a previous 2010 estimate for this stock was published using a subset of this data (Bradford *et al*. 2014), Bradford *et al*. (2020) used a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available. There were no sightings of false killer whales in the NWHI stock area in 2002. The reanalysis may still be subject to potential bias due to vessel attraction as described by Bradford *et al.* (2014), who reported that most (64%) false killer whale groups seen during the 2010 HICEAS survey were moving toward the vessel when detected by the visual observers. Together with an increase in sightings close to the trackline, these behavioral data suggest vessel attraction is likely occurring and may be significant. Bradford *et al.* (2014, 2015, 2020) used a half-normal model to minimize the effect of vessel attraction, because groups originally outside of the survey strip, and therefore unavailable for observation by the visual survey team, may have moved within the survey strip and been sighted. There is some suggestion of such attractive movement within the acoustic and visual data (Bradford *et al.* 2014), though the extent of any bias created by this movement is unknown. The best estimate of current abundance is 477 (CV=1.71) false killer whales from the 2017 survey (Bradford *et al*. 2020).

#### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 abundance estimate for the NWHI stock (Bradford *et al.* 2020), or 178 false killer whales.

#### Current Population Trend

The two available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding evaluation of population trend for this stock.

### CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species Hawaiian waters.

### POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the NWHI false killer whale stock is calculated as the minimum population estimate (178) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.40 (for a stock of unknown status, with a Hawaiian Islands EEZ mortality and serious injury rate CV > 0.8; Wade and Angliss 1997), resulting in a PBR of 1.43 false killer whales per year.

### STATUS OF STOCK

The status of false killer whales in NWHI waters relative to OSP is unknown, and insufficient data exist to evaluate abundance trends. The mean concentration of polychlorinated biphenyls (PCBs) in all Hawaiʻi false killer whale populations (Kratofil *et al.* 2020), including individuals from the NWHI stock, has been shown to exceed the level proposed to cause adverse health effects in other cetaceans (Kannan *et al.* 2020). Biomass of some false killer whale prey species may have declined around the NWHI (Oleson *et al.* 2010, Boggs and Ito 1993, Reeves *et al.* 2009), though waters within the original PMNM have been closed to commercial longlining since 1991 and to other fishing since 2006. This stock is not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The rate of mortality and serious injury to NWHI false killer whales (0.16) is less than the PBR (1.43 animals per year), though cannot be considered to be insignificant and approaching zero (<10% of PBR). A very small portion of the recognized stock range lies outside of the newly expanded PMNM and the expanded LLEZ, such that this stock is likely not exposed to high levels of fishing effort because commercial and recreational fishing is prohibited within Monument waters and longlines are excluded from the majority of the stock range.

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Revised 12/18/2012

# FALSE KILLER WHALE (*Pseudorca crassidens*): Palmyra Atoll Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

False killer whales are found worldwide mainly in tropical and warm-temperate waters (Stacey *et al.*. 1994). In the North Pacific, this species is known from southern Japan, Hawaii, and the eastern tropical Pacific. Four on-effort sightings of false killer whales were recorded during a 2005 shipboard survey of the U.S. Exclusive Economic Zone (EEZ) of Palmyra Atoll (Figure 1; Barlow & Rankin 2007). This species also occurs in U.S. EEZ waters around Hawaii (Barlow 2006, Bradford *et al.*. 2012), Johnston Atoll (NMFS/PIR/PSD unpublished data), and American Samoa (Johnston *et al.*. 2008, Oleson 2009).

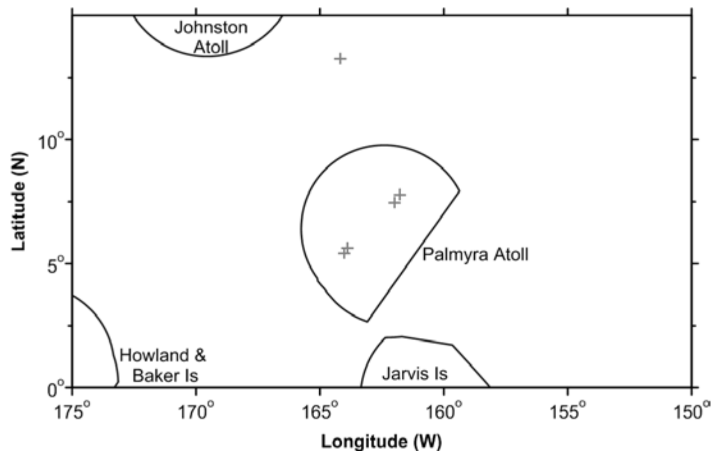


**Figure 1.** False killer whale on-effort sighting locations during a 2005 standardized shipboard survey of the Palmyra U.S. EEZ and pelagic waters of the central Pacific south of the Hawaiian Islands (gray crosses, Barlow and Rankin 2007). Solid lines represent approximate boundary of U.S. EEZs.

Genetic analyses indicate restricted gene flow between false killer whales sampled near the main Hawaiian Islands (MHI), the Northwestern Hawaiian Islands (NWHI), and in pelagic waters of the Eastern (ENP) and Central North Pacific (CNP) (Chivers *et al.*. 2007, 2010, Martien *et al.*. 2011). The Palmyra Atoll stock of false killer whales remains a separate stock, because comparisons amongst false killer whales sampled at Palmyra Atoll and those sampled from the insular stock of Hawaii and the pelagic ENP revealed restricted gene flow, although the sample size remains low for robust comparisons (Chivers *et al.*. 2007, 2010). NMFS will obtain and analyze additional tissue samples from Palmyra and the broader tropical Pacific for genetic studies of stock structure, and will evaluate new information on stock ranges as it becomes available.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are currently five Pacific Islands Region management stocks (Chivers *et al.*. 2008, Martien *et al.*. 2011): 1) the Hawaii insular stock, which includes animals inhabiting waters within 140 km (approx. 75 nmi) of the main Hawaiian Islands, 2) the Northwestern Hawaiian Islands stock, which includes false killer whales inhabiting waters within 93 km (50 nmi) of the NWHI and Kauai, 3) the Hawaii pelagic stock, which includes false killer whales inhabiting waters greater than 40 km (22 nmi) from the main Hawaiian Islands, 4) the Palmyra Atoll stock, which includes false killer whales found within the U.S. EEZ of Palmyra Atoll, and 5) the American Samoa stock, which includes false killer whales found within the U.S. EEZ of American Samoa. Estimates of abundance, potential biological removal, and status determinations for the Palmyra Atoll stock are presented below; the Hawaii Stock Complex and American Samoa Stocks are presented in separate reports.

## POPULATION SIZE

A 2005 line transect survey in the U.S. EEZ waters of Palmyra Atoll produced an estimate of 1,329 (CV = 0.65) false killer whales (Barlow & Rankin 2007). This is the best available abundance estimate for false killer whales within the Palmyra Atoll EEZ.

### Minimum Population Estimate

The log-normal 20th percentile of the 2005 abundance estimate for the Palmyra Atoll EEZ (Barlow & Rankin 2007) is 806 false killer whales.

### Current Population Trend

No data are available on current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Palmyra Atoll waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Palmyra Atoll false killer whale stock is calculated as the minimum population size (806) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.40 (for a stock of unknown status with a mortality and serious injury rate CV >0.80; Wade and Angliss 1997), resulting in a PBR of 6.4 false killer whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Interactions with false killer whales, including depredation of catch, have been identified in logbooks and NMFS observer records from Hawaii pelagic longlines (Nitta and Henderson 1993, NMFS/PIR unpublished data). False killer whales have also been observed feeding on mahi mahi, *Coryphaena hippurus*, and yellowfin tuna, *Thunnus albacares*, and they have been reported to take large fish from the trolling lines of both commercial and recreational fishermen (Shallenberger 1981).

The Hawaii-based deep-set longline (DSLL) fishery targets primarily tunas and operate within U.S. waters and on the high seas near Palmyra Atoll. Between 2006 and 2010, one false killer whale was observed taken in the DSLL fishery within the Palmyra EEZ (≥20% observer coverage) (Forney 2011). Based on an evaluation of the observer's description of each interaction and following the most recently developed criteria for assessing serious injury in marine mammals (Andersen *et al.*. 2008), the single false killer whale taken in the Palmyra EEZ was considered seriously injured (Forney 2011). The total estimated annual and 5-yr average mortality and serious injury of cetaceans in the DSLL fishery operating around Palmyra (with approximately 20% coverage) are reported by McCracken (2011) (Table 1). Although M&SI estimates are shown as whole numbers of animals, the 5-yr average M&SI is calculated based on the unrounded annual estimates.

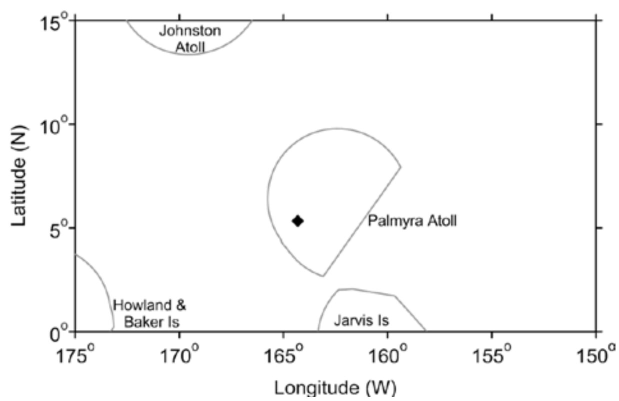


**Figure 2.** Locations of observed false killer whale takes in the Hawaii-based deep-set longline fishery, 2006-2010. Solid gray lines represent the U.S. EEZ. Fishery descriptions are provided in Appendix 1.

Because of high rates of false killer whale mortality and serious injury in Hawaii-based longline fisheries, a Take-Reduction Team (TRT) was established in January 2010 (75 FR 2853, 19 January 2010). The scope of the TRT was to reduce mortality and serious injury in the Hawaii pelagic, main Hawaiian Islands insular, and Palmyra stocks of false killer whales and across the DSLL and SSLL fisheries. The Team submitted a Draft Take-Reduction Plan to NMFS for consideration and NMFS has recently published regulations based on this TRP (77 FR 71260, 29 November, 2012). The Team chose to exclude the Palmyra Atoll stock in the final implementation of the Plan due to low levels of M&SI of this stock for the past 5 years.

**Table 1.** Summary of available information on incidental mortality and serious injury of false killer whales (Palmyra Atoll stock) in the Hawaii-based longline fishery (McCracken 2011). Mean annual takes are based on 2006-2010 estimates unless otherwise indicated. Information on all observed takes (T) and combined mortality events & serious injuries (MSI) is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Observed total interactions (T) and mortality events and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of false killer whales in the Palmyra Atoll EEZ | |
|---|---|---|---|---|---|
| | | | | Observed T/MSI | Estimated Mean Annual Takes (CV) |
| Hawaii-based deep-set longline fishery | 2006 | observer data | 22% | 0/0 | 0 (-) |
| | 2007 | | 20% | 1/1 | 2 (0.7) |
| | 2008 | | 22% | 0/0 | 0 (-) |
| | 2009 | | 20% | 0/0 | 0 (-) |
| | 2010 | | 21% | 0/0 | 0 (-) |
| **Minimum total annual takes within U.S. EEZ** | | | | | **0.3 (1.7)** |

**STATUS OF STOCK**

The status of false killer whales in Palmyra Atoll EEZ waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this stock. They are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The rate of mortality and serious injury to false killer whales within the Palmyra Atoll EEZ in the Hawaii-based longline fishery (0.3 animals per year) does not exceed the PBR (6.4) for this stock and thus, this stock is not considered "strategic" under the MMPA. The total fishery mortality and serious injury for Palmyra Atoll false killer whales is less than 10% of the PBR and, therefore, can be considered to be insignificant and approaching zero. Additional injury and mortality of false killer whales is known to occur in U.S and international longline fishing operations in international waters, and the potential effect on the Palmyra stock is unknown.

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Revised 01/15/2011

# FALSE KILLER WHALE *(Pseudorca crassidens)*: American Samoa Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

False killer whales are found worldwide mainly in tropical and warm-temperate waters (Stacey et al. 1994). The species is well-documented throughout the tropical and sub-tropical south Pacific, from Papua New Guinea and Australia to the line islands (Reeves et al. 1999). The species has been taken in the drive hunt in the Solomon Islands (Reeves et al 1999). ). During small-boat surveys from 2003 to 2006 in the waters surrounding the island of Tutuila, American Samoa, false killer whales were observed during summer surveys on five occasions (Johnston et al 2008). During a shipboard survey in 2006 false killer whales were also encountered just north of the island of Ta‘u, in the Manu’a Group within American Samoa (Johnston et al. 2008). Two false killer whales were entangled near 40-Fathom Bank south of the islands by the American Samoa-based longline fishery in 2008 (Oleson 2009), indicating some false killer whales maintain a more pelagic distribution. Five genetic samples collected near Tutuila are available for comparison to other false killer whale populations throughout the Pacific (Johnston et al 2008). For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are four Pacific management stocks: 1) The Hawaii Insular Stock, which includes animals found within the 25-75 nmi longline exclusion boundary surrounding the main Hawaiian Islands, 2) The Hawaii Pelagic Stock, which includes animals found within the U.S. EEZ of the Hawaiian Islands but outside the 25-75 nmi longline exclusion zone, 3) The Palmyra Stock, which includes animals found within the U.S. EEZ of the Palmyra Atoll, and 4) The American Samoa Stock, which includes animals found within the U.S. EEZ American Samoa (this report).

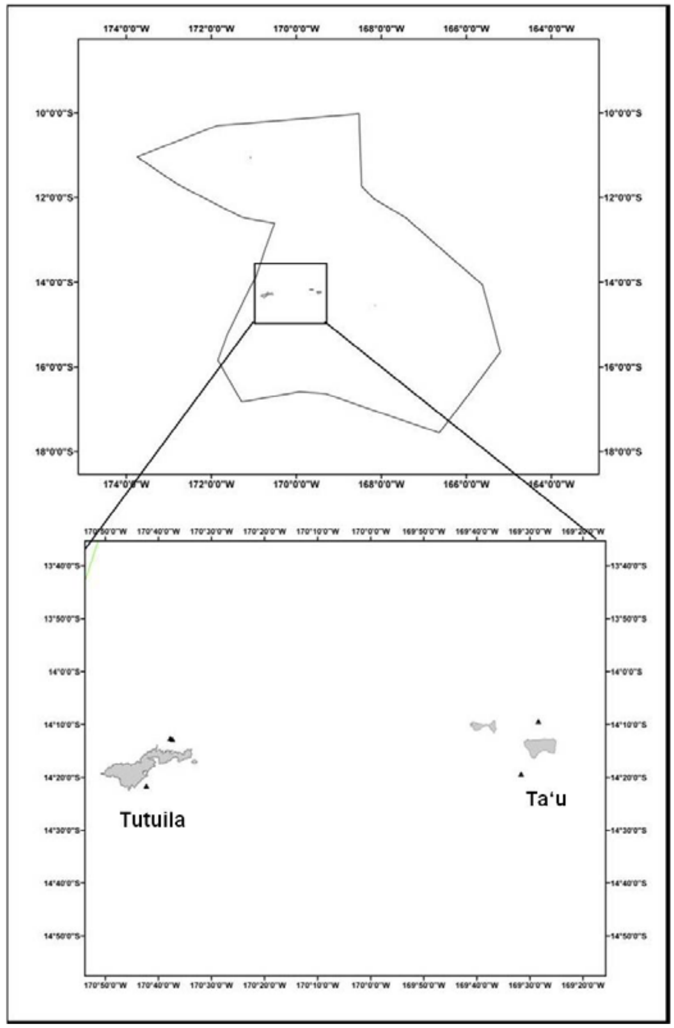


Figure 1. False killer whale sightings during visual surveys from 2003-2006 (Johnston et al 2008).

## POPULATION SIZE

No abundance estimates are currently available for false killer whales in U.S. EEZ waters of American Samoa; however, density estimates for false killer whales in other tropical Pacific regions can provide a range of likely abundance estimates in this unsurveyed region. Published estimates of false killer whales (animals per km$^2$) in the Pacific are: 0.0002 (CV= 0.93) for the U.S. EEZ of the Hawaiian Islands (Barlow and Rankin 2007); 0.0038 (CV=0.65) for the U.S. EEZ around Palmyra, (Barlow and Rankin 2007), 0.0021 (CV=0.64) and 0.0016 (CV=0.31) for the eastern tropical Pacific Ocean (Wade and Gerrodette 1993; Ferguson and Barlow 2003). Applying the lowest and highest of these density estimates to U.S. EEZ waters surrounding American Samoa (area size = 404,578 km$^2$) yields a range of plausible abundance estimates of 87 – 1,538 false killer whales.

**Minimum Population Estimate**

No minimum population estimate is currently available for waters surrounding American Samoa, but the false killer whale density estimates from other tropical Pacific regions (Barlow and Rankin 2007, Wade and Gerrodette 1993, Ferguson and Barlow 2003, see above) can provide a range of likely values. The lognormal 20[th] percentiles of plausible abundance estimates for the American Samoa EEZ, based on the densities observed elsewhere, range from 45 – 936 false killer whales.

**Current Population Trend**

No data are available on current population trend.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No data are available on current or maximum net productivity rate.

**POTENTIAL BIOLOGICAL REMOVAL**

No PBR can presently be calculated for false killer whales within the American Samoa EEZ, but based on the range of plausible minimum abundance estimates (45 - 936), a recovery factor of 0.40 (for a species of unknown status with a fishery mortality and serious injury rate CV > 0.80 within the American Samoa EEZ; Wade and Angliss 1997), and the default growth rate (½ of 4%), the PBR would likely fall between 0.4 and 7.5 false killer whales per year.

**ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

Information on fishery-related mortality of cetaceans in American Samoan waters is limited, but the gear types used in American Samoas fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used, and float lines from lobster traps and longlines can be expected to occasionally entangle cetaceans (Perrin et al. 1994). The primary fishery in American Samoa is the commercial pelagic longline fishery that targets tunas, which was introduced in 1995 (Levine and Allen 2009). In 2008, there were 28 federally permitted vessels within the longline fishery in American Samoa. The fishery has been monitored since March 2006 under a mandatory observer program, which records all interactions with protected species (Pacific Islands Regional Office 2009). Two false killer whales were killed or seriously injured by the fishery in 2008 (Oleson 2009). The average annual serious injury and mortality in commercial fisheries for false killer whales in American Samoa waters is 7.8 (CV=1.7) animals per year (Table 1).

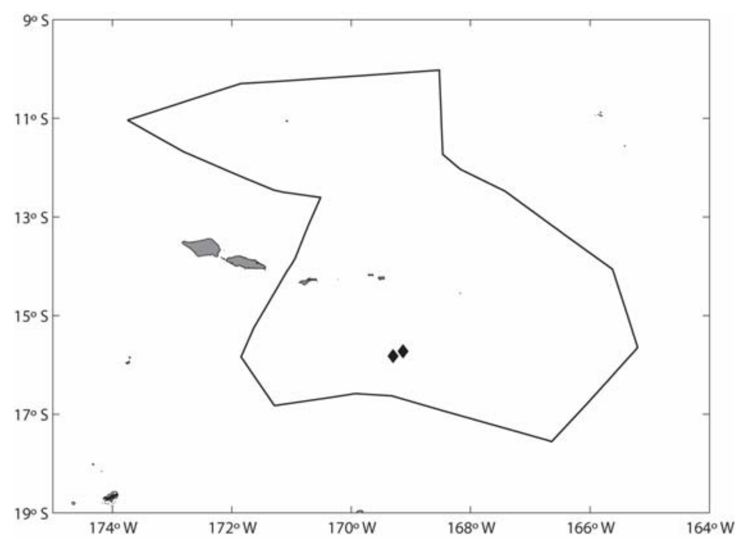


**Figure 2.** Locations of observed false killer whale takes (filled diamonds) in the American Samoa longline fishery, 2006-2008. Solid line represents the U.S. EEZ. Set locations in this fishery are summarized in Appendix 1.

Prior to 1995, bottomfishing and trolling were the primary fisheries in American Samoa but became less prominent after longlining was introduced (Levine and Allen 2009). Nearshore subsistence fisheries include spear fishing, rod and reel, collecting, gill netting, and throw netting (Craig 1993, Levine and Allen 2009). Information on fishery-related mortality of cetaceans in the nearshore fisheries is unknown, but the gear types used in American Samoan fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Gillnets appear to capture marine mammals wherever they are used. Although boat-based nearshore fisheries have been randomly monitored since 1991, by the American Samoa Department of Marine and Wildlife Sources (DMWR), no estimates of annual human-caused mortality and serious injury of cetaceans are available.

## STATUS OF STOCK

The status of false killer whales in American Samoan waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this stock. False killer whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor as "depleted" under the MMPA. The status of the American Samoa stock of false killer whales under the 1994 amendments to the MMPA cannot be determined at this time because no abundance estimates are available and PBR cannot be calculated. However, the estimated rate of fisheries related mortality and serious injury within the American Samoa EEZ (7.8 animals per year) exceeds the range of likely PBRs (0.4 – 7.5) for this region, suggesting that this stock would probably be strategic if abundance estimates were available. Additional research on the abundance of false killer whales in American Samoa is required to resolve this stock's status. Insufficient information is available to determine whether the total fishery mortality and serious injury for false killer whales is insignificant and approaching zero, but this appears unlikely given the estimated takes and likely PBR range.

**Table 1.** Summary of available information on incidental mortality and serious injury of false killer whales (American Samoa stock) in commercial fisheries operating within the U.S. EEZs (Oleson 2009). Longline fishery take estimates represent only those trips with at least 10 sets/trip (Oleson 2009). Mean annual takes are based on 2006-2008 data unless otherwise indicated.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Observed and estimated mortality and serious injury of false killer whales in the American Samoa EEZ | | |
|---|---|---|---|---|---|---|
| | | | | American Samoa EEZ | | |
| | | | | Obs. | Estimated (CV) | Mean Annual Takes (CV) |
| **American Samoa-based longline fishery** | 2006<br>2007<br>2008 | observer data | 9.0%<br>7.7%<br>8.5% | 0<br>0<br>2 | 0 (-)<br>0 (-)<br>23.5 (1.9) | 7.8 (1.7) |
| **Minimum total annual takes within U.S. EEZ waters** | | | | | | 7.8 (1.7) |

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# KILLER WHALE (*Orcinus orca*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

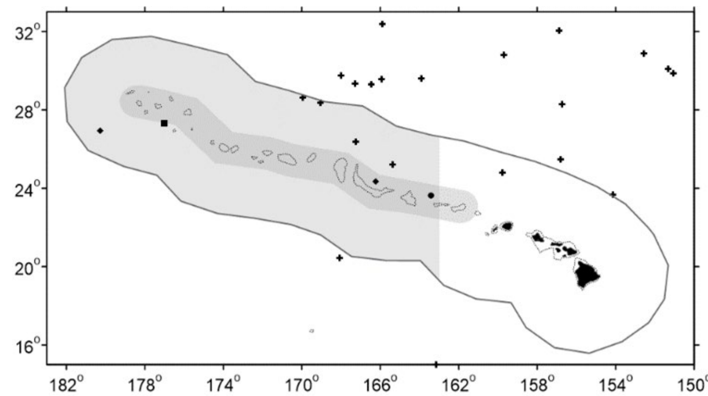


**Figure 1**. Locations of killer whale sightings from longline observer records (crosses; NMFS/PIR, unpublished data) and sighting locations during the 2002 (diamonds), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark and light gray shading indicate the original and the 2016 expanded areas of Papahanaumokuakea Marine National Monument. Dotted line represents the 1000 m isobath.

Killer whales have been observed in all oceans and seas of the world (Leatherwood and Dahlheim 1978). Although reported from tropical and offshore waters (Heyning and Dahlheim 1988), killer whales prefer the colder waters of both hemispheres, with greatest abundances found within 800 km of major continents (Mitchell 1975). They are considered rare in Hawaiian waters. Baird *et al*. (2006) reported 21 sighting records in Hawaiian waters between 1994 and 2004. Summer/fall shipboard surveys of U.S. Exclusive Economic Zone (EEZ) Hawaiian waters resulted in two sightings in 2002, one in 2010, and one in 2017 (Figure 1; Barlow 2006; Bradford *et al.* 2017, Yano *et al.* 2018). Eighteen additional sightings were reported around the main Hawaiian Islands, French Frigate Shoals, and offshore of the Hawaiian Islands (Baird *et al.* 2006). Except in the northeastern Pacific where "resident", "transient", and "offshore" stocks have been described for coastal waters of Alaska, British Columbia, and Washington to California (Bigg 1982; Leatherwood *et al.* 1990, Bigg *et al.* 1990, Ford *et al.* 1994), little is known about stock structure of killer whales in the North Pacific. A global-scale analysis of killer whale phylogeographic structure clustered one animal sampled near Hawaii with eastern and western North Pacific transients. The other Hawaii sample within that analysis did not cluster with any known ecotype, but had divergence time between that of transient and offshore forms (Morin *et al.* 2010). Killer whales in Hawaii have been observed chasing and feeding on both marine mammals and large sharks, including observations of a killer whale attacking a spotted dolphin, chasing a rough-toothed dolphin, and consuming big-eye thresher and hammerhead sharks (Baird 2016).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, eight killer whale stocks are recognized within the Pacific U.S. EEZ: 1) the Eastern North Pacific Alaska Resident stock - occurring from southeastern Alaska to the Aleutian Islands and Bering Sea, 2) the Eastern North Pacific Northern Resident stock - occurring from British Columbia through part of southeastern Alaska, 3) the Eastern North Pacific Southern Resident stock – occurring mainly within the inland waters of Washington State and southern British Columbia, but also in coastal waters from British Columbia through California, 4) the Eastern North Pacific Gulf of Alaska, Aleutian Islands, and Bering Sea Transient stock - occurring mainly from Prince William Sound through the Aleutian Islands and Bering Sea, 5) the AT1 Transient stock - occurring in Alaska from Prince William Sound through the Kenai Fjords, 6) the West Coast Transient stock - occurring from California through southeastern Alaska, 7) the Eastern North Pacific Offshore stock - occurring from California through Alaska, and 8) the Hawaiian stock (this report). The Hawaii stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005). Stock assessment reports for the Southern Resident, Eastern North Pacific Offshore, and Hawaiian stocks can be found in the Pacific Region stock assessment reports; all other killer whale stock assessments are included in the Alaska Region stock assessments.

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of killer whales in the Hawaii EEZ (Bradford *et al*. 2021; Table 1).

Table 1. Line-transect abundance estimates for killer whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 161 | 1.06 | 29-881 |
| 2010 | 145 | 0.98 | 29-726 |
| 2002 | 499 | 0.90 | 111-2,245 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for killer whales from Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al*. (2021), uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available. The best estimate of abundance is based on the 2017 survey, or 161 (CV=1.06) killer whales.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 abundance estimate or 78 killer whales within the Hawaiian Islands EEZ.

### Current Population Trend

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current and maximum net productivity rate in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (78) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 0.8 killer whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other U.S. fisheries. No interactions between nearshore fisheries and killer whales have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch. Killer whale interactions with Hawaii fisheries appear to be rare. In 1990, a solitary killer whale was reported to have removed the catch from a longline in Hawaii (Dollar 1991). There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no killer whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019).

## STATUS OF STOCK

The Hawaii stock of killer whales is not considered strategic under the 1994 amendments to the MMPA. The status of killer whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. No habitat issues are known to be of concern for this stock. Killer whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. Desforges *et al*. (2018) compiled available data on blubber PCB concentrations in killer whales from populations around the world and compared these to established response relationships for reproductive impairment and immunotoxicity-related disease mortality using an individual-based model framework. Model forecasting over 100 years suggested large potential impact of PCBs on the size and long-term viability of some killer whales around the world. The model predicted that killer whales in Hawaiian waters are at high risk of decline due to PCB contaminants, similar to Bigg's killer whale populations sampled in the eastern North Pacific.

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# SHORT-FINNED PILOT WHALE (*Globicephala macrorhynchus*): Hawai‘i Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Short-finned pilot whales are found in all oceans, primarily in tropical and warm-temperate waters. They are commonly sighted during shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, including a high frequency of encounters nearshore within the Northwestern Hawaiian Islands (Figure 1).

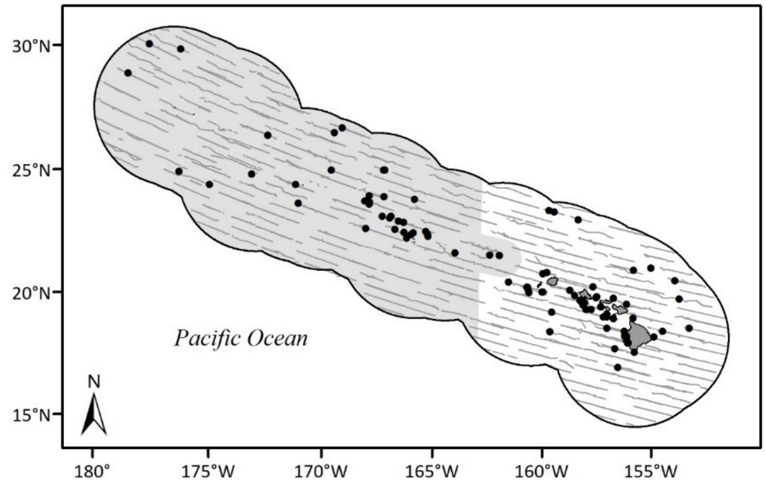


**Figure 1.** Short-finned pilot whale sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al*. 2017), and 2017 (Yano *et al.* 2018) shipboard surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

Two forms of short-finned pilot whales have been identified in Japanese waters based on pigmentation patterns and differences in the shape of the heads of adult males (Kasuya *et al.* 1988). Genetic analysis of samples from throughout the global range of short-finned pilot whales suggest three types within the species, an Atlantic type, a western/central pacific and Indian Ocean (Naisa) type, and an eastern tropical Pacific and northern Japan (Shiho) type. Significant differentiation in mtDNA control region sequences further suggest that the three forms represent two subspecies, the Shiho short-finned pilot whale and the Naisa short-finned pilot whale, with evidence of further divergence among the Naisa types in the Atlantic and Pacific (Van Cise *et al*. 2019). Pilot whales in Hawaiian waters are of the Naisa type. The Shiho and Naisa forms appear also to be distinguishable based on the acoustic features of their whistle and burst-pulse sounds, providing further evidence for divergence between these subspecies (Van Cise *et al*. 2017b).

Photo-identification, telemetry, acoustic, and genetic studies suggest that at least two demographically-independent populations of short-finned pilot whales reside in Hawaiian waters. Resighting and social network analyses of individuals photographed off Hawaii Island suggest the occurrence of one large and several smaller social clusters that use those waters, with some individuals within the smaller social clusters commonly resighted off Hawai‘i Island (Mahaffy *et al.* 2015). Further, two groups of 14 individuals have been seen off Hawai‘i Island and elsewhere in the main Hawaiian Islands, one off O‘ahu and the other off Kaua‘i, indicating some degree of connectivity within the main Hawaiian Islands (MHI). Satellite telemetry data from over 60 individuals tagged throughout the MHI also support the occurrence of at least two populations (Baird 2016, Oleson *et al.* 2013). An assessment of foraging hotspots off Hawai‘i Island revealed tight association between satellite-tagged short-finned pilot whales and the 1000-2500m depth range (Abecassis *et al.* 2015). Further, Van Cise *et al*. (2017a) used nuclear SNPs to assess population structure within short-finned pilot whales around the Hawaiian Archipelago and found evidence for an island-associated population in the MHI. Although there was some support for separation of short-finned pilot whales in the northwestern Hawaiian Islands (NWHI) from other pelagic animals, additional genetic samples may be required to test this separation further. In addition, genetic data combined with social affiliation and habitat associations suggest the MHI population is further divided into social groups, and these groups may even rise to the level of demographic-independence between those found primarily near Hawai‘i Island and those near O‘ahu and Kaua‘i (Van Cise *et al.* 2017a). Differences in the acoustic features of short-finned pilot whale social clusters recorded within the MHI further supports the existence of demographically-independent populations within the MHI (Van Cise *et al*. 2017b). Formal assessment of demographic-independence has not been completed, but division of this population into one or more island-associated stocks may be warranted in the future.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, short-finned pilot whales within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawai'i stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ around the Hawaiian Islands (NMFS 2005).

**POPULATION SIZE**

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated, resulting in updated model-based abundance estimates of short-finned pilot whales in the entirety of the Hawaiian Islands EEZ (Becker *et al*. 2021, 2022; Table 1).

**Table 1.** Model-based line-transect abundance estimates for short-finned pilot whales in the Hawaiian Islands EEZ in 2002 and 2010 (Becker *et al.* 2021) and 2017 and 2020 (Becker *et al.* 2022), derived from NMFS surveys in the central Pacific since 2000. The Becker *et al.* (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

| Year | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2020 | 19,242 | 0.23 | 12,289-30,129 |
| 2017 | 17,237 | 0.23 | 11,009-26,989 |
| 2010 | 15,343 | 0.17 | 11,039-21,326 |
| 2002 | 15,198 | 0.17 | 10,900-21,191 |

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker *et al.* 2021) and 2017-2020 (Becker *et al.* 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al.* (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al.* 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates. The modeling framework incorporated Beaufort-specific trackline detection probabilities for short-finned pilot whales from Barlow *et al.* (2015). Models were used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al*. 2015, Becker *et al*. 2016). Bradford *et al.* (2021) produced design-based abundance estimates for short-finned pilot whales in 2002, 2010, and 2017 that can be used as a point of comparison to the model-based estimates for those years. While on average the estimates are similar between the two approaches, the annual design-based estimates show greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Model based-estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total

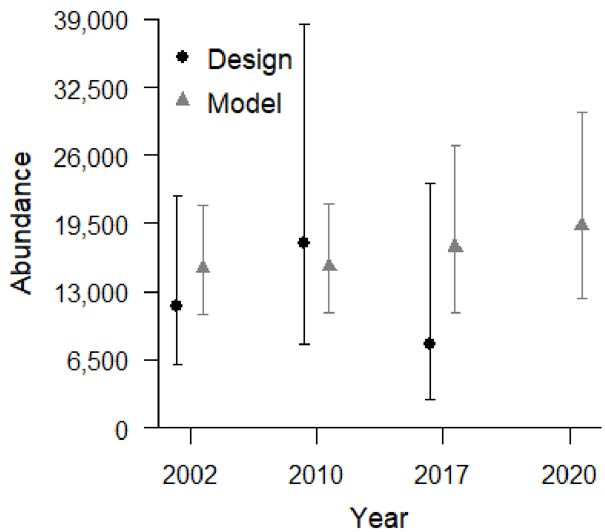


**Figure 2.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al*. 2021, 2022) estimates of abundance for short-finned pilot whales for each survey year (2002, 2010, 2017, 2020).

abundance, the model-based estimates are considered the best available estimate for each survey year. Becker *et al.* (2022) and Bradford *et al.* (2022) evaluated seasonal changes in the abundance of short-finned pilot whales within the main Hawaiian Islands using summer-fall data from 2017 and winter survey data from 2020. Although the model identified moderately lower densities of short-finned pilot whales in the MHI in winter, the design-based analysis showed a 7-fold increase in density during the same period, though confidence limits partly overlap for both analyses. The disparate results may demonstrate the impacts of encounter rate variation on the annual design-based estimates, though also suggest additional data will be needed to understand habitat relationships and seasonal movements of this species in Hawaiian waters. Previously published abundance estimates for the Hawaiian Islands EEZ (Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al*. 2017) used a subset of the dataset used by Becker *et al*. (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 19,242 (CV=0.23) short-finned pilot whales.

### Minimum Population Estimate

The minimum population estimate is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate (from Becker *et al.* 2022) or 15,894 short-finned pilot whales.

### Current Population Trend

The model-based abundance estimates for short-finned pilot whales provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Model-based examination of short-finned pilot whale population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaiʻi short-finned pilot whale stock is calculated as the minimum population estimate (15,894) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.5 (for a species of unknown status with no known fishery mortality within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997), resulting in a PBR of 159 short-finned pilot whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. Entanglement in gillnets and hooking or entanglement in various hook and line fisheries have been reported for small cetaceans in Hawaiʻi (Nitta & Henderson, 1993). Short-finned pilot whales have been observed with fishing gear trailing from their mouths or have stranded with gear and other debris in their stomach, though the specific gear types have not been identified (Baird 2016, Bradford and Lyman 2018, 2019). In 2017, two short-finned pilot whales stranded together as part of a mass stranding event on Kauai. One of the whales had 12-15 lbs of nylon line and plastic present within its forestomach and the other had scarring on the upper right jaw consistent with previous fisheries interaction, though in neither case were these findings considered to be related to the cause of death (Bradford and Lyman 2019). In 2020, a short-finned pilot whale was observed off Hawaiʻi Island with trailing line from its mouth, suggesting the whale was hooked in the mouth or had ingested the hook (Bradford and Lyman 2023), an injury that is considered serious according to criteria for assessing serious injury in marine mammals (NMFS 2023). No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line or gillnet fisheries because these fisheries are not observed or monitored for protected species bycatch.

**Table 2.** Summary of available information on incidental mortality and serious injury (MSI) of short-finned pilot whales (GM) and including those presumed to be short-finned pilot whales based on assignment of unidentified blackfish (UB) to this species in commercial longline fisheries, within and outside of the U.S. EEZ (McCracken & Cooper 2022b). Mean annual takes are based on 2017-2021 data unless otherwise indicated. Information on all observed takes (T) and MSI is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome. UB are prorated as either false killer whales or short-finned pilot whales according to their distance from shore (McCracken 2010). CVs are estimated based on the combination of annual short-finned pilot whale and blackfish variances and do not yet incorporate additional uncertainty introduced by prorating the unidentified blackfish.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Outside U.S. EEZ | | Hawaiian Islands EEZ | |
|---|---|---|---|---|---|---|---|
| | | | | **Observed GM T/MSI**<br>**Observed UB T/MSI** | **Estimated MSI (CV)** | **Observed GM T/MSI**<br>**Observed UB T/MSI** | **Estimated MSI (CV)** |
| **Hawai‘i-based deep-set longline fishery** | 2017 | Observer data | 20% | 0<br>0 | 0 (-) | 0<br>0 | 0 (-) |
| | 2018 | | 18% | 0<br>1/1 | 0.9 (0.8) | 0<br>0 | 0 (-) |
| | 2019 | | 21% | 0<br>1/0 | 0.4 (1.1) | 0 | 0 (-) |
| | 2020 | | 15% | 0<br>0 | 0 | 0 | 0 (-) |
| | 2021 | | 18% | 0<br>1/1 | 5.4 (1.0) | 0<br>0 | 0 (-) |
| **Mean Estimated Annual Take (CV) 2017-2021** | | | | | **1.3 (1.6)** | | **0** |
| **Hawai‘i-based shallow-set longline fishery** | 2017 | | 100% | 0<br>0 | 0 | 0<br>0 | 0 |
| | 2018 | | 100% | 0<br>0 | 0 | 0<br>0 | 0 |
| | 2019 | | 100% | 0<br>0 | 0 | 0 | 0 |
| | 2020 | | 100% | 0<br>0 | 0 | 0 | 0 |
| | 2021 | | 100% | 0<br>0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage) 2017-2021** | | | | 0 | 0 | 0 | 0 |
| **Minimum total annual takes within U.S. EEZ (2017-2021)** | | | | | | | **0** |

There are currently two distinct longline fisheries based in Hawai‘i: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the MHI and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the NWHI. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017-2021, no short-finned pilot whales were observed hooked or entangled in the SSLL fishery (100% observer coverage), and one was observed taken in the DSLL fishery (15-21% observer coverage) (Figure 3, McCracken and Cooper, 2022b), outside the Hawaiian Islands EEZ. Based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2023), this short-finned pilot whale was considered seriously injured. Two additional unidentified "blackfish" (unidentified cetaceans known to be either false killer whales or short-finned pilot whales) were taken during 2017-2021, McCraken and Cooper, 2022b), both within the DSLL fishery. Both of the blackfish interactions occurred outside the Hawaiian Islands EEZ, with one considered seriously injured and one considered non-seriously injured. Takes of unidentified blackfish are prorated to false killer whale and short-finned pilot whale based on distance from shore (McCracken 2010), given patterns of previous bycatch for each species. Proration of unidentified blackfish takes introduces unquantified uncertainty into the bycatch estimates, but until all animals taken can be identified to species (e.g., photos, tissue samples), this approach ensures that potential impacts to all stocks are assessed.

The total estimated number of dead or seriously injured dolphins is calculated based on observer coverage rate, the location of the observed take (inside or outside of the EEZ), and the ratio of observed dead and seriously injured whales versus those judged to be not seriously injured. Observer coverage is measured on a per-trip basis throughout the calendar year as described by McCracken (2019). In years with large fluctuations in observer coverage, such as during the early days of the COVID-19 pandemic when observer coverage dropped to less than 10% during the second quarter of the year, the annual bycatch estimation process may be subset into several periods, as described in McCracken & Cooper (2022a). Average 5-yr estimates of annual mortality and serious injury for 2017-2021 are 1.3 (CV=1.6) short-finned pilot whales outside of the U.S. EEZ, and 0 within the Hawaiian Islands EEZ (Table 2, McCracken and Cooper 2002b). Two additional unidentified cetaceans, likely to be blackfish based on the observer's description, were taken in the DSLL fishery and may have been short-finned pilot whales.

**STATUS OF STOCK**

The Hawai‘i stock of short-finned pilot whales is not considered strategic under the 1994 amendments to the MMPA. The status of short-finned pilot whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Short-finned pilot whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. In the past 5 years, one short-finned pilot whale was observed in nearshore waters seriously injured by fishing gear, although the source of the gear is unknown (Bradford and Lyman 2023). There is no systematic monitoring for interactions with protected species within near-shore fisheries that may take this species, thus total mean annual takes (0.2 yr) are undetermined. Given the absence of recent recorded longline fishery-related mortality or serious injuries and low levels of nearshore fisheries interactions within the U.S. EEZ, the total fishery mortality and serious injury for short-finned pilot whales can be considered to be insignificant and approaching zero. Two short-finned pilot whales were found stranded in separate incidents following Navy sonar training exercises in Hawai‘i in 2014 (Bradford and Lyman 2018). Examination of the whales could not conclusively link these stranding to use of sonar, though other blackfish have shown sensitivity to sonar training events in Hawaiian waters (Southall *et al*. 2006) and elsewhere (Brownell *et al*. 2009). Two of five short-finned pilot whales that died in a mass stranding on Kauai in 2017 had tissues infected with beaked whale circovirus (Clifton *et al.* 2023), which can lead to serious illness and immunosuppression, though it is not clear what effect that infection had in these strandings.

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# BLAINVILLE'S BEAKED WHALE (*Mesoplodon densirostris*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

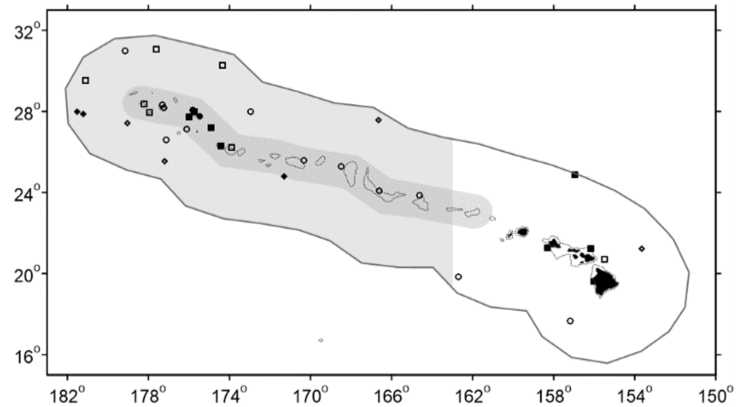


Figure 1. Sighting locations of *Mesoplodon densirostris* during the 2002 (diamond), 2010 (circle), and 2017 (square) and unidentified *Mesoplodon* beaked whales during the 2002 (open diamond), 2010 (open circle), and 2017 (open square) shipboard cetacean surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000m isobath.

Blainville's beaked whale has a cosmopolitan distribution in tropical and temperate waters, apparently the most extensive known distribution of any *Mesoplodon* species (Mead 1989). Forty-five sightings over 13 years were reported from the main islands by Baird *et al.* (2013), who indicated that Blainville's beaked whale represent a small proportion (2-3%) of all odontocete sightings in the main Hawaiian Islands. Shallenberger (1981) suggested that Blainville's beaked whales were present off the Waianae Coast of Oahu for prolonged periods annually. Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in three sightings in 2002, one in 2010, and eight in 2017; however, several sightings of unidentified *Mesoplodon* whales may have also been Blainville's beaked whale (Figure 1; Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018).

Recent analysis of Blainville's beaked whale resightings and movements near the main Hawaiian Islands (MHI) suggest the existence of insular and offshore (pelagic) populations of this species in Hawaiian waters (McSweeney *et al.* 2007, Schorr *et al.* 2009, Baird *et al.* 2013, Baird 2019). Photo-identification of individual Blainville's beaked whales from Hawaii Island since 1986 reveal repeated use of this area by individuals for over 17 years (Baird *et al.* 2011) and 75% of individuals seen off Hawaii Island link by association into a single social network (Baird *et al.* 2013). Those individuals seen farthest from shore and in deep water (>2100m) have not been resighted, suggesting they may be part of an offshore, pelagic population (Baird *et al.* 2011). Twelve Blainville's beaked whales linked to the social network have been satellite tagged off Hawaii Island. All 12 individuals had movements restricted to the MHI, extending to nearshore waters of Oahu, with average distance from shore of 21.6 km (Baird *et al.* 2013, Abecassis *et al.* 2015). One individual tagged 32km from Hawaii Island did not link to the social network and had movements extending far from shore, moving over 900km from the tagging location in 20 days, approaching the edge of the Hawaiian EEZ west of Nihoa (Baird *et al.* 2011). An assessment of foraging hotspots off Hawaii Island revealed tight association between satellite-tagged Blainville's beaked whales and the 250-2500m depth contour and the occurrence of the island-associated deep mesopelagic boundary community (Abecassis *et al.* 2015). The available movement, social structure, and habitat data suggest there is likely a separate island-associated population of Blainville's beaked whales within the MHI (Baird 2019). Formal assessment of demographic-independence has not been completed, but division of this population into a separate island-associated stock may be warranted in the future.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, three *Mesoplodon* stocks are defined within the Pacific U.S. EEZ: 1) *M. densirostris* in Hawaiian waters (this report), 2) *M. stejnegeri* in Alaskan waters, and 3) all *Mesoplodon* species off California, Oregon and Washington. The Hawaii stock of Blainville's beaked whales includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of Blainville's beaked whales in the Hawaii EEZ (Bradford *et al*. 2021; Table 1).

Table 1. Line-transect abundance estimates for Blainville's beaked whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 1,132 | 0.99 | 224-5,731 |
| 2010 | 1,740 | 1.05 | 320-9,468 |
| 2002 | 839 | 1.05 | 155-4,536 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for Blainville's beaked whales from Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al*. (2021*),* uses a consistent approach for estimating all abundance parameters and the resulting estimates are considered the best available for each survey year.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2010 abundance estimate or 564 Blainville's beaked whales within the Hawaiian Islands EEZ.

### Current Population Trend

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (564) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a species of unknown status with no recent fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 5.6 Hawaii Blainville's beaked whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaii fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No interactions between nearshore fisheries and Blainville's beaked whales have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no Blainville's beaked whale was observed killed or seriously injured in the SSLL fishery (100% observer coverage) or the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019) within the Hawaiian EEZ. One unidentified beaked whale was observed taken, but not seriously injured, within the Hawaiian EEZ in the DSLL fishery (Bradford 2018a). Average 5-yr estimates of annual mortality and serious injury for 2014-2018 are zero Blainville's beaked whales within or outside of the U.S. EEZs, and 0.5 (CV = 1.2) unidentified beaked whales within the U.S. EEZs (Table 1).

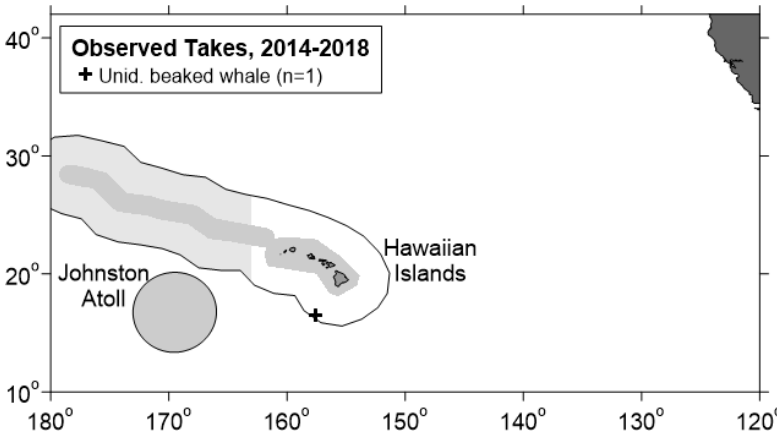


**Figure 2.** Location of an unidentified beaked whale take (cross) in Hawaii-based longline fisheries, 2014-2018. Solid lines represent the U.S. EEZ. Gray shading notes areas closed to commercial fishing, with the PMNM Expansion area closed since August 2016.

**Table 1.** Summary of available information on incidental mortality and serious injury of Blainville's beaked whales (Hawaii stock) in commercial longline fisheries, within and outside of the Hawaiian Islands EEZ (McCracken 2019). Mean annual takes are based on 2014-2018 data unless otherwise indicated. Information on all observed takes (T) and combined mortality events & serious injuries (MSI) is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| | | | | **Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of Blainville's beaked whales (MD), unidentified Mesoplont whales (UM) and unidentified beaked whales (ZU)** | | | |
|---|---|---|---|---|---|---|---|
| | | | | **Outside U.S. EEZs** | | **Hawaiian EEZ** | |
| **Fishery Name** | **Year** | **Data Type** | **Percent Observer Coverage** | **Obs. MD T/MSI Obs. UM+ZU T/MSI** | **Estimated MD M&SI (CV) Estimated UM+ZU MSI (CV)** | **Obs. MD T/MSI Obs. UM+ZU T/MSI** | **Estimated MD M&SI (CV) Estimated UM+ZU MSI (CV)** |
| **Hawaii-based deep-set longline fishery** | 2014 | Observer Data | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2015 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2016 | | 20% | 0 | 0 | 0<br>1/0 | 0<br>3 (0.9) |
| | 2017 | | 20% | 0 | 0 | 0 | 0 |
| | 2018 | | 18% | 0 | 0 | 0 | 0 |
| **Mean Estimated Annual MD Take (CV)** | | | | | **0 (-)** | | **0 (-)** |
| **Mean Estimated Annual UM+ZU Take (CV)** | | | | | **0 (-)** | | **0.5 (1.2)** |
| **Hawaii-based shallow-set longline fishery** | 2014 | Observer Data | 100% | 0 | 0 | 0 | 0 |
| | 2015 | | 100% | 0 | 0 | 0 | 0 |
| | 2016 | | 100% | 0 | 0 | 0 | 0 |
| | 2017 | | 100% | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 0 | 0 | 0 | 0 |
| **Mean Annual MD Takes (100% coverage)** | | | | | **0** | | **0** |
| **Mean Annual UM + ZU Takes (100% coverage)** | | | | | **0.6** | | **0** |
| **Minimum total annual MD takes within U.S. EEZ** | | | | | | | **0 (-)** |

**Other Mortality**

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to the remote nature of such activities and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. Similarly, Simonis *et al.* (2020) reported a statistically significant correlation between sonar use and single and mass stranding events of beaked whales in the Mariana Archipelago In Hawaiian waters, Faerber & Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt the foraging dives and echolocation activities of tagged beaked whales (Tyack *et al.* 2011, DeRuiter *et al.* 2013). Cuvier's beaked whales tagged and tracked during simulated mid-frequency sonar exposure showed avoidance reactions, including prolonged diving, cessation of echolocation click production associated with foraging, and directional travel away from the simulated sonar source (DeRuiter *et al.* 2013). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al.* 2011). Fernández *et al.* (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al.* 2008, Carretta and Barlow 2011). The impact of sonar exercises on resident versus offshore beaked whales may be significantly different with offshore animals less frequently exposed, and possibly subject to more extreme reactions (Baird *et al.* 2009). No estimates of potential mortality or serious injury are available for U.S. waters.

## STATUS OF STOCK

The Hawaii stock of Blainville's beaked whales is not considered strategic under the 1994 amendments to the MMPA The status of Blainville's beaked whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Blainville's beaked whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recorded recent fishery-related mortality or serious injuries within U.S. EEZs, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. The impacts of anthropogenic sound on beaked whales remain a concern (Barlow and Gisiner 2006, Cox *et al.* 2006, Hildebrand *et al.* 2005, Weilgart 2007). One Blainville's beaked whale found stranded on the main Hawaiian Islands has tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in the 3 known species of beaked whales in Hawaiian waters, raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including the cumulative impacts of disease with other stressors.

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# LONGMAN'S BEAKED WHALE (*Indopacetus pacificus*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

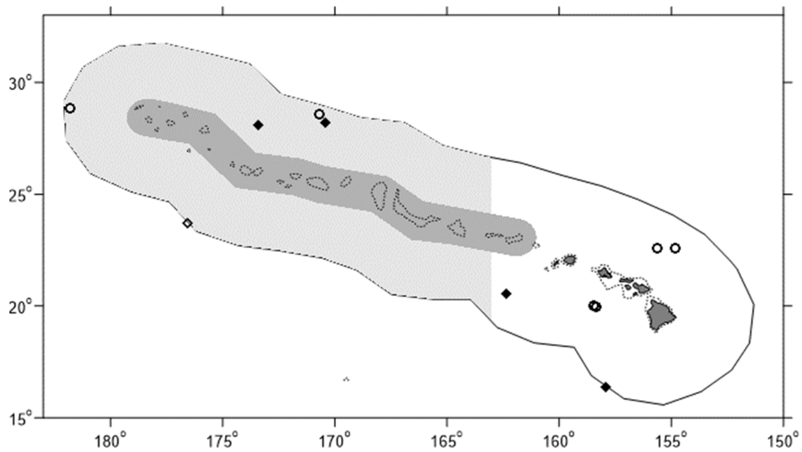


**Figure 1.** Sighting locations of Longman's beaked whale during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard cetacean surveys of U.S. waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Longman's beaked whale is considered one of the least known cetacean species (Jefferson *et al.* 1993; Rice 1998; Dalebout *et al.* 2003), originally known only from two skulls found in Australia and Somalia (Longman 1926; Azzaroli 1968). Genetic studies (Dalebout *et al.* 2003) have revealed that sightings of 'tropical bottlenose whales' (*Hyperoodon* sp.; Pitman *et al.* 1999) in the Indo-Pacific region were in fact Longman's beaked whales, providing the first description of the external appearance of this species. Although originally described as *Mesoplodon pacificus* (Longman 1926), it has been proposed that this species is sufficiently unique to be placed within its own genus, *Indopacetus* (Moore 1968; Dalebout *et al.* 2003). The distribution of Longman's beaked whale, as determined from stranded specimens and sighting records of 'tropical bottlenose whales', includes tropical waters from the eastern Pacific westward through the Indian Ocean to the eastern coast of Africa. A single stranding of Longman's beaked whale has been reported in Hawaii, in 2010 near Hana, Maui (West *et al.* 2012), and there was a single sighting off Kona over 13 years of nearshore surveys off in the leeward waters of the main Hawaiian Islands (Baird 2016). Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in one sighting in 2002, three in 2010, and seven in 2017 (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018; Figure 1).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there is one Pacific stock of Longman's beaked whales, found within waters of the Hawaiian Islands EEZ. This stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of Longman's beaked whales in the Hawaii EEZ (Bradford *et al.* 2021; Table 1).

Table 1. Line-transect abundance estimates for Longman's beaked whale derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 2,550 | 0.67 | 771-8,432 |
| 2010 | 7,003 | 0.63 | 2,260-21,697 |
| 2002 | 871 | 1.06 | 158-4,798 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for Longman's beaked whales from Barlow *et al.* (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021), uses a consistent approach for estimating all abundance parameters and are considered the best available estimates for each survey year. The best estimate of abundance is based on the 2017 survey, or 2,550 (CV=0.67) whales.

**Minimum Population Estimate**

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) around the 2017 abundance estimate, or 1,527 Longman's beaked whales within the Hawaiian Islands EEZ.

**Current Population Trend**

The three available abundance estimates for this stock have very broad and overlapping confidence intervals, precluding robust evaluation of population trend for this stock.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for Longman's beaked whales.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (1,527) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 15 Longman's beaked whales per year.

## HUMAN CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other U.S. fisheries. No interactions between nearshore fisheries and Longman's beaked whales have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch. There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate in U.S. waters and on the high seas. Between 2014 and 2018, no Longman's beaked whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (20-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019).

**Other Mortality**

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to their remote nature and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. Similarly, Simonis *et al.* (2020) reported a statistically significant correlation between sonar use and single and mass stranding events of beaked whales in the Mariana Archipelago. In Hawaiian waters, Faerber and Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt foraging dives and echolocation activities of tagged beaked whales (Tyack *et al*. 2011, DeRuiter *et al*. 2013). Cuvier's beaked whales tagged and tracked during simulated mid-frequency sonar exposure showed avoidance reactions, including prolonged diving, cessation of foraging echolocation click production, and directional travel away from the simulated sonar source (DeRuiter *et al*. 2013). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al*. 2011). Fernández *et al*. (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al*. 2008, Carretta and Barlow 2011). No estimates of potential mortality or serious injury are available for U.S. waters.

## STATUS OF STOCK

The Hawaii stock of Longman's beaked whales is not considered strategic under the 1994 amendments to the MMPA. The status of Longman's beaked whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Longmans' beaked whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. The impacts of anthropogenic sound on beaked whales remain a concern (Barlow and Gisiner 2006, Cox *et al*. 2006, Hildebrand *et al*. 2005, Weilgart 2007). The first confirmed case of *morbillivirus* in a Hawaiian cetacean was found in a subadult Longman's beaked whale stranded on Maui in 2010 (West *et al*. 2012). The presence of *morbillivirus* in all 3 known species of beaked whales in Hawaiian waters (Jacob *et al*. 2016), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including cumulative impacts of disease with other stressors.

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# CUVIER'S BEAKED WHALE (*Ziphius cavirostris*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Cuvier's beaked whales occur in all oceans and major seas (Heyning 1989). Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands, resulted in 4 sightings in 2002, 22 in 2010, and 11 in 2017 including markedly higher sighting rates during nearshore surveys in the Northwestern Hawaiian Islands. (Figure 1; Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018).

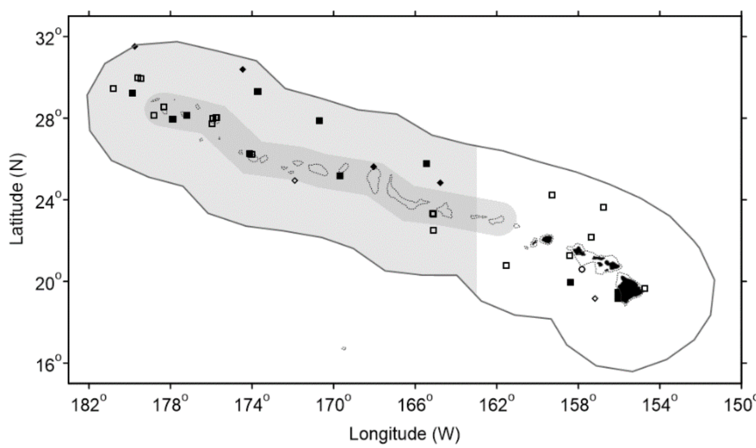


**Figure 1**. Cuvier's beaked whale sighting locations during the 2002 (diamonds), 2010 (circle), and 2017 (square) shipboard surveys, as well as sightings of unidentified *Ziphiid* during 2002 (open diamond), 2010 (open circle), and 2017 (open square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates of the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Resighting and movement data of individual Cuvier's beaked whales suggest the existence of insular and offshore populations of this species in Hawaiian waters (Baird 2019). A 21-yr study off Hawaii Island suggests long-term site fidelity and year round occurrence (McSweeney *et al.* 2007). Ten Cuvier's beaked whales have been tagged off Hawaii Island since 2006, with all remaining close to the island of Hawaii or to Maui for the duration of tag data received (Baird *et al.* 2013, Baird 2019). Approximately 95% of all locations were within 45 km of shore and the farthest offshore an individual was documented was 67 km (Baird *et al.* 2013). The available satellite data suggest that a resident population may occur near Hawaii Island, distinct from offshore, pelagic Cuvier's beaked whales. This conclusion is further supported by the long-term site fidelity evident from photo-identification data (McSweeney *et al.* 2007). The available movement, social structure, and habitat data suggest there is likely a separate island-associated population of Cuvier's beaked whales within the main Hawaiian Islands (Baird 2019). Formal assessment of demographic-independence has not been completed, but division of this population into a separate island-associated stock may be warranted in the future.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, Cuvier's beaked whales within the Pacific U.S. EEZ are divided into three discrete, non-contiguous areas: 1) Hawaiian waters (this report), 2) Alaskan waters, and 3) waters off California, Oregon and Washington. The Hawaii stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters. Because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in updated abundance estimates of Cuvier's beaked whale in the Hawaii EEZ (Bradford *et al.*2021; Table 1).

Table 1. Line-transect abundance estimates for Cuvier's beaked whale derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al.* 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 4,431 | 0.41 | 2,036-9,644 |
| 2010 | 338 | 1.02 | 65-1,771 |
| 2002 | 1,216 | 0.77 | 319-4,633 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for Cuvier's beaked whale from Barlow *et al.* (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021), uses a consistent approach for estimating all abundance parameters and the resulting estimates are considered the best available for each survey year. Wade and Gerrodette (1993) estimated population size for Cuvier's beaked whales in the eastern tropical Pacific, but it is not known whether any of these animals are part of the same population that occurs around the Hawaiian Islands.

**Minimum Population Estimate**

Minimum population size is calculated as the lower 20$^{th}$ percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2010 abundance estimate, or 3,180 Cuvier's beaked whales.

**Current Population Trend**

The three available abundance estimates for this stock have very broad confidence intervals, and for 2010 and 2017, they do not overlap. Annual encounter rate variation may have a large impact on abundance estimates for species with low density and patchy distribution. Bradford *et al.* (2021) indicate that the high sighting rate, and correspondingly higher abundance estimate, may be the result of extreme encounter rate variability for this species, though animal movement in response to environmental conditions or the influence of sightings of island-associated groups during systematic survey in 2017 cannot be discounted.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No data are available on current or maximum net productivity rate.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for the pelagic stock of Cuvier's beaked whales is calculated as the minimum population estimate for the U.S. EEZ of the Hawaiian Islands (3,180) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no known fishery mortality within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 32 Cuvier's beaked whales per year.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Fishery Information**

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no Cuvier's beaked whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019). One unidentified beaked whale was taken, but not seriously injured, within the Hawaiian EEZ in the DSLL fishery (Bradford 2018a). Average 5-yr estimates of annual mortality and serious injury for 2014-2018 are zero Cuvier's beaked whales within or outside of the U.S. EEZs, and 0.5 (CV = 1.2) unidentified beaked whales within the U.S. EEZs (Table 2).

**Table 2.** Summary of available information on incidental mortality and serious injury of Cuvier's beaked whales (Hawaii stock) in commercial longline fisheries, within and outside of the Hawaiian Islands EEZ (McCracken 2019). Mean annual takes are based on 2014-2018 data unless otherwise indicated. Information on all observed takes (T) and

combined mortality events & serious injuries (MSI) is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| | | | | **Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of Cuvier's beaked whales (ZI),and unidentified beaked whales (ZU)** | | | |
|---|---|---|---|---|---|---|---|
| | | | | **Outside U.S. EEZs** | | **Hawaiian EEZ** | |
| **Fishery Name** | **Year** | **Data Type** | **Percent Observer Coverage** | **Obs. ZI T/MSI<br>Obs. ZU T/MSI** | **Estimated ZI M&SI (CV)<br>Estimated ZU MSI (CV)** | **Obs. ZI T/MSI<br>Obs. ZU T/MSI** | **Estimated ZI M&SI (CV)<br>Estimated ZU MSI (CV)** |
| **Hawaii-based deep-set longline fishery** | 2014 | Observer data | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2015 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2016 | | 20% | 0 | 0 | 0<br>1/0 | 0<br>3 (0.9) |
| | 2017 | | 20% | 0 | 0 | 0 | 0 |
| | 2018 | | 18% | 0 | 0 | 0 | 0 |
| **Mean Estimated Annual ZI Take (CV)** | | | | | **0 (-)** | | **0 (-)** |
| **Mean Estimated Annual ZU Take (CV)** | | | | | **0 (-)** | | **0.5 (1.2)** |
| **Hawaii-based shallow-set longline fishery** | 2014 | Observer data | 100% | 0 | 0 | 0 | 0 |
| | 2015 | | 100% | 0 | 0 | 0 | 0 |
| | 2016 | | 100% | 0 | 0 | 0 | 0 |
| | 2017 | | 100% | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 0 | 0 | 0 | 0 |
| **Mean Annual ZI Takes (100% coverage)** | | | | | **0** | | **0** |
| **Mean Annual ZU Takes (100% coverage)** | | | | | **0.6** | | **0** |
| **Minimum total annual ZI takes within U.S. EEZ** | | | | | | | **0 (-)** |

## Other Mortality

Anthropogenic sound sources, such as military sonar and seismic testing have been implicated in the mass strandings of beaked whales, including atypical events involving multiple beaked whale species (Simmonds and Lopez-Jurado 1991, Frantiz 1998, Anon. 2001, Jepson *et al.* 2003, Cox *et al.* 2006). While D'Amico *et al.* (2009) note that most mass strandings of beaked whales are unassociated with documented sonar activities, lethal or sub-lethal effects of such activities would rarely be documented, due to the remote nature of such activities and the low probability that an injured or dead beaked whale would strand. Filadelpho *et al.* (2009) reported statistically significant correlations between military sonar use and mass strandings of beaked whales in the Mediterranean and Caribbean Seas, but not in Japanese and Southern California waters, and hypothesized that regions with steep bathymetry adjacent to coastlines are more conducive to stranding events in the presence of sonar use. Similarly, Simonis *et al.* (2020) reported a statistically significant correlation between sonar use and single and mass stranding events of beaked whales in the Mariana Archipelago. In Hawaiian waters, Faerber and Baird (2010) suggest that the probability of stranding is lower than in some other regions due to nearshore currents carrying animals away from beaches, and that stranded animals are less likely to be detected due to low human population density near many of Hawaii's beaches. Actual and simulated sonar are known to interrupt the foraging dives and echolocation activities of tagged beaked whales (Tyack *et al.* 2011, DeRuiter *et al.* 2013). Cuvier's beaked whales tagged and tracked during simulated mid-frequency sonar exposure showed avoidance reactions, including prolonged diving, cessation of echolocation click production associated with foraging, and directional travel away from the simulated sonar source (DeRuiter *et al.* 2013). Blainville's beaked whale presence was monitored on hydrophone arrays before, during, and after sonar activities on a Caribbean military range, with evidence of avoidance behavior: whales were detected throughout the range prior to sonar exposure, not detected in the center of the range coincident with highest sonar use, and gradually returned to the range center after the cessation of sonar activity (Tyack *et al.* 2011). Fernández *et al.* (2013) report that there have been no mass strandings of beaked whales in the Canary Islands following a 2004 ban on sonar activities in that region. The absence of beaked whale bycatch in California drift gillnets following the introduction of acoustic pingers into the fishery implies additional sensitivity of beaked whales to anthropogenic sound (Carretta *et al.* 2008, Carretta and Barlow 2011). The impact of sonar exercises on resident versus offshore beaked whales may be significantly different with offshore animals less frequently exposed, and possibly subject to more extreme reactions

(Baird *et al.* 2009). No estimates of potential mortality or serious injury are available for U.S. waters.

## STATUS OF STOCK

The Hawaii stock of Cuvier's beaked whales is not considered strategic under the 1994 amendments to the MMPA. The status of Cuvier's beaked whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Cuvier's beaked whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reported fishery related mortality or injuries within the Hawaiian Islands EEZ, such that the total mortality and serious injury can be considered to be insignificant and approaching zero. The impacts of anthropogenic sound on beaked whales remain a concern (Barlow and Gisiner 2006, Cox *et al.* 2006, Hildebrand *et al.* 2005, Weilgart 2007). One Cuvier's beaked whale found stranded on the main Hawaiian Islands tested positive for *Morbillivirus* (Jacob *et al.* 2016). The presence of *morbillivirus* in all 3 known species of beaked whales in Hawaiian waters (Jacob *et al*. 2016), raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts, including the cumulative impacts of disease with other stressors.

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Revised 5/5/2021

# PYGMY SPERM WHALE (*Kogia breviceps*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Pygmy sperm whales are found throughout the world in tropical and warm-temperate waters (Caldwell and Caldwell 1989). Pygmy sperm whales have been observed in nearshore waters off Oahu, Maui, Niihau, and Hawaii Island (Shallenberger 1981, Mobley *et al*. 2000, Baird 2005, Baird *et al*. 2013). Two sightings were made during a 2002, and three during 2017 shipboard survey of waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands (Figure 1; Barlow 2006, Yano *et al*. 2018). A freshly dead pygmy sperm whale was picked up approximately 100 nmi north of French Frigate Shoals on a similar 2010 survey (NMFS, unpublished data). Nothing is known about stock structure for this species.

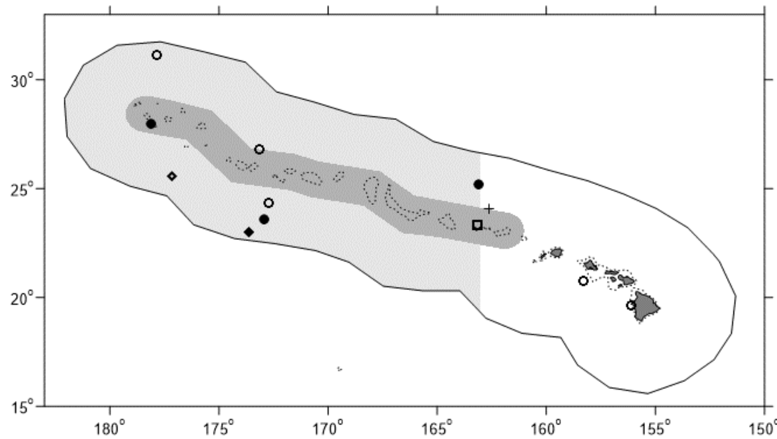


**Figure 1**. Sighting locations for pygmy sperm whales during 2002 (filled diamond) and 2017 (square) shipboard surveys, as well as sightings of and unidentified *Kogia* during 2002 (open diamond) 2010 (open circle), and 2017 (open square) shipboard cetacean surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2013, Yano *et al*. 2018). A freshly dead pygmy sperm whale was also retrieved during the 2010 survey (cross). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, pygmy sperm whales within the Pacific U.S. EEZ are divided into two discrete areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawaii stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of pygmy sperm whales in the Hawaii EEZ (Bradford *et al*. 2021; Table 1). A 2010 shipboard line-transect survey within the Hawaiian EEZ did not result in any sightings of pygmy sperm whales (Bradford *et al*. 2017).

Table 1. Line-transect abundance estimates for pygmy sperm whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al*. 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 42,083 | 0.64 | 13,406-132,103 |
| 2010 | N/A | | |
| 2002 | 12,036 | 1.04 | 2,248-64,434 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for pygmy sperm whales from Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, estimates within Bradford *et al*. (2021) use a consistent approach for estimating all abundance parameters and are considered the best available estimates for each survey year. The best estimate of abundance is from the 2017 survey, or 42,083 (CV=0.64) whales.

**Minimum Population Estimate**

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) around the 2017 abundance estimate, or 25,695 pygmy sperm whales within the Hawaiian Islands EEZ.

**Current Population Trend**

No data are available on current population abundance or trend.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No data are available on current or maximum net productivity rate.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (25,695) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 257 pygmy sperm whales per year.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Fishery Information**

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, one pygmy sperm whale was observed hooked in the DSLL fishery (20-22% observer coverage) (Figure 2, Bradford 2018a, 2018b, 2020, Bradford and Forney 2017, McCracken 2019). Based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2012), this animal was considered seriously injured (Bradford and Forney 2017). No pygmy sperm whales were observed hooked or entangled in the SSLL fishery (100% observer coverage). There was one additional unidentified cetacean taken in the DSLL fishery during this period that was likely a species of *Kogia,* based on the Observer's description.

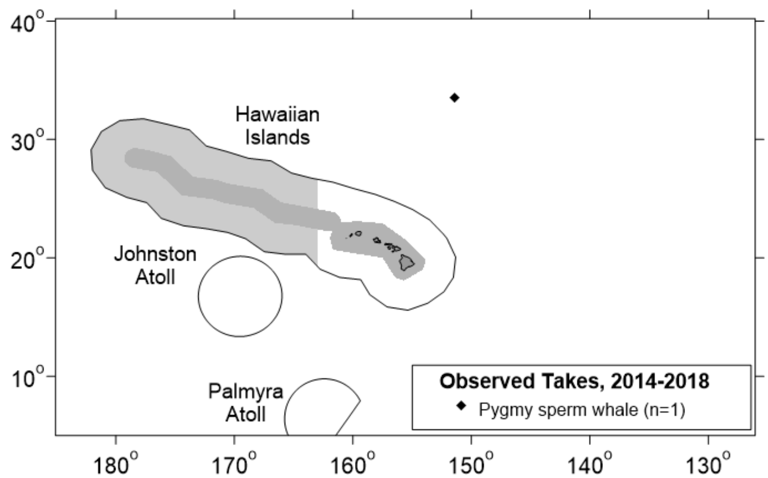


**Figure 2.** Location of pygmy or dwarf sperm whale takes (filled diamond) in Hawaii-based longline fisheries, 2014-2018. Solid lines represent the U.S. EEZs. Gray shading notes areas closed to commercial fishing, with the PMNM Expansion area closed since August 2016.

**Table 2.** Summary of available information on incidental mortality and serious injury of pygmy sperm whales (Hawaiian stock) in commercial longline fisheries within and outside of the Hawaiian Islands EEZ (McCracken 2019). Mean annual takes are based on 2014-2018 data unless otherwise indicated. Information on all observed takes (T) and combined mortality events & serious injuries (MSI) is included. Total takes were prorated to deaths, serious injuries, and non-serious injuries based on the observed proportions of each outcome.

| Fishery Name | Year | Data Type | Percent Observer Coverage | Observed total interactions (T) and mortality events, and serious injuries (MSI), and total estimated mortality and serious injury (M&SI) of pygmy sperm whales | | | |
|---|---|---|---|---|---|---|---|
| | | | | Outside U.S. EEZs | | Inside Hawaiian EEZ | |
| | | | | Obs. T/MSI | Estimated M&SI (CV) | Obs. T/MSI | Estimated M&SI (CV) |
| Hawaii-based deep-set longline fishery | 2014 | Observer data | 20% | 1/1 | 10 (0.9) | 0 | 0 (-) |
| | 2015 | | 22% | 0 | 0 (-) | 0 | 0 (-) |
| | 2016 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2017 | | 21% | 0 | 0 (-) | 0 | 0 (-) |
| | 2018 | | 20% | 0 | 0 (-) | 0 | 0 (-) |
| **Mean Estimated Annual Take (CV)** | | | | | **2.0 (1.2)** | | **0 (-)** |
| Hawaii-based shallow-set longline fishery | 2014 | Observer data | 100% | 0 | 0 | 0 | 0 |
| | 2015 | | 100% | 0 | 0 | 0 | 0 |
| | 2016 | | 100% | 0 | 0 | 0 | 0 |
| | 2017 | | 100% | 0 | 0 | 0 | 0 |
| | 2018 | | 100% | 0 | 0 | 0 | 0 |
| **Mean Annual Takes (100% coverage)** | | | | | **0** | | **0** |
| **Minimum total annual takes within U.S. EEZ** | | | | | | | **0 (-)** |

## STATUS OF STOCK

The Hawaii stock of pygmy sperm whales is not considered strategic under the 1994 amendments to the MMPA. The status of pygmy sperm whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Pygmy sperm whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. The estimated rate of mortality and serious injury within the Hawaiian Islands EEZ (zero animals per year) is less than the PBR (257). Based on the available data, which indicate total fishery-related takes are less than 10% of PBR, the total fishery mortality and serious injury for pygmy sperm whales can be considered to be insignificant and approaching zero. The increasing level of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for whales (Richardson *et al*. 1995), particularly for deep-diving whales like pygmy sperm whales that feed in the oceans' "sound channel". One pygmy sperm whale found stranded in the main Hawaiian Islands tested positive for *Morbillivirus* (Jacob 2012). Although *Morbillivirus* is known to trigger lethal disease in cetaceans (Van Bressem *et al*. 2009), its impact on the health of the stranded animal is unknown (Jacob 2012). The presence of *Morbillivirus* in 10 species of cetacean in Hawaiian waters (Jacob 2012) raises concerns about the history and prevalence of this disease in Hawaii and the potential population impacts on Hawaiian cetaceans.

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Revised 5/5/2021

# DWARF SPERM WHALE (*Kogia sima*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Dwarf sperm whales are found throughout the world in tropical to warm-temperate waters (Nagorsen 1985). Dwarf sperm whales are seen infrequently during nearshore surveys. Although they have been seen throughout the main Hawaiian Islands, they appear to be more common near Kauai, Niihau, and Oahu than around the other islands (Baird 2016). Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in five sightings of dwarf sperm whales during 2002 and one during 2010 (Figure 1; Barlow 2006, Bradford *et al*. 2013). There were no sightings of confirmed dwarf sperm whales during a 2017 survey, though five sightings of *Kogia* sp., not identified to species, were recorded (Yano *et al*. 2018).

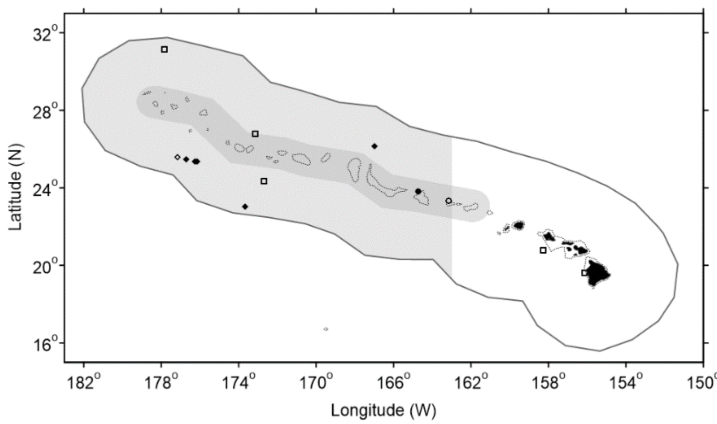


**Figure 1.** Dwarf sperm whale sighting locations during the 2002 (diamonds) and 2010 (circle) shipboard surveys, as well as sightings of unidentified *Kogia* during 2002 (open diamond), 2010 (open circle), and 2017 (open square) shipboard cetacean surveys of U.S. waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2013). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates of the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Small boat surveys within the main Hawaiian Islands (MHI) since 2002 have documented dwarf sperm whales on 73 occasions, most commonly in water depths between 500m and 1,000m (Baird *et al*. 2013). Long-term site-fidelity is evident off Hawaii Island, with one third of the distinctive individuals seen there encountered in more than one year. Resighting data from 25 individuals documented at Hawaii Island suggest an island-resident population with restricted range, with all encounters in less than 1,600m water depth and less than 20 km from shore (Baird *et al* 2013). Division of this population into a separate island-associated stock may be warranted in the future. For the Marine Mammal Protection Act (MMPA) stock assessment reports, dwarf sperm whales within the Pacific U.S. EEZ are divided into two discrete, non-contiguous areas: 1) Hawaiian waters (this report), and 2) waters off California, Oregon and Washington. The Hawaii stock includes animals found within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ was recently reevaluated for each survey year, resulting in updated abundance estimates of dwarf sperm whales in the Hawaii EEZ (Bradford *et al*. 2021; Table 1).

Table 1. Line-transect abundance estimates for dwarf sperm whales or unidentified *Kogia* derived from surveys of the entire Hawaii EEZ in 2017 (Bradford *et al*. 2021).

| Year | Species | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|---|
| 2017 | Unidentified *Kogia* | 53,421 | 0.63 | 17,083-167,056 |
| 2010 | | NA | | |
| 2002 | Dwarf sperm whale | 37,440 | 0.78 | 9,758-143,648 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for *Kogia* from Barlow *et al*. (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al*. (2021), uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available for each survey year. Wade and Gerrodette (1993) provided an estimate for the eastern tropical Pacific, but it is not known whether these animals are part of the same population that occurs in the central North Pacific. This species' small size, tendency to avoid vessels, and deep-diving habits, combined with the high proportion of *Kogia* sightings that are not identified to species, may result in negatively biased estimates of relative abundance in this region.

### Minimum Population Estimate

The log-normal 20th percentile of the 2002 abundance estimate (Bradford *et al*. 2021) is 20,953 dwarf sperm whales within the Hawaiian Islands EEZ; however, the minimum abundance estimate for the entire Hawaiian EEZ is ≥ 8 years old and will no longer be used (NMFS 2005). No minimum estimate of abundance is available for this stock, as there were no sightings of dwarf sperm whales during a 2017 shipboard line-transect survey of the Hawaiian EEZ.

### Current Population Trend

No data are available on current population abundance or trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997). Because there is no minimum population size estimate for Hawaii pelagic dwarf sperm whales, the PBR is undetermined.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No interactions between nearshore fisheries and dwarf sperm whales have been reported in Hawaiian waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, one unidentified cetacean, identified as probable pygmy or dwarf sperm whale (*Kogia* sp.) was observed hooked in the DSLL fishery (18% observer coverage) (Bradford 2020). Based on an evaluation of the observer's description of the interaction and following the most recently developed criteria for assessing serious injury in marine mammals (NMFS 2012), the injury state could not be determined (Bradford 2020). No dwarf sperm whales were observed hooked or entangled in the SSLL fishery (100% observer coverage).

## STATUS OF STOCK

The Hawaii stock of dwarf sperm whales is not considered strategic under the 1994 amendments to the MMPA. The status of dwarf sperm whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Dwarf sperm whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. There have been no reported fishery related mortality or injuries within the Hawaiian Islands EEZ, such that the total mortality and serious injury can be considered to be insignificant and approaching zero. The increasing levels of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for whales (Richardson *et al*. 1995), particularly for deep-diving whales like dwarf sperm whales that feed in the oceans' "sound channel".

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# SPERM WHALE (*Physeter macrocephalus*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Sperm whales are widely distributed across the entire North Pacific and into the southern Bering Sea in summer but the majority are thought to be south of 40ºN in winter (Rice 1974, 1989; Gosho *et al.* 1984; Miyashita *et al.* 1995). For management, the International Whaling Commission (IWC) had divided the North Pacific into two management regions (Donovan 1991) defined by a zig-zag line which starts at 150ºW at the equator to 160ºW between 40-50ºN, and ending at 180ºW north of 50ºN; however, the IWC has not reviewed this stock boundary in many years (Donovan 1991). Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly westward towards the middle of the tropical Pacific (near the IWC stock boundary at 150ºW) and tapers off northward towards the tip of Baja California. The Hawaiian Islands marked the center of a major nineteenth century whaling ground for sperm whales (Gilmore 1959; Townsend 1935). Sperm whales have been sighted throughout the Hawaiian EEZ, including nearshore waters of the main and Northwestern Hawaiian Islands (NWHI) (Rice 1960; Baird 2016, Lee 1993; Mobley *et al.* 2000, Shallenberger 1981). In addition, the sounds of sperm whales have been recorded throughout the year within the main and NWHI (Thompson and Friedl 1982, Merkens *et al*. 2019). Summer/fall shipboard surveys of waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in 43 sperm whale sightings in 2002, 46 in 2010, and 24 in 2017 throughout the study area (Figure 1; Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018).

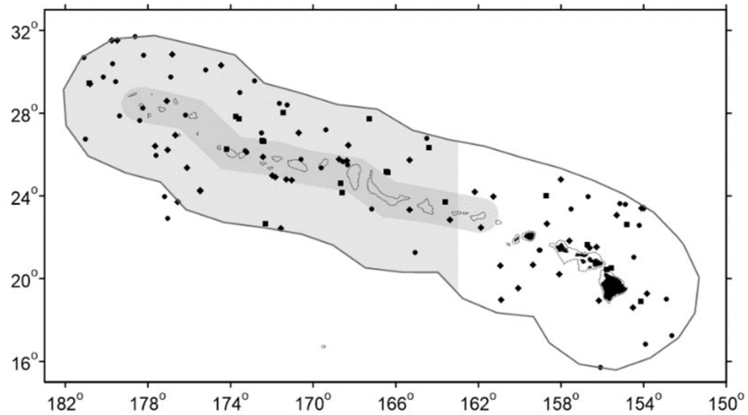


**Figure 1**. Sperm whale sighting locations during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2017, Yano *et al*. 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobaths.

The stock identity of sperm whales in the North Pacific has been inferred from historical catch records (Bannister and Mitchell 1980) and from trends in CPUE and tag-recapture data (Ohsumi and Masaki 1977). A 1997 survey designed specifically to investigate stock structure and abundance of sperm whales in the northeastern temperate Pacific revealed no apparent hiatus in distribution between the U.S. EEZ off California and areas farther west, out to Hawaii (Barlow and Taylor 2005). Genetic analyses revealed significant differences in mitochondrial and nuclear DNA and in single-nucleotide polymorphisms between sperm whales sampled off the coast of California, Oregon and Washington and those sampled near Hawaii and in the eastern tropical Pacific (ETP) (Mesnick *et al*. 2011). These results suggest demographic independence between matrilineal groups found California, Oregon, and Washington, and those found elsewhere in the central and eastern tropical Pacific. Further, assignment tests identified male sperm whales sampled in the sub-Arctic with each of the three regions, suggesting mixing of males from potentially several populations during summer (Mesnick *et al*. 2011).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, sperm whales within the Pacific U.S. EEZ are divided into three discrete, non-contiguous stocks: 1) waters around Hawaii (this report), 2) California, Oregon and Washington waters, and 3) Alaskan waters. The Hawaii stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of the Hawaii stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the entire Hawaiian Islands EEZ was recently reevaluated for each survey year, resulting in the following abundance estimates of sperm whales in the Hawaii EEZ (Becker *et al*. 2021; Table 1).

Table 1. Model-based line-transect abundance estimates for sperm whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Becker *et al*. 2021).

| Year | Model-based abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 5,707 | 0.23 | 2,961-10,998 |
| 2010 | 5,497 | 0.22 | 2,863-10,555 |
| 2002 | 5,387 | 0.22 | 2,668-10,878 |

Sighting data from 2002 to 2017 within the Hawaii EEZ were used to derive habitat-based models of animal density for the overall period. The models were then used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al*. 2015, Becker *et al*. 2016). The modeling framework incorporated Beaufort-specific trackline detection probabilities for sperm whales from Barlow *et al*. (2015). Bradford *et al*. (2021) produced design-based abundance estimates for sperm whales for each survey year that can be used as a point of comparison to the model-based estimates. While on average, the estimates are similar between the two approaches, the annual design-based estimates show much greater uncertainty for some years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Model based-estimates are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. There are insufficient data to explicitly incorporate a trend term into the model due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Previously published design-based estimates for the Hawaii EEZ from 2002 and 2010 surveys (e.g. Barlow 2006, Bradford *et al*. 2017) used a subset of the dataset used by Becker *et al*. (2021) and Bradford *et al*. (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is from the 2017 survey, or 5,707 (CV=0.23).

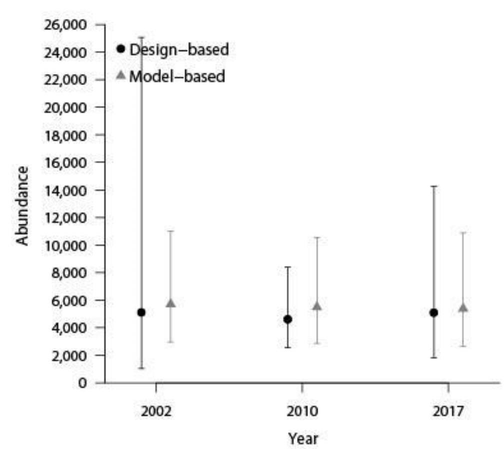


**Figure 2.** Comparison of design-based (circles, Bradford *et al*. 2021) and model-based (triangles, Becker *et al*. 2021) estimates of abundance for sperm whales for each survey year (2002, 2010, 2017).

A large 1982 abundance estimate for the entire eastern North Pacific (Gosho *et al*. 1984) was based on a CPUE method which is no longer accepted as valid by the International Whaling Commission. A spring 1997 combined visual and acoustic line-transect survey conducted in the eastern temperate North Pacific resulted in estimates of 26,300 (CV=0.81) sperm whales based on visual sightings, and 32,100 (CV=0.36) based on acoustic detections and visual group size estimates (Barlow and Taylor 2005). Sperm whales appear to be a good candidate for acoustic surveys due to the increased range of detection; however, visual estimates of group size are still required (Barlow and Taylor 2005). In the eastern tropical Pacific, the abundance of sperm whales has been estimated as 22,700 (95% CI = 14,800 - 34,600; Wade and Gerrodette 1993). However, it is not known whether any or all of these animals routinely enter the U.S. EEZ of the Hawaiian Islands.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) around the 2017 abundance estimate or 4,486 sperm whales within the Hawaiian Islands EEZ.

**Current Population Trend**

The model-based abundance estimates for sperm whales provided by Becker *et al.* (2021) do not explicitly allow for examination of population trend other than that driven by environmental factors. Model-based examination of sperm whale population trends including sighting data beyond the Hawaii EEZ will be required to more fully examine trend for this stock. .

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data on current or maximum net productivity rate are available.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of sperm whales is calculated as the minimum population size (4,486) within the U.S. EEZ of the Hawaiian Islands <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.2 (for an endangered species with $N_{min} > 1,500$ and $CV_N <= 0.50$, with low vulnerability to extinction; (Taylor *et al*. 2003), resulting in a PBR of 18 sperm whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. No estimates of human-caused mortality or serious injury are currently available for nearshore hook and line fisheries because these fisheries are not observed or monitored for protected species bycatch.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no sperm whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or in the DSLL fishery (18-21% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017). On unidentified cetacean taken in the DSLL fishery was identified as a large whale based on the Observer's description and may have been a sperm whale.

**Historical Mortality**

Between 1800 and 1909, about 60,842 sperm whales were estimated taken in the North Pacific (Best 1976). The reported take of North Pacific sperm whales by commercial whalers between 1947 and 1987 totaled 258,000 (C. Allison, pers. comm.). Factory ships operated as far south as 20°N (Ohsumi 1980). Ohsumi (1980) lists an additional 28,198 sperm whales taken mainly in coastal whaling operations from 1910 to 1946. Based on the massive under-reporting of Soviet catches, Brownell *et al*. (1998) estimated that about 89,000 whales were additionally taken by the Soviet pelagic whaling fleet between 1949 and 1979. Japanese coastal operations apparently also under-reported catches by an unknown amount (Kasuya 1998). Thus a total of at least 436,000 sperm whales were taken between 1800 and the end of commercial whaling for this species in 1987. Of this total, an estimated 33,842 were taken by Soviet and Japanese pelagic whaling operations in the eastern North Pacific from the longitude of Hawaii to the U.S. West coast, between 1961 and 1976 (Allen 1980, IWC statistical Areas II and III), and 965 were reported taken in land-based U.S. West coast whaling operations between 1947 and 1971 (Ohsumi 1980). In addition, 13 sperm whales were taken by shore whaling stations in California between 1919 and 1926 (Clapham *et al*. 1997). There has been a prohibition on taking sperm whales in the North Pacific since 1988, but large-scale pelagic whaling stopped earlier, in 1980. Some of the whales taken during the whaling era were certainly from a population or populations that occur within Hawaiian waters.

## STATUS OF STOCK

The only estimate of the status of North Pacific sperm whales in relation to carrying capacity (Gosho *et al*. 1984) is based on a CPUE method no longer accepted as valid. The status of sperm whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Sperm whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the Hawaiian stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries in U.S. EEZ waters, total fishery mortality and serious injury for sperm whales can be considered insignificant and approaching zero. The increasing level of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for whales (Richardson *et al*. 1995), particularly for deep-diving whales like sperm whales that feed in the oceans' "sound channel". One sperm whale stranded in the main Hawaiian Islands tested positive for both *Brucella* and *Morbillivirus* (Jacob *et al*. 2016). *Brucella* is a bacterial infection that if common in the population may limit recruitment by compromising male and female reproductive systems, and can also cause neurological disorders that may result in death (Van Bressem *et al*. 2009). *Morbillivus* is known to trigger lethal disease in cetaceans (Van Bressem *et al*. 2009); however, investigation of the pathology of the stranded sperm whale suggests that *Brucella* was more likely the cause of death in this sperm whale. The presence of *Morbillivirus* in 10 species (Jacob *et al*. 2016) and *Brucella* in 3 species (Cherbov 2010) raises concerns about the history and prevalence of these diseases in Hawaii and the potential population impacts on Hawaiian cetaceans. It is not known if *Brucella or Morbillivirus* are common in the Hawaii stock.

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Revised 5/25/2018

# BLUE WHALE (*Balaenoptera musculus musculus*): Central North Pacific Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

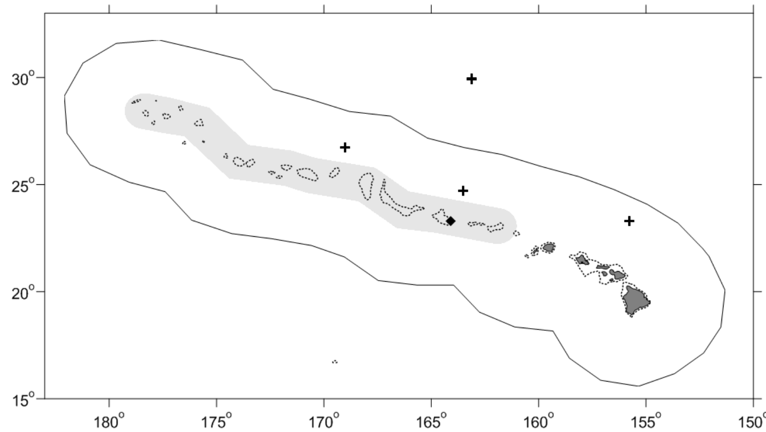


**Figure 1.** Locations of blue whale sightings made by observers aboard Hawaii-based longline fishing vessels between July 1994 and December 2009 (crosses, NMFS/PIR unpublished data), and location of a single blue whale sighting during a 2010 (black diamond) shipboard cetacean survey of U.S. EEZ waters surrounding the Hawaiian Islands (Bradford et al. 2017; see Appendix 2 for details on timing and location of survey effort). Outer line indicates approximate boundary of survey area and U.S. EEZ. Gray shading indicates area of Papahanaumokuakea Marine National Monument. Dotted line represents the 1000 m isobath.

The International Whaling Commission (IWC) has formally considered only one management stock for blue whales in the North Pacific (Donovan 1991), but up to five populations have been proposed (Reeves et al. 1998). Rice (1974) hypothesized that blue whales from Baja California migrated far offshore to feed in the eastern Aleutians or Gulf of Alaska and returned to feed in California waters; though more recently concluded that the California population is separate from the Gulf of Alaska population (Rice 1992). Length frequency analyses (Gilpatrick et al. 1996) and photo-identification studies (Calambokidis et al. 1995) through the 1990s supported separate populations for blue whales feeding off California and those feeding in Alaskan waters. Whaling catch data indicated that whales feeding along the Aleutian Islands were probably part of a central Pacific stock (Reeves et al. 1998), which was thought to migrate to offshore waters north of Hawaii in winter (Berzin and Rovnin 1966). Blue whale feeding aggregations have not been found in Alaska despite several surveys (Leatherwood et al. 1982; Stewart et al. 1987; Forney and Brownell 1996). More recently, analyses of acoustic data obtained throughout the North Pacific (Stafford et al. 2001; Stafford 2003) have revealed two distinct blue whale call types, suggesting two North Pacific stocks: eastern and central (formerly western). The regional occurrence patterns suggest that blue whales from the eastern North Pacific stock winter off Mexico, Central America, and as far south as 8° S (Stafford et al. 1999), and feed during summer off the U. S. West Coast and to a lesser extent in the Gulf of Alaska. This stock has previously been observed to feed in waters off California (and occasionally as far north as British Columbia; Calambokidis et al. 1998) in summer/fall (from June to November) migrating south to productive areas off Mexico (Calambokidis et al. 1990) and as far south as the Costa Rica Dome (10° N) in winter/spring (Mate et al. 1999, Stafford et al. 1999). Blue whales belonging to the central Pacific stock appear to feed in summer southwest of Kamchatka, south of the Aleutians, and in the Gulf of Alaska (Stafford 2003; Watkins et al. 2000), and in winter migrate to lower latitudes in the western and central Pacific, including Hawaii (Stafford et al. 2001).

The first published sighting record of blue whales near Hawaii is that of Berzin and Rovnin (1966), though recently, two blue whales were seen with fin whales and an unidentified rorqual in November 2010 during a survey of Hawaiian U.S. EEZ waters (Bradford et al. 2017). Four sightings have been made by observers on Hawaii-based longline vessels (Figure 1; NMFS/PIR, unpublished data). Additional evidence that blue whales occur in this area comes from acoustic recordings made off Oahu and Midway Islands (Northrop et al. 1971; Thompson and Friedl 1982), which likely included at least some whales within the EEZ. The recordings made off Hawaii showed bimodal peaks throughout the year (Stafford et al. 2001), with central Pacific call types heard during winter and eastern Pacific calls heard during summer. For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are two blue whale stocks within the Pacific U.S. EEZ: 1) the central North Pacific stock (this report), which includes whales found around the Hawaiian Islands during winter and 2) the eastern North Pacific stock, which feeds primarily off California.

## POPULATION SIZE

A 2010 shipboard line-transect survey of the entire Hawaiian Islands EEZ resulted in a summer/fall abundance estimate of 133 (CV = 1.09) blue whales (Bradford et al. 2017). This is currently the best available abundance estimate for this stock within the Hawaii EEZ, but the majority of blue whales would be expected to be at higher latitudes feeding grounds at this time of year. Wade and Gerrodette (1993) estimated 1,400 blue whales for the eastern tropical Pacific from summer-fall line-transect surveys in the 1980s, though it is unclear how much overlap there is between blue whales there and those found near Hawaii. No blue whale sightings were made during summer/fall 2002 shipboard surveys of the entire Hawaiian Islands EEZ (Barlow 2006).

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow et al. 1995) of the 2010 abundance estimate, or 63 blue whales within the Hawaiian Islands EEZ.

### Current Population Trend

The first sightings of blue whales during systematic surveys occurred in 2010, and there is currently insufficient data to assess population trends.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Central North Pacific stock of blue whales is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (63) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (the default value for an endangered species with $N_{min}$ <1500; Taylor et al. 2003), resulting in a PBR of 0.1 Central Pacific blue whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2011 and 2015, no blue whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (20-22% observer coverage) (Bradford 2017, Bradford and Forney 2017, McCracken 2017).

### Historical Mortality

At least 9,500 blue whales were taken by commercial whalers throughout the North Pacific between 1910 and 1965 (Ohsumi and Wada 1972). Some proportion of this total may have been from a population or populations that migrate seasonally into the Hawaiian EEZ. The species has been protected in the North Pacific by the IWC since 1966.

## STATUS OF STOCK

The status of blue whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Blue whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the central Pacific stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. Because there have been no reported fishery related mortality or serious injuries of blue whales within the Hawaiian Islands EEZ, the total fishery-related mortality and serious injury of this stock can be considered to be insignificant and approaching zero. Increasing levels of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for blue whales (Reeves et al. 1998). Tagged blue whales exposed to simulated mid-frequency sonar and pseudo-random noise demonstrated a variety of behavioral responses, including no change in behavior, termination of deep dives, directed travel away from sound sources, and cessation of feeding (Goldbogen et al. 2013). Behavioral responses were highly dependent upon the type of sound source and the behavioral state of the animal at the time of exposure (Friedlaender et al. 2016), with more clear and significant response from deep-feeding whales than those in other behavioral states.. The authors stated that behavioral responses to such sounds are influenced by a complex interaction of behavioral state, environmental context, and prior exposure of individuals to such sound sources. One concern expressed by the authors is if blue whales did not habituate to such sounds near feeding areas that "repeated exposures could negatively impact individual feeding performance, body condition and ultimately fitness and potentially population health." Currently, no evidence indicates that such reduced population health exists, but such evidence would be difficult to differentiate from natural sources of reduced fitness or mortality in the population.

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Revised 5/5/2021

# FIN WHALE (*Balaenoptera physalus velifera*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

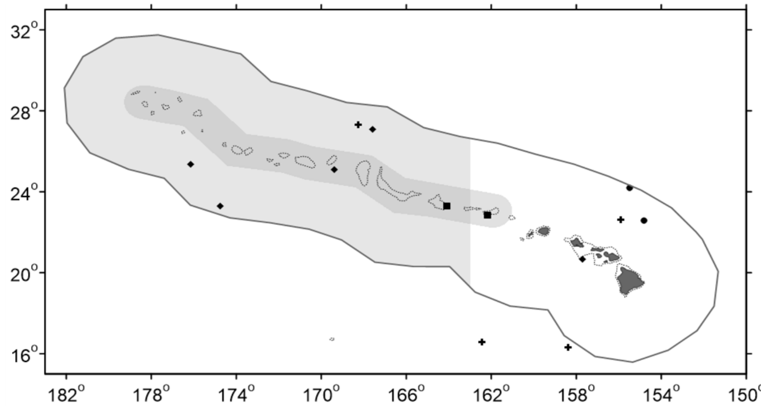


**Figure 1**. Locations of fin whale sightings from longline observer records (crosses; NMFS/PIR, unpublished data) and sighting locations during the 2002 (diamonds), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al.* 2017, Yano *et al.* 2018). Outer line represents approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates of the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Fin whales are found from temperate to subpolar ocean around the world, with a distributional hiatus between the Northern and Southern Hemispheres between 20º and 30º on either side of the equator (Edwards *et al*. 2015). In the North Pacific fin whales tend to occur in temperate to sub-polar latitudes (Mizroch *et al*. 1984, 2009). North Pacific fin whales are genetically distinct from those in the North Atlantic and Southern Hemisphere (Archer *et al*. 2013). Archer *et al*. (2019a) used mitochondrial DNA and single-nucleotide polymorphisms (SNPs) to demonstrate that North Atlantic and North Pacific genetic samples could be correctly assigned to their respective ocean basins with 99% accuracy. North Pacific fin whales are recognized as a separate subspecies: *Balaenoptera physalus velifera*. Fin whales are considered rare in Hawaiian waters, though occasional sightings have been reported. Balcomb (1987) observed 8-12 fin whales in a multispecies feeding assemblage on 20 May 1966 approx. 250 mi. south of Honolulu. Additional sightings were reported north of Oahu in May 1976, in the Kauai Channel in February 1979 (Shallenberger 1981), north of Kauai in February 1994 (Mobley *et al.* 1996), and off Lanai in 2012 (Baird 2016). Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in five sightings in 2002, two sightings in 2010, and two sightings in 2017 (Figure 1; Barlow 2006, Bradford *et al*. 2017, Yano *et al.* 2018). Oleson *et al*. (2014) reported fin whale song recorded near Hawaii Island from October through April, and noted the occurrence of a predominant song pattern similar to that recorded off southern California and in the Bering Sea, suggesting a broadly connected population throughout that range. Further examination of the spectral features of individual fin whale song notes and the inter-note timing within fin whale song throughout the Pacific indicates a broad diversity of song types, with songs recorded in Hawaii being distinctly different than those heard at the other eastern and central Pacific monitoring sites or in arctic waters (Archer *et al*. 2019b)

The International Whaling Commission (IWC) recognized two stocks of fin whales in the North Pacific: the East China Sea and the rest of the North Pacific (Donovan 1991). Mizroch *et al.* (1984) cite evidence for additional fin whale subpopulations in the North Pacific, and acoustic evidence provides additional support for finer population structure (Archer *et al*. 2019b). There is still insufficient information to accurately determine population structure, but from a conservation perspective it may be risky to assume panmixia in the entire North Pacific. In the North Atlantic, fin whales were locally depleted in some feeding areas by commercial whaling (Mizroch *et al.* 1984), in part because subpopulations were not recognized. The Marine Mammal Protection Act (MMPA) stock assessment reports recognize three stocks of fin whales in the North Pacific: 1) the Hawaii stock (this report), 2) the California/Oregon/Washington stock, and 3) the Alaska stock. The Hawaiian stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from summer/fall shipboard line-transect surveys of the entire Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in the following abundance estimates of fin whales in the Hawaii EEZ (Bradford *et al.* 2021; Table 1).

Table 1. Line-transect abundance estimates for fin whales derived from surveys of the entire Hawaii EEZ in 2002, 2010, and 2017 (Bradford *et al.* 2021).

| Year | Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2017 | 203 | 0.99 | 40-1,028 |
| 2010 | 158 | 1.07 | 29-871 |
| 2002 | 509 | 0.73 | 141-1,842 |

The updated design-based abundance estimates use sighting data from throughout the central Pacific to estimate the detection function and use Beaufort sea-state-specific trackline detection probabilities for fin whales from Barlow *et al.* (2015). Although previous estimates from the Hawaii EEZ have been published using subsets of this data, Bradford *et al.* (2021), uses a consistent approach for estimating all abundance parameters and resulting estimates are considered the best available for the summer/fall period. Winter surveys have not been carried out in the Hawaiian Archipelago and it is likely that winter abundance of this migratory species within the Hawaiian EEZ is not accurately reflected by the summer/fall estimates. The best estimate of abundance is 203 (CV=0.99) fin whales based on a 2017 survey (Bradford *et al.* 2021). Using passive acoustic detections from a hydrophone north of Oahu, MacDonald and Fox (1999) estimated an average density of 0.027 calling fin whales per 1000 $km^2$ within about 16 km from shore. However, the relationship between the number of whales present and the number of calls detected is not known, and therefore this acoustic method does not provide an estimate of absolute abundance for fin whales.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al*. 1995) around the 2017 abundance estimate or 101 fin whales within the Hawaiian Islands EEZ during the summer/fall survey period.

### Current Population Trend

Trend information for this stock cannot be assessed from summer/fall abundance surveys alone, as the species is not expected to reside in Hawaiian waters in large numbers during that period. Winter surveys or alternative observations (e.g. acoustic studies) will be required to assess the trend for fin whales in Hawaiian waters.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of fin whales is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (101) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.1 (the default value for an endangered species with Nmin <1500; Taylor *et al*. 2003), resulting in a PBR of 0.2 fin whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, one fin whale was observed entangled in the SSLL fishery (100% observer coverage), and none were observed in the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford & Forney 2017, McCracken 2019). The SSLL entanglement occurred outside of the Hawaiian Islands EEZ and the whale was judged to be not seriously injured (Bradford and Forney 2017). The 5-yr annual mortality and serious injury estimate for fin whales is 0 both inside and outside the Hawaiian Islands EEZ (McCracken 2019). Two additional unidentified cetaceans judged to be large whales based on the observer's description were taken in the DSLL, and some of these may have been fin whales.

**Historical Mortality**

Large numbers of fin whales were taken by commercial whalers throughout the North Pacific from the early 20th century until the 1970s (Tønnessen and Johnsen 1982). Approximately 46,000 fin whales were taken from the North Pacific by commercial whalers between 1947 and 1987 (C. Allison, IWC, pers. comm.). Some of the whales taken may have been from a population or populations that migrate seasonally into the Hawaiian EEZ. The species has been protected in the North Pacific by the IWC since 1976.

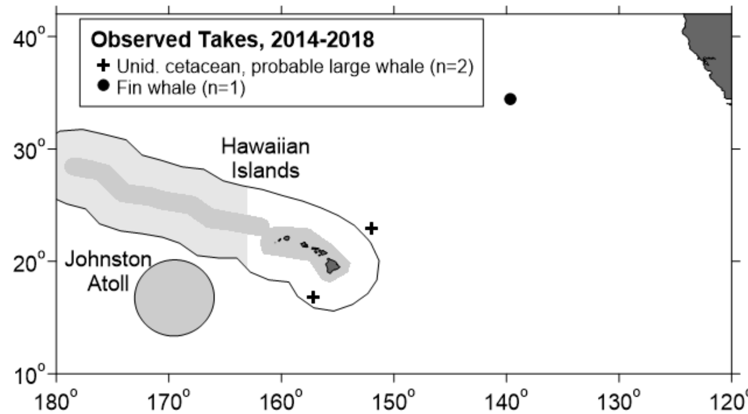


**Figure 2.** Location of observed fin whale take within the shallow-set fishery (circle), and unidentified cetaceans considered to be large whales based on the observer's description (crosses) in the deep-set fishery, 2014-2018. Solid lines represent the U. S. EEZ. Gray shading notes areas closed to commercial fishing, with the PMNM Expansion area closed since August 2016.

**STATUS OF STOCK**

The status of fin whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Fin whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the Hawaiian stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. Because there have been no reported fishery related mortality or serious injuries within the Hawaiian Islands EEZ, the total fishery-related mortality and serious injury of this stock can be considered to be insignificant and approaching zero. Increasing levels of anthropogenic sound in the world's oceans has been suggested to be a habitat concern for whales, particularly for baleen whales that may communicate using low-frequency sound (Croll *et al.* 2002). Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al.* 2013), but it is unknown if fin whales respond in the same manner to such sounds.

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Revised 05/31/2024

# BRYDE'S WHALE (*Balaenoptera edeni*): Hawai‘i Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

Bryde's whales occur in tropical and warm temperate waters throughout the world. Leatherwood *et al.* (1982) described the species as relatively abundant in summer and fall on the Mellish and Miluoki banks northeast of Hawai‘i and around Midway Island. Ohsumi and Masaki (1975) reported the tagging of "many" Bryde's whales between the Bonin and Hawaiian Islands in the winters of 1971 and 1972 (Ohsumi 1977). Periodic shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands have regularly encountered Bryde’s whales throughout the EEZ (Figure 1). There is currently no biological basis for defining separate stocks of Bryde's whales in the central North Pacific. Bryde's whales were seen occasionally off southern California (Morejohn and Rice 1973) in the 1960s, but their seasonal occurrence has increased since at least 2000 based on detection of their distinctive calls (Kerosky *et al.* 2012).

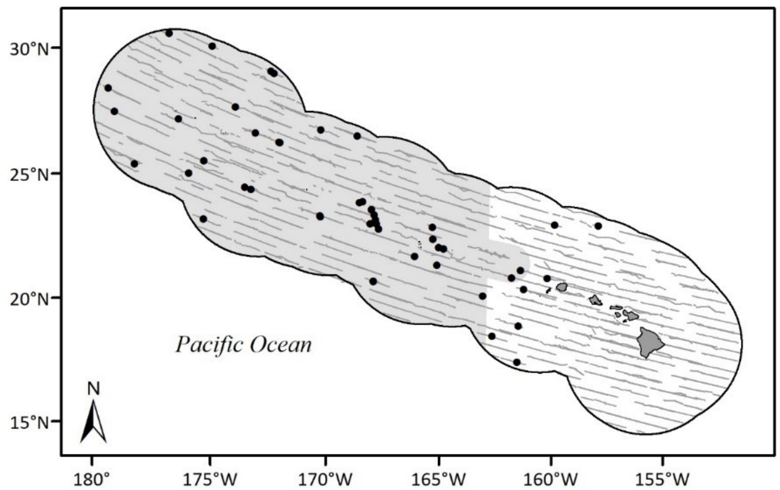


**Figure 1**. Bryde’s whale sighting locations (circles) and survey effort (gray lines) during the 2002 (Barlow 2006), 2010 (Bradford *et al.* 2017), and 2017 (Yano *et al. 2018*) shipboard surveys of the U.S. EEZ around the Hawaiian Islands (outer black line). The Papahānaumokuākea Marine National Monument in the western portion of the EEZ is shaded gray.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, Bryde's whales within the Pacific U.S. EEZ are divided into two areas: 1) Hawaiian waters (this report), and 2) the eastern Pacific (east of 150ºW and including the Gulf of California and waters off California). The Hawai‘i stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from the U.S. EEZ of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from shipboard line-transect surveys of the Hawaiian Islands EEZ were recently reevaluated for each survey year, resulting in updated model-based abundance estimates of Bryde's whales in the entirety of the Hawaiian Islands EEZ (Becker *et al.* 2021, 2022) (Table 1).

Sighting data from 2002 to 2020 within the Hawaiian Islands EEZ were used to derive habitat-based models of animal density for two periods: 2002-2017 (Becker et al. 2021) and 2017-2020 (Becker *et al*. 2022). The most recent set of models include three notable changes from the 2002-2017 models: use of calibrated group size estimates, as in Bradford *et al*. (2021), exclusion of a spatial term on model selection, requiring more explicit reliance on environmental variables, and incorporating new approaches (Miller *et al*. 2022) for more comprehensively estimating uncertainty in model predictions that account for the combined uncertainty around all parameter estimates. The modeling framework incorporated Beaufort-specific trackline detection probabilities for Bryde’s whales from Barlow *et al*. (2015). Models were used to predict density and abundance for each survey year based on the environmental conditions within that year (see Forney *et al.* 2015, Becker *et al.* 2016).

**Table 1.** Model-based line-transect abundance estimates for Bryde's whales in the Hawaiian Islands EEZ in 2002 and 2010 (Becker *et al.* 2021) and 2017 and 2020 (Becker *et al*. 2022), derived from NMFS surveys in the central Pacific

since 2000. The Becker *et al*. (2022) analysis incorporates a more comprehensive model-based approach to estimating model uncertainty, such that the CVs and 95% confidence limits for 2002/2010 and 2017/2020 are not directly comparable.

| Year | Model-based Abundance | CV | 95% Confidence Limits |
|---|---|---|---|
| 2020 | 791 | 0.29 | 456-1,372 |
| 2017 | 679 | 0.29 | 392-1,175 |
| 2010 | 822 | 0.20 | 554-1,220 |
| 2002 | 562 | 0.21 | 375-842 |

Bradford *et al.* (2021) produced design-based abundance estimates for Bryde's whales in 2002, 2010, and 2017 that can be used as a point of comparison to the model-based estimates for those years. While on average, the estimates are broadly similar between the two approaches, the annual design-based estimates show much greater variability between years than do the model-based estimates (Figure 2). The model-based approach reduces variability through explicit examination of habitat relationships across the full dataset, while the design-based approach evaluates encounter data for each year separately and thus is more susceptible to the effects of encounter rate variation. Bradford *et al.* (2021) found through simulation that the pronounced decrease in the design-based estimates between 2010 and 2017 could not be explained by encounter rate variation alone and likely reflected true changes in distribution of Bryde's whales in the study area between those survey years. The model based-estimates demonstrated a much smaller decrease between 2010 and 2017, but are based on the implicit assumption that changes in abundance are attributed to environmental variability alone. Explicitly incorporating a trend term into the model is not possible due to the insufficient sample size to test for temporal effects. Despite not fully accounting for inter-annual variation in total abundance, the model-based estimates are considered the best available estimate for each survey year. Previously published abundance estimates for the Hawaiian Islands EEZ (Barlow 2006, Becker *et al.* 2012, Forney *et al.* 2015, Bradford *et al.* 2017) used a subset of the dataset used by Becker *et al.* (2021, 2022) and Bradford *et al.* (2021) to derive line-transect parameters, such that these estimates have been superseded by the estimates presented here. The best estimate of abundance is based on the 2020 survey, or 791 (CV=0.29) Bryde's whales.

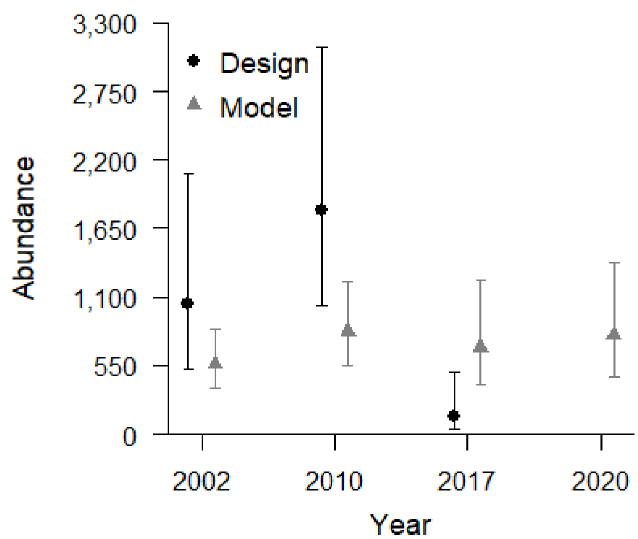


**Figure 2.** Comparison of design-based (black circles, Bradford *et al.* 2021) and model-based (gray triangles, Becker *et al.* 2021, 2022) estimates of abundance for Hawai'i Bryde's whales for each survey year (2002, 2010, 2017, 2020).

Tillman (1978) concluded from Japanese and Soviet CPUE data that the stock size in the North Pacific pelagic whaling grounds, mostly to the west of the Hawaiian Islands, declined from approximately 22,500 in 1971 to 17,800 in 1977. An estimate of 13,000 (CV=0.202) Bryde's whales was made from vessel surveys in the eastern tropical Pacific between 1986 and 1990 (Wade and Gerrodette 1993). The area to which this estimate applies is mainly east and somewhat south of the Hawaiian Islands, and it is not known whether these animals are part of the same population that occurs around the Hawaiian Islands.

## Minimum Population Estimate

The minimum population estimate is calculated as the lower $20^{th}$ percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2020 abundance estimate (Becker *et al.* 2022), or 623 Bryde's whales.

## Current Population Trend

The model-based abundance estimates for Bryde's whales provided by Becker *et al.* (2021, 2022) do not explicitly allow for examination of population trend other than that driven by environmental factors. Although annual encounter rate variation may have a large impact on abundance estimates for species with low density and patchy distribution, Bradford *et al.* (2021) suggest that the very high sighting rate in 2010 and very low sighting rate in 2017 cannot be explained through encounter rate variation alone and that there may be true fluctuations in Bryde's whale abundance within the Hawaiian Islands EEZ. Model-based examination of Bryde's whale population trends including sighting data beyond the Hawaiian Islands EEZ will be required to more fully examine trend for this stock.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No data are available on current or maximum net productivity rate for this species in Hawaiian waters.

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) level for the Hawaii stock of Bryde's whales is calculated as the minimum population size within the U.S EEZ of the Hawaiian Islands (623) times one half the default maximum net growth rate for cetaceans (½ of 4%) times a recovery factor of 0.50 (for a stock of unknown status with no known fishery mortality or serious injury within the Hawaiian Islands EEZ; Wade and Angliss 1997), resulting in a PBR of 6.2 Bryde's whales per year.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Fishery Information**

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas, but are prohibited from operating within the Papahānaumokuākea Marine National Monument (PMNM) and within the Longline Exclusion Zone around the main Hawaiian Islands and the Pacific Remote Islands and Atolls (PRIA) MNM around Johnston Atoll. The PMNM originally included the waters within a 50 nmi radius around the Northwestern Hawaiian Islands. In August, 2016, the PMNM area was expanded to extend to the 200 nmi EEZ boundary west of 163° W. Between 2017 and 2021, no Bryde's whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (15-21% observer coverage) (McCracken and Cooper 2022). Balaenopterid whales have been observed entangled in longline gear off the Hawaiian Islands in the past (Bradford 2018).

**Historical Mortality**

Small numbers of Bryde's whales were taken near the Northwestern Hawaiian Islands by Japanese and Soviet whaling fleets in the early 1970s (Ohsumi 1977). Pelagic whaling for Bryde's whales in the North Pacific ended after the 1979 season (IWC 1981), and coastal whaling for this species ended in the western Pacific in 1987 (IWC 1989).

**STATUS OF STOCK**

The Hawaiʻi stock of Bryde's whales is not considered strategic under the 1994 amendments to the MMPA. The status of Bryde's whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Bryde's whales are not listed as "threatened" or "endangered" under the Endangered Species Act (1973), nor designated as "depleted" under the MMPA. Given the absence of recent recorded fishery-related mortality or serious injuries within the Hawaiian Islands EEZ, the total fishery mortality and serious injury can be considered to be insignificant and approaching zero. The increasing level of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for large whales (Richardson *et al.* 1995, Weilgart 2007).

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Revised 6/1/2018

# SEI WHALE (Balaenoptera borealis borealis): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) only considers one stock of sei whales in the North Pacific (Donovan 1991), but some evidence exists for multiple populations (Masaki 1977; Mizroch *et al.* 1984; Horwood 1987). Sei whales are distributed far out to sea in temperate regions of the world and do not appear to be associated with coastal features. Whaling effort for this species was distributed continuously across the North Pacific between 45-55ºN (Masaki 1977). Two sei whales that were tagged off California were later killed in whaling operations off Washington and British Columbia (Rice 1974) and the movement of tagged animals has been noted in many other regions of the North Pacific. There is still insufficient information to accurately determine population structure, but from a conservation perspective it may be risky to assume panmixia in the entire North Pacific. Summer/fall shipboard surveys of the waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands resulted in four sightings in 2002 and three in 2010 (Figure 1; Barlow 2003; Bradford et al 2017).

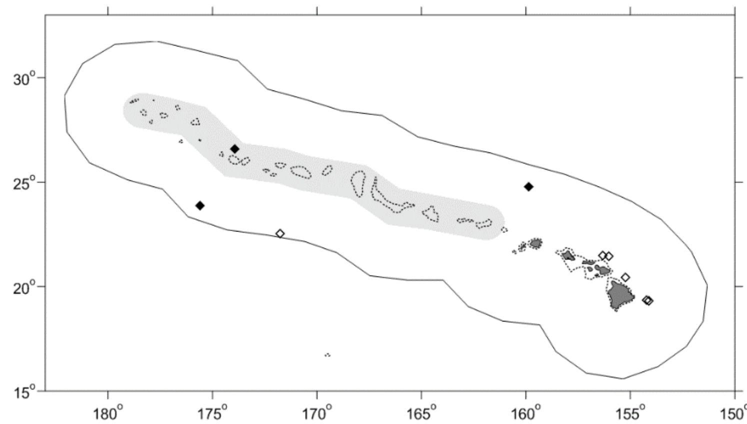


**Figure 1.** Sei whale sighting locations during the 2002 (open diamonds) and 2010 (black diamonds) shipboard cetacean surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2003, Bradford *et al.* 2017; see Appendix 2 for details on timing and location of survey effort). Outer line indicates approximate boundary of survey area and U.S. EEZ. Gray shading indicates area of Papahanaumokuakea Marine National Monument. Dotted line represents the 1000 m isobath.

For the Marine Mammal Protection Act (MMPA) stock assessment reports, sei whales within the Pacific U.S. EEZ are divided into three discrete, non-contiguous areas: 1) waters around Hawaii (this report), 2) California, Oregon and Washington waters, and 3) Alaskan waters. The Hawaiian stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from a 2010 shipboard line-transect survey of the entire Hawaiian Islands EEZ was recently evaluated using Beaufort sea-state-specific trackline detection probabilities for sei whales, resulting in an abundance estimate of 391 (CV = 0.9) sei whales (Bradford *et al.* 2017) in the Hawaii stock. This is currently the best available abundance estimate for this stock, but the majority of sei whales would be expected to be in higher-latitude feeding grounds at this time of year. A 2002 shipboard line-transect survey of the same area resulted in a summer/fall abundance estimate of 77 (CV=1.06) sei whales (Barlow 2003). Species abundances estimated from the 2002 HICEAS survey used pooled small dolphin, large dolphin, and large whale g(0) (the probability of sighting and recording an animal directly on the track line) estimates stratified by group size (Barlow 1995). Since then, Barlow (2015) developed a more robust method for estimating species-specific g(0) values that are adjusted for the Beaufort sea states that are encountered during a survey. This new method was used for analyzing the data from the 2010 survey, but has not yet been used to analyze the 2002 data. Ohsumi and Wada (1974) estimate the pre-whaling abundance of sei whales to be 58,000-62,000 in the North Pacific. Later, Tillman (1977) used a variety of different methods to estimate the abundance of sei whales in the North Pacific and revised this pre-whaling estimate to 42,000. His estimates for the year 1974, following 27 years of whaling, ranged from 7,260 to 12,620. All methods depend on using the history of catches and trends in CPUE or sighting rates; there have been no direct estimates of sei whale abundance in the entire North Pacific based on sighting surveys.

### Minimum Population Estimate

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2010 abundance estimate or 204sei whales within the Hawaiian Islands EEZ.

### Current Population Trend

No data are available on current population trend. Although the population in the North Pacific is expected to have grown since being given protected status in 1976, the possible effects of continued unauthorized takes (Yablokov 1994) make this uncertain. Abundance analyses of the 2002 and 2010 datasets used different g(0) values. The 2002 survey data have not been reanalyzed using this method. This change precludes evaluation of population trends at this time. Assessment of population trend will likely require additional survey data and reanalysis of all datasets using comparable methods.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for sei whales.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for this stock is calculated as the minimum population size within the U.S. EEZ of the Hawaiian Islands (204) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.1 (the default value for an endangered species with Nmin <1500; Taylor et al 2003), resulting in a PBR of 0.4 sei whales per year.

## HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

### Fishery Information

Information on fishery-related mortality of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. In March 2011 a subadult sei whale was found near Lahaina, Maui entangled with one or two wraps of heavy-gauge polypropylene line around the tailstock and trailing about 30 feet of line including a large bundle (Bradford & Lyman 2015). Closer examination also revealed line scars on the body near the dorsal fin. Although disentanglement was attempted, the gear could not be removed. Although the source of the line entangling the whale could not be determined, this injury is considered serious based on extent of trailing gear and condition of the whale (Bradford & Lyman 2015, NMFS 2012). This serious injury record results in a 5-yr average annual serious injury and mortality rate of 0.2 sei whales for the period 2011 to 2015.

There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2011 and 2015, no sei whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (20-21% observer coverage) (Bradford 2017, Bradford and Forney 2017, McCracken 2017).

### Historical Whaling

The reported take of North Pacific sei whales by commercial whalers totaled 61,500 between 1947 and 1987 (C. Allison, IWC, pers. comm.). There has been an IWC prohibition on taking sei whales since 1976, and commercial whaling in the U.S. has been prohibited since 1972.

## STATUS OF STOCK

Previously, sei whales were estimated to have been reduced to 20% (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). Sei whales are formally listed as "endangered" under the Endangered Species Act (ESA), and consequently the Hawaiian stock is automatically considered as a "depleted" and "strategic" stock under the MMPA. The observed rate of fisheries related mortality or serious injury within the Hawaiian Islands EEZ (0.2 animals per year) is less than the PBR (0.4). The increasing level of anthropogenic noise in the world's oceans has been suggested to be a habitat concern for whales (Richardson *et al.* 1995 Behavioral changes associated with exposure to simulated mid-frequency sonar, including no change in behavior, cessation of feeding, increased swimming speeds, and movement away from simulated sound sources has been documented in tagged blue whales (Goldbogen *et al.* 2013), but it is unknown if sei whales respond in the same manner to such sounds.

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Revised 5/5/2021

# MINKE WHALE (*Balaenoptera acutorostrata scammoni*): Hawaii Stock

## STOCK DEFINITION AND GEOGRAPHIC RANGE

The International Whaling Commission (IWC) recognizes 3 stocks of minke whales in the North Pacific: one in the Sea of Japan/East China Sea, one in the rest of the western Pacific west of 180ºN, and one in the "remainder" of the Pacific (Donovan 1991). The "remainder" stock only reflects the lack of exploitation in the eastern Pacific and does not imply that only one population exists in that area (Donovan 1991). In the "remainder" area, minke whales are relatively common in the Bering and Chukchi seas and in the Gulf of Alaska, but are not considered abundant in any other part of the eastern Pacific (Leatherwood *et al*. 1982, Brueggeman *et al*. 1990). In the Pacific, minke whales are usually seen over continental shelves (Brueggeman *et al*. 1990). In the extreme north, minke whales are believed to be migratory, but in inland waters of Washington and in central California they appear to establish home ranges (Dorsey *et al*. 1990).

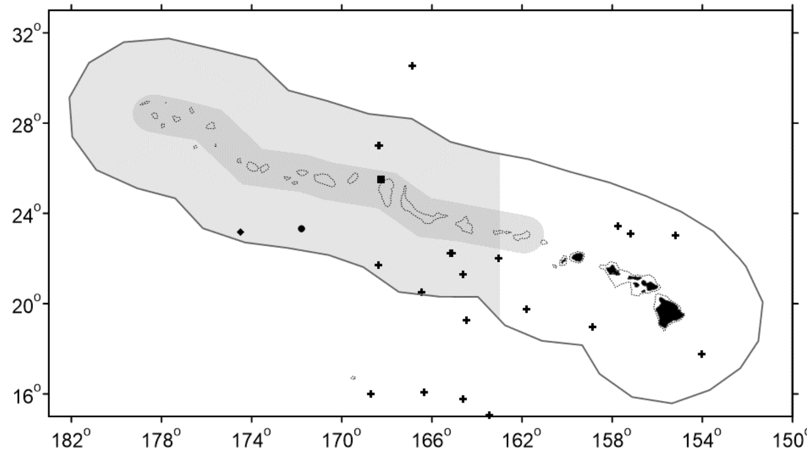


**Figure 1.** Locations of minke whale sightings from longline observer records (crosses; NMFS/PIR, unpublished data), and sighting locations during the 2002 (diamond), 2010 (circle), and 2017 (square) shipboard surveys of U.S. EEZ waters surrounding the Hawaiian Islands (Barlow 2006, Bradford *et al*. 2013, Yano *et al*. 2018). Outer line indicates approximate boundary of survey area and U.S. EEZ. Dark gray shading indicates the original Papahanaumokuakea Marine National Monument, with the lighter gray shading denoting the full 2016 Expansion area. Dotted line represents the 1000 m isobath.

Minke whales occur seasonally around the Hawaiian Islands (Barlow 2003, Rankin and Barlow, 2005), and their migration routes or destinations are unknown. Minke whale "boing" sounds have been detected near the Hawaiian Islands for decades, with detections by the U.S. Navy during February and March (Thompson and Friedl 1982) and at the ALOHA Cabled Observatory 100km north of Oahu from October to May (Oswald *et al*. 2011). Three confirmed sightings of minke whale were made, one in 2002, one in 2010, and one in 2017 during shipboard surveys of waters within the U.S. Exclusive Economic Zone (EEZ) of the Hawaiian Islands (Barlow 2003, Bradford *et al*. 2013, Yano *et al*. 2018).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there are three stocks of minke whale within the Pacific U.S. EEZ: 1) a Hawaiian stock (this report), 2) a California/Oregon/ Washington stock, and 3) an Alaskan stock. The Hawaiian stock includes animals found both within the Hawaiian Islands EEZ and in adjacent high seas waters; however, because data on abundance, distribution, and human-caused impacts are largely lacking for high seas waters, the status of this stock is evaluated based on data from U.S. EEZ waters of the Hawaiian Islands (NMFS 2005).

## POPULATION SIZE

Encounter data from a 2017 summer/fall shipboard line-transect surveys of the entire Hawaiian Islands EEZ were used to estimate the seasonal abundance of minke whales in Hawaiian waters. The design-based abundance estimate for 2017 is 438 (CV = 1.05) animals in the Hawaiian EEZ during the summer/fall (Bradford *et al*. 2021). The abundance estimate used sighting data from throughout the central Pacific to estimate the detection function and Beaufort sea-state-specific trackline detection probabilities for minke whales from Barlow *et al*. (2015). Summer/fall 2002 and 2010 shipboard line-transect surveys of the Hawaiian EEZ each resulted in one 'off effort' sighting of a minke whale (Barlow 2003, Bradford *et al*. 2013). These sightings were not part of regular survey operations and, therefore, could not be used to calculate estimates of abundance. The majority of each of these surveys took place during summer and early fall, when the Hawaiian stock of minke whale would be expected to be farther north. Using passive acoustic detections from an array of seafloor hydrophones north of Kauai, Martin *et al*. (2012) estimate a preliminary average density of 2.15 ”boing” calling minke whales per 1000 km$^2$ during the period February through April and within an area of 8,767 km$^2$ centered on the seafloor array positioned roughly 50km from shore. However, the relationship between the number of whales present and the number of calls detected is not known, and therefore this acoustic method does not provide an estimate of absolute abundance for minke whales.

**Minimum Population Estimate**

The minimum population size is calculated as the lower 20th percentile of the log-normal distribution (Barlow *et al.* 1995) of the 2017 abundance estimate or 212 minke whales.

**Current Population Trend**

No data are available on population size or current population trend.

## CURRENT AND MAXIMUM NET PRODUCTIVITY RATES

No data are available on current or maximum net productivity rate for Hawaiian minke whales.

## POTENTIAL BIOLOGICAL REMOVAL

The potential biological removal (PBR) level for the Hawaii stock of minke whales is calculated as the minimum population estimate (212) <u>times</u> one half the default maximum net growth rate for cetaceans (½ of 4%) <u>times</u> a recovery factor of 0.50 (for a species of unknown status with no estimated fishery mortality or serious injury within the U.S. EEZ of the Hawaiian Islands; Wade and Angliss 1997) resulting in a PBR of 2.1 minke whales per year

## HUMAN CAUSED MORTALITY AND SERIOUS INJURY

**Fishery Information**

Information on fishery-related mortality and serious injury of cetaceans in Hawaiian waters is limited, but the gear types used in Hawaiian fisheries are responsible for marine mammal mortality and serious injury in other fisheries throughout U.S. waters. There are currently two distinct longline fisheries based in Hawaii: a deep-set longline (DSLL) fishery that targets primarily tunas, and a shallow-set longline fishery (SSLL) that targets swordfish. Both fisheries operate within U.S. waters and on the high seas. Between 2014 and 2018, no minke whales were observed hooked or entangled in the SSLL fishery (100% observer coverage) or the DSLL fishery (18-22% observer coverage) (Bradford 2018a, 2018b, 2020, Bradford and Forney 2017).

## STATUS OF STOCK

The Hawaii stock of minke whales is not considered strategic under the 1994 amendments to the MMPA. The status of minke whales in Hawaiian waters relative to OSP is unknown, and there are insufficient data to evaluate trends in abundance. Minke whales are not listed as “threatened” or “endangered” under the Endangered Species Act (1973), nor designated as “depleted” under the MMPA. Because there has been no reported fisheries related mortality or serious injury within the Hawaiian Islands EEZ, the total fishery mortality and serious injury for minke whales can be considered insignificant and approaching zero mortality and serious injury rate. A recent examination of the behavioral response of minke whales to mid-frequency sonar transmissions within the Pacific Missile Range Facility north of Kauai indicated a reduction in minke whale calling during sonar operations (Harris *et al*. 2019). Whether the reduction in calling was the result of displacement or a change in vocal behavior could not be determined with the data available, but does suggest that minke whales are responsive to military sonar activity within their range. The increasing level of anthropogenic sound in the world’s oceans has been suggested to be a habitat concern for whales (Richardson *et al*. 1995). Minke whale increase the source level of their calls during periods of higher ambient noise, though are either unable or unwilling to increase calling levels at the same rate as increases in noise (Helble *et al*. 2020) suggesting masking of calls will occur at high noise levels.

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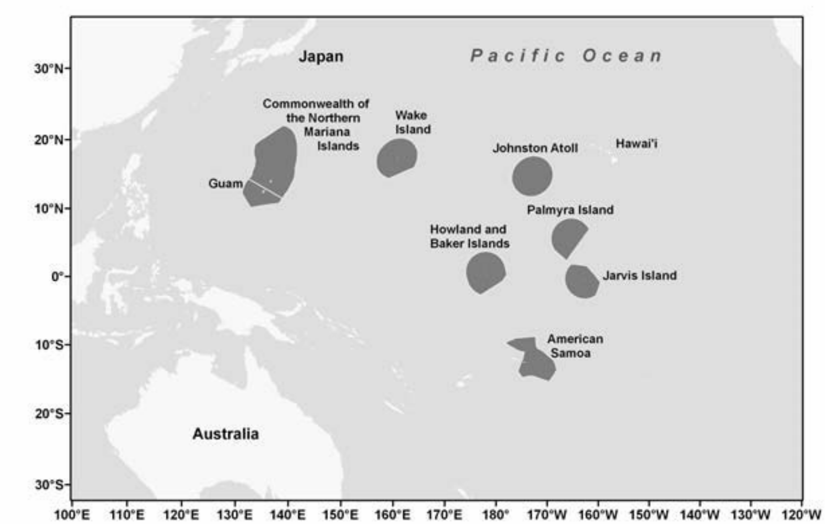
Revised 10/15/2009

# HUMPBACK WHALE (*Megaptera novaeangliae*)
## IUCN Oceania subpopulation – American Samoa Stock

### STOCK DEFINITION AND GEOGRAPHIC RANGE




Samoa stock of humpback whales in this report is derived from survey work conducted within the American Samoa EEZ, although animals range well outside this area (see text).

The humpback whale has a global distribution. Humpback whales migrate long distances between their feeding grounds at mid- to high latitudes and their calving and mating grounds in tropical waters. The Oceania subpopulation (as defined by the IUCN Red List process, see Childerhouse *et al*. 2008) ranges throughout the South Pacific, except the west coast of South America, and from the equator to the edges of the Antarctic ice. Humpback whales have been recorded across most of the lower latitudes of the South Pacific from approximately 30°S northwards to the equator during the austral autumn and winter. Although there have been no comprehensive surveys of this huge area, humpback whale densities are known to vary extensively from high densities in East Australia to low densities at many island groups. Many regional research projects have documented the presence of these whales around various island groups, but they are also found in open water away from islands (SPWRC 2008). Movements of individual whales between the tropical wintering grounds and the Antarctic summer feeding grounds have been documented by a variety of methods including Discovery tagging, photo-identification, matching genotypes from biopsies or carcasses, and satellite telemetry (Mackintosh 1942; Chittleborough 1965; Dawbin 1966; Mikhalev 2000; Rock *et al*. 2006, Franklin *et al*. 2007, Robbins *et al*. 2008). However, migratory routes and specific destinations remain poorly known. Unlike the other humpback stocks found in U. S. waters, the IUCN Oceania subpopulation is defined by structure on its calving grounds (Garrigue *et al*. 2006b, Olavarria *et al*. 2006, 2007) rather than on its feeding grounds. The Oceania subpopulation consists of breeding stocks E (including E1, E2 and E3) and F recognized by the International Whaling Commission (IWC). It is found in the area defined by the following approximate boundaries: 145°E (eastern Australia) in the west, 120°W (between French Polynesia and South America) in the east, the equator in the north, and 30°S in the south (Childerhouse *et al.* 2008).

For the Marine Mammal Protection Act (MMPA) stock assessment reports, there is need for only one South Pacific Island region management stock of humpback whales, the American Samoa stock. American Samoa lies at the boundary of breeding stocks E3 and F. Surveys have been undertaken annually at the primary island of Tutuila since 2003. A total of 150 unique individuals were identified by fluke photographs during 58 days at sea, 2003-2008 (D. Mattila and J. Robbins, unpublished data). Individuals have been resighted on multiple days in a single breeding season, but only three inter-annual re-sightings have been made to date (two based on dorsal fin photographs) (D. Mattila and J. Robbins, unpublished data). Breeding behavior and the presence of very young calves has been documented in American Samoa waters. One whale that was sighted initially without a calf was re-sighted later in the season with a calf. Individual exchange has been documented with Western Samoa (SPWRC 2008), as well as Tonga, French Polynesia and the Cook Islands (Garrigue *et al*. 2007). Although the feeding range of American Samoan whales has not yet been defined, there has been one photo-ID match to the Antarctic Peninsula (IWC Antarctic Area I, Robbins *et al*. 2008). Whales at Tonga have exhibited exchange with both Antarctic Area V (Dawbin 1959) and Area I (Brown 1957, Dawbin 1956) and so whales from American Samoa may have a similarly wide feeding range.

On-going photographic studies indicate a higher frequency of certain types of skin lesions on humpback whales at American Samoa as compared to humpback whale populations at Hawaii or the Gulf of Maine (Mattila and Robbins, 2008). However, the cause and implications have yet to be determined. Some similar skin lesions on blue whales in Chilean waters have been observed (Brownell *et al*. 2008).

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

**Historic whaling**

Southern Hemisphere humpback whales were hunted extensively during the last two centuries, and it is thought that populations have been reduced to a small percentage of their former levels (Chapman 1974). After correcting catch records for illegal Soviet whaling, (Clapham & Baker 2002) estimated that over 200,000 Southern Hemisphere humpback whales were killed from 1904 to 1980. Humpback whales were protected from commercial whaling in 1966 by the IWC but they continued to be killed illegally by the Soviet Union until 1972. Illegal Soviet catches of 25,000 humpback whales in two seasons (1959/60 and 1960/61) precipitated a population crash and the closure of land stations in Australia and New Zealand, including Norfolk Island (Mikhalev 2000; Clapham *et al*. 2005).

**POPULATION SIZE**

There is currently no estimate of abundance for humpback whales in American Samoan waters. The South Pacific Whale Research Consortium produced a number of preliminary mark-recapture estimates of abundance for Oceania and its subregions (SPWRC, 2006). A closed population estimate of 3,827 (CV 0.15) was calculated for eastern Oceania (breeding stocks E3 and F) for 1999-2004 and this may be the most relevant of those currently available, given observed exchange between American Samoa, Tonga, the Cook Islands, and French Polynesia (Garrigue *et al.* 2006a). However, the extent and biological significance of the documented interchange is still poorly understood.

**Minimum Population Estimate**

The minimum population estimate for this stock is 150 whales, which is the number of individual humpbacks identified in the waters around American Samoa between 2003-2008 by fluke photo identification (J. Robbins, personal communication). This is clearly an underestimation of the true minimum population size as photo ID studies have been conducted over a few weeks per year and there is also evidence of exchange with other areas in Oceania. There are also insufficient data to estimate the proportion of time Oceania humpback whales spend in waters of American Samoa.

**Current Population Trend**

No data are available on current population trend.

**CURRENT AND MAXIMUM NET PRODUCTIVITY RATES**

No estimates of current or maximum net productivity rates are available for this species in Samoan waters. However, the maximum plausible growth rate for Southern Hemisphere humpback whale populations is estimated as 10.6% (Clapham *et al*. 2006).

**POTENTIAL BIOLOGICAL REMOVAL**

The potential biological removal (PBR) for this stock is calculated as the minimum population size (150) times one half the estimated maximum growth rate for humpback whales in the Southern Hemisphere (1/2 of 10.6%) times a recovery factor of 0.1 (for an endangered species with a total population size of less than 1,500), resulting in a PBR of 0.8. This stock of humpback whales is migratory and thus, it is reasonable to expect that animals spend at least half the year outside of the relatively small American Samoa EEZ. Therefore, the PBR allocation for U.S. waters is half of 0.8, or 0.4 whales.

**HUMAN-CAUSED MORTALITY AND SERIOUS INJURY**

No human-related mortalities of humpback whales have been recorded in American Samoan waters. Human-related mortality of humpback whales due to entanglements in fishing gear and collisions with ship have been reported elsewhere in the Southern Hemisphere. Entanglement of humpback whales in pot lines has been reported in both New Zealand and Australia but there are no estimated rates available. There is little information from the rest of the South Pacific but a humpback mother (with calf) was reported entangled in a longline in 2007 in the Cook Islands (N. Hauser, reported in SPWRC 2008).

A photographic-based scar study of the humpback whales of American Samoa has been initiated and there is some indication of healed entanglement and ship strike wounds, although perhaps not at the levels found in some Northern Hemisphere populations (D. Mattila and J. Robbins, unpublished data). However, the sample size to date is insufficient for robust comparison and the study is ongoing.

**STATUS OF STOCK**

The status of humpback whales in American Samoan EEZ waters relative to OSP is unknown and there are insufficient data to estimate trends in abundance. However, humpback whale populations throughout the South Pacific were drastically reduced by historical whaling and IUCN classifies the Oceania subpopulation as "Endangered" (Childerhouse *et al.* 2008). Worldwide humpback whales are listed as "endangered" under the Endangered Species Act (1973) so the Samoan stock is automatically considered a "depleted" and "strategic" stock under the MMPA. There are no habitat concerns for the stock.

Japan has proposed killing 50 humpback whales as part of its program of scientific research under special permit (scientific whaling) called JARPA II in the IWC management areas IV and V in the Antarctic (Gales *et al.* 2005). Areas IV and V have demonstrated links with breeding stock E. Japan postponed their proposed catch in the 2007/08 and 2008/09 seasons but have not removed them from their future whaling program. The JARPA II program has the potential to negatively impact the recovery of humpbacks in Oceania.

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**Appendix 1. Fishery Classifications, Fishery Descriptions and Interactions with Marine Mammals.**

This appendix contains fishery information that may be outdated and in need of revision. Readers should refer to the annual NOAA List of Fisheries for the most recent information on commercial fisheries that interact with marine mammals.

The Marine Mammal Protection Act (MMPA) requires NMFS to publish a list of commercial fisheries (List Of Fisheries or "LOF") and classify each fishery based on whether incidental mortality and serious injury of marine mammals is frequent (Category I), occasional (Category II), or unlikely or unknown (Category III). The LOF is published annually in the Federal Register. The categorization of a fishery in the LOF determines whether participants in that fishery are subject to certain provisions of the MMPA, such as registration, observer coverage, and take reduction plan requirements. The categorization criteria as they appear in the LOF are reprinted below:

The fishery classification criteria consist of a two-tiered, stock-specific approach that first addresses the total impact of all fisheries on each marine mammal stock, and then addresses the impact of individual fisheries on each stock. This approach is based on consideration of the rate, in numbers of animals per year, of incidental mortality and serious injury of marine mammals due to commercial fishing operations relative to the Potential Biological Removal (PBR) level for each marine mammal stock. The MMPA (16 U.S.C. 1362 (20)) defines the PBR level as the maximum number of animals, not including natural mortality, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population. This definition can also be found in the implementing regulations for section 118 at 50 CFR 229.2.

Tier 1: If the total annual mortality and serious injury across all fisheries that interact with a stock is less than or equal to 10 percent of the PBR level of the stock, all fisheries interacting with the stock would be placed in Category III. Otherwise, these fisheries are subject to the next tier (Tier 2) of analysis to determine their classification.

Tier 2, Category I: Annual mortality and serious injury of a stock in a given fishery is greater than or equal to 50 percent of the PBR level.

Tier 2, Category II: Annual mortality and serious injury of a stock in a given fishery is greater than 1 percent and less than 50 percent of the PBR level.

Tier 2, Category III: Annual mortality and serious injury of a stock in a given fishery is less than or equal to 1 percent of the PBR level.

While Tier 1 considers the cumulative fishery mortality and serious injury for a particular stock, Tier 2 considers fishery- specific mortality and serious injury for a particular stock. Additional details regarding how the categories were determined are provided in the preamble to the final rule implementing section 118 of the MMPA (60 FR 45086, August 30, 1995). Since fisheries are categorized on a per-stock basis, a fishery may qualify as one Category for one marine mammal stock and another Category for a different marine mammal stock. A fishery is typically categorized on the LOF at its highest level of classification (e.g., a fishery that qualifies for Category III for one marine mammal stock and for Category II for another marine mammal stock will be listed under Category II).

**Other Criteria That May Be Considered**

In the absence of reliable information indicating the frequency of incidental mortality and serious injury of marine mammals by a commercial fishery, NMFS will determine whether the incidental serious injury or mortality qualifies for Category II by evaluating other factors such as fishing techniques, gear used, methods used to deter marine mammals, target species, seasons and areas fished, qualitative data from logbooks or fisher reports, stranding data, and the species and distribution of marine mammals in the area, or at the discretion of the Assistant Administrator for Fisheries (50 CFR 229.2).

This appendix describes commercial fisheries that occur in California, Oregon, Washington, and Hawaiian waters and that interact or may interact with marine mammals. The first three sections describe sources of marine mammal mortality data for these fisheries. The fourth section describes the commercial fisheries for these states. A list of all known fisheries for these states was published as a proposed rule in the Federal Register, 71 FR 20941, 24 April 2006.

**Sources of Mortality/Injury Data**

There are three major sources of marine mammal mortality/injury data for the active commercial fisheries in California, Oregon, Washington, and Hawaii. These sources are the NMFS Observer Programs, the Marine Mammal Authorization Program (MMAP) data, and the NMFS Marine Mammal Stranding Network (MMSN) data. Each of these data sources has a unique objective. Data on marine mammal mortality and injury are reported to the MMAP by fishers in any commercial fisheries. Marine mammal mortality and injury is also monitored by the NMFS Marine Mammal Stranding Network (MMSN). Data provided by the MMSN is not duplicated by either the NMFS Observer Program or MMAP reporting. Human-related data from the MMSN include occurrences of mortality due to entrainment in power station intakes, ship strikes, shooting, evidence of net and line fishery entanglement (net remaining on animal, net marks, severed flukes), and ingestion of hooks.

**Marine Mammal Reporting from Fisheries**

In 1994, the MMPA was amended to implement a long-term regime for managing mammal interactions with commercial fisheries (the Marine Mammal Authorization Program, or MMAP). Logbooks are no longer required - instead vessel owners/operators in any commercial fishery (Category I, II, or III) are required to submit one-page pre-printed reports for all interactions (including those that occur while an observer is onboard) resulting in an injury to or death of a marine mammal. The report must include owner/operator's name and address, vessel name and ID, where and when the interaction occurred, the fishery, species involved, and type of injury (if the animal was released alive). These postage-paid report forms are mailed to all Category I and II fishery participants that have registered with NMFS, and must be completed and returned to NMFS within 48 hours of returning to port for trips in which a marine mammal injury or mortality occurred. The number of self-reported marine mammal interactions is considerably lower than the number reported by fishery observers, even though observer reports are typically based on 20% observer effort. For example, from 2000-2004, there were 112 fisher self-reports of marine mammal interactions in the California swordfish/thresher shark drift gillnet fishery. This compares with 141 observed interactions over the same period, based on only 20% observer coverage. This suggests that fisher self-reports are negatively-biased. From 2007-2011 there were 12 fisher self-reports of marine mammal interactions in the Hawaii-based deep-set longline fishery, 11 of which corresponded to observer records. This compares with 50 observed interactions over the same period, based on 20-22% observer coverage. This suggests fisher self-reports are significantly negatively biased.

**NMFS Marine Mammal Stranding Network data**

From 2000-2004, there were 1,022 cetacean and 13,215 pinniped strandings recorded in California, Oregon, and Washington states. Approximately 10% of all cetacean and 6% of all pinniped strandings showed evidence of human- caused mortality during this period. From 2007-2011, there were 144 cetacean strandings recorded in Hawaii, with 42% of all cetacean strandings showing evidence of human-caused mortality during this period. Human-related causes of mortality include: entrainment in power station intakes, shooting, net fishery entanglement, and hook/line, set-net and trap fishery interaction.

**Fishery Descriptions**

**Category I, CA/OR thresher shark/swordfish drift gillnet fishery (≥14 inch mesh)**

Number of permit holders: The numbers of eligible permit holders in California for, 2008 to 2012 ranged between 78 and 84 (data source: California Department of Fish and Wildlife Permits are non-transferable and are linked to individual fishermen, not vessels.

Number of active permit holders: The numbers of vessels active in this fishery declined from 40 in 2008 to 16 vessels in 2012.

Total effort: Both estimated and observed effort for the drift-net fishery during the calendar years 1990 through 2012 are shown in Figure 2.

Geographic range: Effort in this fishery ranges from the U.S./Mexico border north to waters off the state of Oregon. For this fishery there are area-season closures (see below). Figures 1 shows locations of observed sets for the period 1990 to 2012.

Seasons: This fishery is subject to season-area restrictions. From February 1 to May 15 effort must be further than 200 nautical miles (nmi) from shore; from May 16 to August 14, effort must be further than 75 nmi from shore, and from August 15 to January 31 there is only the 3 nmi off-shore restriction for all gillnets in southern California (see halibut and white seabass fishery below). The majority of the effort occurs from October through December. A season-area closure to protect leatherback sea turtles was implemented in this fishery in August 2001. The closure area prohibits drift gillnet fishing from August 15 through November 15, in the area bounded by straight lines from Point Sur, California (N36$^{o}$ 17') to N 34$^{o}$ 27' W 123$^{o}$ 35', west to W129$^{o}$, north to N 45$^{o}$, then east to the Oregon coast. An additional season-area closure south of Point Conception and east of W120 degrees longitude is effective during the months of June, July, and August during El Niño years to protect loggerhead turtles (Federal Register, 68 FR 69962, 16 December 2003).

Gear type and fishing method: Typical gear used for this fishery is a 1000-fathom gillnet with a stretched mesh size typically ranging from 18-22 inches (14 inch minimum). The net is set at dusk and allowed to drift during the night after which, it is retrieved. The fishing vessel is typically attached to one end of the net. Soak duration is typically 12-14 hours depending on the length of the night. Net extender lengths of a minimum 36 ft. became mandatory for the 1997-1998 fishing season. The use of acoustic warning devices (pingers) became mandatory 28 October 1997.

Regulations: The fishery is managed under a Fishery Management Plan (FMP) developed by the Pacific Fishery Management Council and NMFS.

Management type: The drift-net fishery is a limited-entry fishery with seasonal closures and gear restrictions (see above). The state of Oregon restricts landing to swordfish only.

Comments: This fishery has had a NMFS observer program in place since 1990. Due to bycatch of strategic stocks including short-finned pilot whales, beaked whales, sperm whales and humpback whales, a Take Reduction Team was formed in 1996. Since then, the implementation of increased extender lengths and the deployment of pingers have substantially decreased cetacean entanglement. The fraction of active vessels in this fishery that are not observed owing to a lack of berthing space for observers has been increasing. The fishery currently operates under an emergency rule designed to reduce to the bycatch of sperm whales (Federal Register 4 September 2013, Volume 78: pages 54548-54552.

**Category I, Hawaii deep-set (tuna target) longline/set line fishery**

Note: The Hawaii-based longline fisheries of the Pelagic Fishery Ecosystem Plan (FEP) consist of two separately managed longline fisheries. One is the deep-set (tuna targeted) fishery which is classified as a Category I fishery under the MMPA. This fishery is discussed here. The classification of this fishery was elevated to Category I in 2004 based on revised PBR levels of false killer whales and observed false killer whale mortality in this fishery (Federal Register 69 FR 48407 1, 10 August 2004). The other Hawaii-based longline fishery is the Hawaii shallow-set longline (swordfish targeted) fishery which is classified as a Category II fishery under the MMPA and is discussed in the Category II section of this Appendix.

Number of permit holders: The number of Hawaii longline limited access permit holders is 164. Not all such permits are renewed and used every year. Permit holders may use the permits for either deep-set or shallow-set fishing, but must notify NMFS how they will fish before each trip. Most holders of Hawaii longline limited access permits are based in, or operate out of, Hawaii**.**

Number of active deep-set longline vessels targeting tuna: From 2007 to 2011, the number of active longline vessels based and landing in Hawaii was 129, 127, 127, 122, and 129, respectively.

Total effort: The number of trips ranged from a low of approximately 500 (in 1992) to 1,427 in 2007. Figure 4 shows the number of fishing trips by longline vessels based and landing in Hawaii, by year and trip type, 1991-

2009. The number of sets for the deep-set tuna fishery in 2007-2011 was 17,885, 16,810, 16,070, and 17,155. The number of hooks set in 2007- 2011 was 38.8 million, 40.1 million, 37.7 million, 37.1 million, and 40.7 million.

Geographic range: The Hawaii-based pelagic, deep-set longline fishery operates inside and outside the EEZ, primarily around the main Hawaiian Islands and Northwestern Hawaiian Islands, with some trips to the EEZs around the remote U.S. Pacific islands (however there are restricted areas, please refer to "Regulations"). Vessels vary their fishing grounds depending on their target species. Most of the deep-set fishing occurs south of 25° N.

Seasons: This fishery operates year-round, although vessel activity increases during the fall and is greatest during the winter and spring months.

Gear type: Deep-set longline gear typically consists of a continuous main line set on the surface and supported in the water column horizontally by floats with branch lines connected at intervals to the main line. In addition radio buoys are also used to keep track of the mainline as it drifts at sea. A line shooter is used on deep-sets to deploy the mainline faster than the speed of the vessel, thus allowing the longline gear to sink to its target depth (average target depth is 167 m, target depth for bigeye tuna is approximately 400 m). The main line is typically 30 to 100 km (18 to 60 nm) long. A minimum of 15, but typically 20 to 30, weighted branch lines (gangions) are clipped to the mainline at regular intervals between the floats. Each gangion terminates with a single baited hook. The branch lines are typically 11 to 15 meters (35 to 50 feet) long. Sanma (saury) or sardines are used for bait. Lightsticks are not typically attached to the gangions on this type of longline set. Deep-set longline gear is set in the morning and hauled in the evening and at night.

Regulations: This fishery is managed under the Pelagics FEP and subject to Federal regulation. Measures that are currently applicable to the fishery include, but are not limited to, limited access (requirement for a permit), vessel and gear marking requirements, vessel length restrictions, Federal catch and effort logbooks, large longline restricted areas around the Hawaiian Archipelago, vessel monitoring system (VMS), annual protected species workshops, use of circle hooks with wire diameter not greater than 4.5mm and branch line not less than 2.0mm, and the use of sea turtle, seabird, and marine mammal handling and mitigation gear and techniques. The vessel operator must notify NMFS prior to departure whether the vessel is undertaking a deep-set or shallow-set trip. Once the trip type is set, it cannot be changed during the trip. Vessel operators must take a NMFS contracted observer if requested by NMFS – target observer coverage is 20 percent of trips. If any marine mammal interaction (hooking or entanglement) resulting in injury or mortality occurs, the vessel operator must complete and mail a pre-addressed, postage paid form to NOAA Fisheries within 48 hours of the end of the trip. Additional information on all applicable regulations for the deep-set longline fishery is available at NOAA. This fishery is subject to the False Killer Whale Take-Reduction Team. NMFS is currently implementing the Take-Reduction Plan and associated regulations.

Management type: Federal limited access program. This fishery is managed under a Fishery Ecosystem Plan (FEP) developed by the Western Pacific Fishery Management Council and NMFS.

Comments: Non-target species are caught incidentally. Interactions with common bottlenose dolphins, false killer whales, humpback whales, short-finned pilot whales, pantropical spotted dolphins, Blainville's beaked whale, sperm whales, striped dolphins and Risso's dolphins have been documented. Due to interactions with protected species, especially turtles, this fishery has been observed since February 24, 1994. Initially, observer coverage was less than 5%, increased to 10% in 2000, and equaled or exceeded 20% since 2001. *Observed* marine mammal deaths and injuries form 2007-2011 included 24 false killer whales, 4 short-finned pilot whales, 3 Risso's dolphins, 2 common bottlenose dolphins, 1 sperm whale, 1 pantropical spotted dolphin, one striped dolphin, and 14 unidentified cetaceans. Four of the interactions were deaths, 32 were serious injuries, nine were non-serious injuries, one involved prorating a large whale interaction as 0.75 serious (NMFS, 2012), and four were classified as cannot-be-determined.

**Category II , CA halibut/white seabass and other species set gillnet fishery (>3.5 inch mesh).**

Note: Halibut are typically targeted using 8.5 inch mesh while the remainder of the fishery targets white seabass and yellowtail using 6.5 inch mesh. In recent years, there has been an increasing number of 6.0-6.5 inch mesh sets fished using drifting methods; this component is now identified as a separate fishery (see "**CA yellowtail, barracuda, white seabass, and tuna drift gillnet fishery (>3.5 and <14 in mesh)**" fishery described below).

Number of permit holders: There is no specific permit category for this fishery. Overall, the current number of legal permit holders for gill and trammel nets, excluding swordfish drift gillnets and herring gillnets were between 141 and 154 annually. Information on permit numbers is available from the California Department of Fish and Wildlife website.

Number of active permit holders: Approximately 50 vessels participate in this fishery (NMFS List of Fisheries, Federal Register 29 August 2013).

Total effort: Total fishing effort for the period 2008 to 2012 has been approximately 2,000 sets annually.

Geographic range: Effort in this fishery previously ranged from the U.S./Mexico border north to Monterey Bay and was localized in more productive areas: San Ysidro, San Diego, Oceanside, Newport, San Pedro, Ventura, Santa Barbara, Morro Bay, and Monterey Bay. Fishery effort is now predominantly in the Ventura Flats area off of Ventura, the San Pedro area between Pt. Vicente and Santa Catalina Island and in the Monterey Bay area. The central California portion of the fishery from Point Arguello to Point Reyes has been closed since September 2002 when a ban on gillnets inshore of 60 fathoms took effect.

Seasons: This fishery operates year round. Effort generally increases during the summer months and declines during the last three months of a year.

Gear type and fishing method: Typical gear used for this fishery is a 200 fathom gillnet with a stretched mesh size of 8.5 inches. The component of this fishery that targets white seabass and yellowtail utilizes 6.5 inch mesh. The net is generally set during the day and allowed to soak for up to 2 days. Soak duration is typically 8-10, 19-24, or 44-49 hours. The depth of water ranges from 15-50 fathoms with most sets in water depths of 15-35 fathoms.

Regulations: This fishery is managed by the California Department of Fish and Game in accordance with state and federal laws.

Management type: The halibut and white seabass set-net fishery is a limited-entry fishery with gear restrictions and area closures.

Comments: An observer program for the halibut and white seabass portion of this fishery operated from 1990-94 and was discontinued after area closures were implemented in 1994, which prohibited gillnets within 3 nmi of the mainland and within 1 nmi of the Channel Islands in southern California. NMFS re-established an observer program for this fishery in Monterey Bay in 1999-2000 due to a suspected increase in harbor porpoise mortality in Monterey Bay. In 1999 and 2000, fishery mortality exceeded PBR for the Monterey Bay harbor porpoise stock, which at that time, was designated as strategic [the stock is currently non-strategic]. In the autumn of 2000, the California Department of Fish and Game implemented the first in a series of emergency area closures to set gillnets within 60 fathoms along the central California coast in response to mortality of common murres and threats to sea otters. This effectively reduced fishing effort to negligible levels in 2001 and 2002 in Monterey Bay. A ban on gill and trammel nets inside of 60 fathoms from Point Reyes to Point Arguello became effective in September 2002. Bycatch of marine mammals, including California sea lions and harbor seals, continues in this fishery, based on limited observer data.

**Category II, Hawaii shallow-set (swordfish target) longline/set line fishery**

Note: The Hawaii-based longline fisheries of the Pelagic Fishery Ecosystem Plan (FEP) consist of two separately managed longline fisheries. One is the deep-set (tuna targeted) fishery which is classified as a Category I fishery under the MMPA. The other is the Hawaii shallow-set longline (swordfish targeted) fishery which is classified as a Category II fishery under the MMPA and is discussed here.

Number of permit holders: The number of Hawaii longline limited access permit holders is 164. Not all such permits are renewed and used every year. Permit holders may use the permits for either deep-set or shallow-set fishing, but must notify NMFS how they will fish before each trip. Most holders of Hawaii longline limited access permits are based in, or operate out of, Hawaii**.** Longline general permits are not limited by number. These general permits are open access and usable in Guam, CNMI, and the Pacific Remote Island Areas; they are usually not more than a half dozen a year.

Number of active shallow-set longline vessels targeting swordfish: From 2007 to 2011, the number of active shallow-set longline vessels based in and landing in Hawaii was 28, 27, 28, 28, and 20.

Total effort: The number of trips since 1991 has ranged from zero (2002-2003) to approximately 300 in 1993. Figure 4 shows the number of fishing trips by longline vessels based and landing in Hawaii, by year and trip type, 1991-2011. The number of sets for the shallow-set swordfish fishery in 2007-2011 was 1,570, 1,597, 1,762, 1,833, and 1,468. The number of hooks set in 2007-2011 was 1.4 million, and 1.5 million, 1.7 million, 1.8 million, 1.5 million.

Geographic range: The most productive swordfishing areas for Hawaii-based longline vessels are north of Hawaii outside the U.S. Exclusive Economic Zone (EEZ) on the high seas, and this fishery operates almost entirely north of Hawaii (north of approximately 20° N). In some years, when influenced by seawater temperature, this fishery may operate mostly north of 30° N.

Seasons: Shallow-set effort is highest in either the first or second quarter of the calendar year and drops off substantially in the latter half of the year.

Gear type: Shallow-set longline gear typically consists of a continuous main line set on the surface and supported in the water column horizontally by floats with branch lines connected at intervals to the main line. In addition radio buoys are also used to keep track of the mainline as it drifts at sea. Longline fishing for swordfish is known as shallow-set longline fishing as the bait is set at depths of 30–90 m. The portion of the mainline with branchlines attached is suspended between floats at about 20–75 m of depth, and the branchlines hang off the mainline another 10–15 m. Only 4-6 branchlines are clipped to the mainline between floats, and a typical set for swordfish uses about 1,000-1,200 hooks. Shallow-set longline gear is set at night, with luminescent light sticks attached to the branchlines. Formerly, J-hooks and squid bait were used, but since 2004, circle hooks and mackerel-type bait have been required. These gear restrictions were implemented to reduce sea turtle bycatch.

Regulations: This fishery is managed under the Pelagics FEP and subject to Federal regulation. Measures that are currently applicable to the fishery include, but are not limited to, limited access (requirement for a permit), vessel and gear marking requirements, vessel length restrictions, Federal catch and effort logbooks, 100-percent observer coverage, large longline restricted areas around the Hawaiian Archipelago, vessel monitoring system (VMS), annual protected species workshops, and the use of sea turtle, seabird, and marine mammal handling and mitigation gear and techniques. The vessel operator must notify NMFS prior to departure whether the vessel is undertaking a shallow-set or a deep-set trip. Once the trip type is set, the type cannot be changed during the trip. All shallow-set trips must have a NMFS contracted observer. If any marine mammal interaction (hooking or entanglement) resulting in injury or mortality occurs, the vessel operator must complete and mail a pre-addressed, postage paid form to NOAA Fisheries within 48 hours of the end of the trip. More information on all applicable regulations is available at NOAA. This fishery is subject to the False Killer Whale Take-Reduction Team. NMFS is currently implementing the Take-Reduction Plan and associated regulations.

Management type: Federal limited access program. This fishery is managed under a Fishery Ecosystem Plan (FEP) by the Western Pacific Fishery Management Council and NMFS.

Comments: Non-target species are caught incidentally. Interactions with common bottlenose dolphins, false killer whales, humpback whales, short-finned pilot whales, striped dolphins, Bryde's whales, Risso's dolphins, sperm whales, spinner dolphins, pygmy sperm or dwarf sperm whales, Blainville's beaked whales, and common dolphins have been documented. The shallow-set fishery was completely closed in 2001 and reopened in 2004. One hundred percent observer coverage is required in this fishery. *Observed* injuries of marine mammals in this fishery in 2007-2011 included 3 false killer whales, 21 Risso's dolphins, 2 humpback whale, 1 pygmy or dwarf sperm whale, 3 striped dolphins, 8 common bottlenose dolphins, 1 short-beaked common dolphin, 1 Blainville's beaked whale, 2 unidentified beaked whales, and 2 unidentified dolphins. Three of the interactions were deaths, 31 were serious injuries, 10 were non-serious injuries, and 2 involved prorating a large whale interaction as 0.75 serious.

**Category II, Hawaii shortline fishery**

Note: The Hawaii shortline fishery was added to the 2010 List of Fisheries as a Category II fishery under the MMPA based on analogy with the Category I "HI deep-set (tuna-target) longline/set line" and Category II "HI shallow-set (swordfish- target) longline/set line" fisheries (Federal Register 74 FR 58859, 16 November 2009).

Number of permit holders: There are no specific fishing permits issued for this fishery. However, all persons with a State of Hawaii Commercial Marine License (CML) may participate in any fishery, including the "HI shortline" fishery.

Number of active shortline vessels: Of those persons possessing CMLs, shortline participation has varied between 5 and 14 vessels from 2003 - 2011.

Total effort: From 2003-2008, there was an average of 135,757 pounds (lbs) of fish landed each year. In 2008 alone, 104,152 lbs of fish were landed.

Geographic range: The Category II "HI shortline" fishery is a small-scale system operating off the State of HI, and targeting bigeye tuna (*Thunnus obesus*) or the lustrous pomfret (*Eumigistes illustris*). This fishery was developed to target these fish species when they concentrate over the summit of Cross Seamount, 290 km (180 mi) south of the State of HI.

Seasons: This fishery has no seasonal component and may operate year-round.

Gear type: The gear style is designed specifically to target the aggregating fish species over seamount structures. The primary gear type used is a horizontal main line (monofilament) less than 1 nautical mile long, and includes two baskets of approximately 50 hooks each. The gear is set before dawn and has a short soak time, with the gear retrieved about two hours after it is set.

Regulations: All persons with a State of Hawaii Commercial Marine License (CML) may participate in the "HI shortline" fishery. The mainline length must be less than 1 nautical mile.

Management type: Hawaii State managed fishery.

Comments: Currently, there is no Federal reporting system in place to document potential marine mammal interactions in this fishery. However, there are anecdotal reports of interactions off the north side of Maui, but the species and extent of interactions are unknown.

**Category II, American Samoa longline fishery**

Note: The American Samoa longline fishery was added to the 2006 List of Fisheries as a Category II fishery under the MMPA based on analogy with Category I "HI deep-set (tuna-target) longline/set line" and Category II "HI shallow-set (swordfish-target) longline/set line" fisheries.

Number of permit holders: 46

Number of active longline vessels: From 2007 to 2011, the number of active vessels was 29, 28, 26, 26, and 24.

Total effort: The number of trips for 2007-2011 was 377, 287, 175, 264, and 274. The number of sets for the American Samoa longline fishery in 2007-2011 was 5,910, 4,730, 4,601, 4,496, and 3,776. The number of hooks set in 2007-2011
was 17,524, 14,372, 14,207, 13,067, and 10,767.

Geographic range: Waters surrounding American Samoa year-round.

Seasons: Shallow-set effort is highest in either the first or second quarter of the calendar year and drops off substantially in the latter half of the year.

Gear type: This fishery uses longline gear. Vessels over 50 ft (15.2 m) may set 1,500-2,500 hooks and have a greater fishing range and capacity for storing fish (8-40 metric tons). The fleet reached a peak of 66 vessels in 2001, and set a peak of almost 7,000 sets in 2002. It is more common for fishermen to set their gear in the day and haul in the afternoon, mainly to improve their catch rates.

Regulations: This fishery is a limited entry fishery for pelagic longline vessels in the U.S. EEZ around American Samoa. In 2000, the fishery began to expand rapidly with the influx of large (more than 50 ft (15.2m m) overall length) conventional mono hull vessels, similar to the type used in the Hawaii-based longline fisheries. Regulations implemented in 2002 prohibit any large U.S. vessels (50 ft (15.2 m) and longer) from fishing within 50 nmi around the islands of American Samoa. In 2005, the rapid expansion of longline fishing effort within the U.S. EEZ waters around American Samoa prompted the implementation of a limited entry system. Under the limited access program, NMFS issued a total of 60 initial longline limited entry permits in 2005 to qualified candidates, spread among 4 vessel size classes: 22 permits issued in Class A (less than or equal to 40 ft (12.2 m) length); 5 in Class B (40-50 ft (12.2-15.2m)); 12 in Class C (50.1–70 ft (15.2–21.3 m)); and 21 in Class D (more than 70 ft (21.3 m)). The number of active vessels has shifted to large vessels (Class C and D), with only a couple of small vessels active in the past two years. Permits may be transferred and renewed. Under the limited entry program, vessel operators must submit federal catch and effort logbooks, vessels over 40 ft (12.2 m) must carry observers if requested by NMFS, and vessels over 50 ft (15.2 m) must have an operational vessel monitoring system (VMS). In addition, vessel owners and operators must attend a protected species workshop annually, carry and use dip nets, clippers, and bolt cutters, and follow handling, resuscitation, and release requirements for incidentally hooked or entangled sea turtles.

Management type: Federal limited access program. This fishery is managed under a Fishery Ecosystem Plan (FEP) by the Western Pacific Fishery Management Council and NMFS.

Comments: Non-target species are caught incidentally. Interactions with false killer whales, Risso's dolphins, and Cuvier's beaked whale have been documented. One hundred percent observer coverage is required in this fishery. *Observed* injuries of marine mammals in this fishery in 2007-2011 included 3 false killer whales, 21 Risso's dolphins, 2 humpback whale, 1 pygmy or dwarf sperm whale, 3 striped dolphins, 8 common bottlenose dolphins, 1 short-beaked common dolphin, 1 Blainville's beaked whale, 2 unidentified beaked whales, and 2 unidentified dolphins. Three of the interactions were deaths, 31 were serious injuries, 10 were non-serious injuries, and 2 involved prorating a large whale interaction as 0.75 serious.

**Category II, CA yellowtail, barracuda, white seabass, and tuna drift gillnet fishery (>3.5 and <14 in mesh)**

Number of permit holders: There are approximately 24 active permit holders in this fishery.

Total effort: From 2008 to 2012, there were between 207 and 271 small-mesh drift gillnet sets fished annually, as determined from California Department of Fish and Game logbook data.

Geographic range: This drift gillnet component of this fishery operates primarily south of Point Conception. Observed sets have been clustered around Santa Cruz Island, the east Santa Barbara Channel, and Cortez and Tanner Banks. Some effort has also been observed around San Clemente Island and San Nicolas Island.

Seasons: This fishery operates year round. Targeted species is typically determined by market demand on a short-term basis.

Gear type and fishing method: Typical gear used for this fishery is a 150 to 200-fathom gillnet, which is allowed to drift. The mesh size depends on the target species but typical values observed are 6.0 and 6.5 inches.

Regulations: This fishery is managed by the California Dept. of Fish and Game in accordance with State and Federal laws.

Management type: This fishery is a limited-entry fishery with gear restrictions and area closures.

Comments: This fishery primarily targets white seabass and yellowtail but also targets barracuda and albacore tuna. From 2002-2004, there have been 63 sets observed from 17 vessel trips. Marine mammal mortality includes two long- beaked common dolphin and 3 California sea lions. Also, 4 California sea lions were entangled and released alive during this period. In 2003, there was one coastal bottlenose dolphin stranded with 3.5-inch gillnet wrapped around its tailstock, the responsible fishery is unknown. Observer coverage in this fishery was 12% in 2002, 10% in 2003, and 17% in 2004.

**Category II, California anchovy, mackerel, and sardine purse seine fishery**

Number of permit holders: There are 63 limited-entry permits (Pacific Fishery Management Council. 2005. Status of the Pacific Coast coastal pelagic species fishery and recommended acceptable biological catches. Stock Assessment and Fishery Evaluation Report 2005).

Number of active permit holders: There are 61 vessels actively fishing.

Total effort: The fishery is managed under a capacity goal, with gross tonnage of vessels used as a proxy for fishing capacity. Capacity for the fleet is approximately 5,400 gross tons. Harvest guidelines for sardine and mackerel are also set annually.

Geographic range: These fisheries occur along the coast of California predominantly from San Pedro, including the Channel Islands, north to San Francisco.

Seasons: This fishery operates year round. Targeted species vary seasonally with availability and market demand.

Gear type and fishing method: Purse seine, drum seine and lampara nets utilizing standard seining techniques.

Regulations: This is a limited-entry fishery.

Management type: The fishery is managed under a Coastal Pelagic Species Fisheries Management Plan developed by the Pacific Fishery Management Council and NMFS.

Comments: A NMFS pilot observer program began in July 2004 and continued through January 2006. A total of 93 sets have been observed. Observed marine mammal interactions with the fishery have included one California sea lion killed, 54 sea lions released alive, and one sea otter released alive. Under the MMAP self-reporting program, the following mortality was reported: In 2003, four California sea lions drowned after chewing through a bait barge net used by the anchovy lampara net fishery.

**Category III, California tuna purse seine fishery**

Note: This fishery was previously included in the CA anchovy, mackerel, and sardine purse seine fishery (see above). Vessels in the anchovy, mackerel, and sardine fishery target tuna when oceanographic conditions result in an influx of tuna into southern California waters. Data for this fishery were obtained from the 'Status of the U.S. West Coast Fisheries for Highly Migratory Species through 2004', available at the Pacific Fishery Management Council website.

Number of permit holders: There are 63 limited-entry permits (Pacific Fishery Management Council. 2005. Status of the Pacific Coast coastal pelagic species fishery and recommended acceptable biological catches. Stock Assessment and Fishery Evaluation Report 2005).

Number of active permit holders: Between one and 23 vessels actively purse seined for tunas during the period 2000-2004.

Total effort: The number of vessels landing bluefin, yellowfin, skipjack, and albacore in 2000-2004 varied between one and 23. Logbooks are not required for this fishery, and the overall number of sets fished is unknown.

Geographic range: Observed sets in this fishery have occurred in the southern California Bight.

Seasons: Observed sets occurred in August and September. The timing of fishing effort varies with the availability of tuna species in this region.

Gear type and fishing method: Small coastal purse seine vessels with a <640 mt carrying capacity target bluefin, yellowfin, albacore and skipjack tuna during warm-water periods in southern California.

Regulations: This is a limited-entry fishery.

Management type: This fishery is managed under a Highly Migratory Species Management Plan developed by the Pacific Fishery Management Council and NMFS.

Comments: A pilot observer program for this fishery began in July 2004 and ended in January 2006. A total of 9 trips and 15 sets were observed with no marine mammal interactions.

**Category II, WA Puget Sound Region salmon drift gillnet fishery**

Number of permit holders: This commercial fishery includes all inland waters south of the US-Canada border and east of the Bonilla/Tatoosh line, at the entrance to the Strait of Juan de Fuca. Treaty Indian salmon gillnet fishing is not included in this commercial fishery. The number of permit holders is reported to be 210 in the NMFS 2013 List of Fisheries (Federal Register 29 August 2013).

Number of active permit holders: The number of "active" permits is assumed to be equal to or less than the number of permits that are eligible to fish.

Total effort: Effort in the Puget Sound salmon drift gillnet fishery is regulated by systematic openings and closures that are specific to area and target salmon species.

Geographic Range: The fishery occurs in the inland marine waters south of the U.S./Canada border and east of the Bonilla/Tatoosh line at the entrance to the Strait of Juan de Fuca. The inland waters are divided into smaller statistical catch areas which are regulated independently.

Seasons: This fishery has multiple seasons throughout the year that vary among local areas dependent on local salmon runs. The seasons are managed to access harvestable surplus of robust stocks of salmon while minimizing impacts on weak stocks.

Gear type and fishing methods: Vessels operating in this fishery use a drift gillnet of single web construction, not exceeding 300 fathoms in length. Minimum mesh size for gillnet gear varies by target species. Fishing directed at sockeye and pink salmon are limited to gillnet gear with a 5-inch minimum mesh and a 6 inch maximum, with an additional "bird mesh" requirement that the first 20 meshes below the corkline be constructed of 5-inch opaque white mesh for visibility; the chinook season has a 7-inch minimum mesh; the coho season has a 5-inch minimum mesh; and the chum season has a 6- to 6.25-inch minimum mesh. The depth of gillnets can vary depending upon the fishery and the area fished. Normally they range from 180 to 220 meshes in depth, with 180 meshes as a common depth. It is the intention of the fisher to keep the net off the bottom. The vessel is attached to one end of the net and drifts with the net. The entire net is periodically retrieved onto the vessel and catch is removed. Drift times vary depending on fishing area, tidal condition and catch.

Regulations: The fishery is a limited-entry fishery with seasonal openings, area closures, and gear restrictions.

Management type: The fishery occurs in State waters and is managed by the Washington Department of Fish and Wildlife consistent with the U.S.-Canada Pacific Salmon Commission management regimes and the ocean salmon management objectives of the Pacific Fishery Management Council. U.S. and Canadian Fraser River sockeye and pink salmon fisheries are managed by the bilateral Fraser Panel in Panel Area waters. This includes the entire U.S. drift gillnet fishery for Fraser sockeye and pink salmon. For U.S. fisheries, Fraser Panel Orders are given effect by federal regulations that consist of In-season Orders issued by the NMFS Regional Administrator of the NMFS Northwest Region. These regulations are filed in the Federal Register post-season.

Comments: Salmon drift gillnet fisheries in Washington inland waters were last observed in 1993 and 1994, with observer coverage levels typically 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 (NMFS NW Region, unpublished data). Past marine mammal entanglements in this fishery included harbor porpoise, Dall's porpoise, and harbor seals.

**Category III, CA squid purse seine fishery**

Number of Permit Holders: A permit has been required to participate in the squid fishery since April 1998. Originally, only two types of permits were issued, either a vessel or light boat permit during the moratorium period from 1998 to 2004. Since the adoption of the Market Squid Fishery Management Plan (MSFMP) in 2005, a total of seven different permit types are now allowed under the restricted access program. Permit types include both transferable and non-transferable vessel, brail and light boat permits whose qualifying criteria are based on historical participation in the fishery during the moratorium period. Market squid vessel and brail permits allow a vessel to use lights to attract and capture squid using either purse seines or brail gear. Light boat owner permits only allow the use of attracting lights to attract and aggregate squid. In addition, three experimental non-transferable permits are allowed for vessel fishing outside of historical fishing areas north of San Francisco. In the 2006/2007 season there were 91 vessel permits, 14 brail permits, 64 light boat permits and 3 experimental permits issued. A permit is not required when fishing for live bait or when landing two short tons or less, which is considered incidental.

Number of Active Permit Holders: The number of active permits varies by year depending on market conditions and availability of squid. During the 2006/2007 season (1 April 2006 – 31 March 2007) there were approximately 84 vessels active during some portion of the year. Twenty-nine vessels harvested 86% of the total landings greater than two tons. The 1999/2000 season had the highest squid landings to date (115,437mt), with 132 vessels making squid landings.

Total Effort: Logbooks have been mandatory for the squid fishery since May 2000. Results for the 2006 calendar year indicate that each hour of fishing required 1.4 hours of search time by light boats. Combined searching and fishing effort resulted in 6.9 metric tons (mt) of catch per hour. In the 2006/2007 season, the fishery made 1,611 landings. This is a 47% decrease from the previous season. In addition, the average landing decreased from 23.9 mt to 21.7 mt.

Geographic Range: Since the 1960's there have been two distinct fisheries in operation north and south of Point Conception. Since the mid-1980's the majority of the squid fishing harvest has occurred in the southern fishery, with efforts focused around the Channel Islands and along the mainland from Port Hueneme to La Jolla. In the 2006/2007 season, the southern fishery landed 98% of the catch with the majority of landings occurring around the northern Channel Islands. In contrast, during the 2005/2006 season, landings in the southern fishery were primarily around Catalina Island. The northern fishery, centered primarily in Monterey Bay, has been in operation since the mid-1860's and has historical significance to California. During the 2002/2003 season, a moderate El Niño condition resulted in nearly 60% of the catch being landed in northern California.

Seasons: The fishery can occur year-round; however, fishing efforts differ north and south of Point Conception. Typically, the northern fishery operates from April through September while the southern fishery is most active from October through March. El Niño conditions generally hamper the fishery in the southern fishery and squid landings are minimal during these events. In contrast, landings in the northern fishery often increase during El Niño events and then are depressed for several years after.

Gear Type: There are several gears employed in this fishery. From 1996 to 2006, the vast majority (95%) of vessels use either purse (69%) or drum (26%) seine nets. Other types of nets used include brail (5%) and lampara nets (<1%). Another gear type associated with the fishery is attracting lights (30,000 watts maximum) that are used to attract and aggregate spawning squid in shallow waters.

Regulations: Since March 2005, the fishery operates under a restricted access program that requires all vessels to be permitted. A mandatory logbook program for fishing and lighting vessels has been in place since May 2000. A monitoring program has been in place since 2000 that samples the landings is designed to evaluate the impact of the fishery on the resource. Attracting lights were regulated with each vessel restricted to no more than 30,000 watts of light during fishing activities. These lights must also be shielded and oriented directly downward to reduce light scatter. The lighting restrictions were enacted to avoid risks to nesting brown pelicans and interactions with other seabird species of concern. A seabird closure area restricting the use of attracting lights for commercial purposes in any waters of the Gulf of the Farallones National Marine Sanctuary was enacted. A seasonal catch limitation of 107,047 mt (118,000 short tons) was established to limit further expansion of the fishery. Commercial squid fishing is prohibited between noon on Friday and noon on Sunday of each week to allow an uninterrupted consecutive two-day period of spawning. Additional closure areas to the fishery to protect squid spawning habitat include the Channel Islands Marine Protected Areas (MPAs) and the newly established MPAs along the central California coast as well as areas closed to the use of purse seine gear including the leeward side of Catalina Island, Carmel and Santa Monica Bays.

Management Type: The market squid fishery is under California State management. The fishery was largely unregulated until 1998 when it came under regulatory control of the California Fish and Game Commission and the Department of Fish and Game. The MSFMP was enacted on March 28, 2005. The MSFMP was developed to ensure sustainable long-term conservation and to be responsive to environmental and socioeconomic changes. Market squid is also considered a monitored species under the Pacific Fishery Management Council's (PFMC) Coastal Pelagic Species Fishery Management Plan.

Comments: During the 1980's, California's squid fishery grew rapidly in fleet size and landings when international demand for squid increased due to declining fisheries in other parts of the world. In 1997 industry-sponsored legislation halted the growth of fleet size with a moratorium on new permits. Landing records were set several times during the 1990's, but landings seem to fluctuate with changing environmental and atmospheric conditions of the California Current. Encounters with marine mammals and sea birds are documented in logbooks. Seal bombs are used regularly, but fishermen report that they no longer have an effect. A pilot observer program began in July 2004 and has documented one unidentified common dolphin death in 135 sets through January 2006. In addition, there have been 96 California sea lions and three harbor seals released alive (NMFS, Southwest Region, unpublished data). In addition to the observed death, there were three strandings of Risso's dolphin from 2002-2003 where evidence of gunshot wounds was confirmed, suggesting interaction with this fishery (NMFS Southwest Regional Office, unpublished data). The squid fishery operates primarily at night and targets spawning aggregations of adult squid. In recent years the amount of daylight fishing has increased, especially in Monterey, in part due to better sonar gear, but also to reduce interactions with California sea lions. The PFMC adopted the egg escapement method to monitor the impact of market squid fishery since no reliable biomass estimate has been developed. It is a proxy for Maximum Sustainable Yield (MSY), setting an egg escapement threshold level at which to evaluate the magnitude of fishing mortality on the spawning potential of the squid stock. The egg escapement method was developed on conventional spawning biomass "per-recruit" theory. In general, the MSY Control Rule for market squid is based on evaluating levels of egg escapement associated with the exploited population. The egg escapement threshold, initially set at 30%, represents a biological reference point from which to evaluate fishery related impacts.

**Category II, CA Dungeness crab pot fishery**

Notes: For all commercial pot and trap fisheries in California, a general trap permit is required, in addition to any specific permits required for an individual fishery. All traps are required to be tended and serviced at least every 96 hours, weather permitting.

Number of permit holders: The Dungeness crab fishery is a limited access fishery requiring a vessel-based permit that is transferable. This program was initiated in 1994 based on landing histories. The number of vessels participating on an annual basis does vary, but approximately 400 vessels have been landing crab in recent years.

Number of active permit holders: Approximately 400 vessels have been landing crabs in recent years.

Total effort: There is no restriction on the number of traps that may be fished at one time by a single vessel. Some vessels use as many as 1000 or more traps at the peak of the season (December/January).

Geographic range: This fishery operates in central and northern California.

Seasons: The fishery is divided into two management areas. The central region (south of the Mendocino-Sonoma county line) fishery opens November 15 and continues through June 30. The northern region (north of the Mendocino-Sonoma county line) is annually scheduled to open on December 1, but may be delayed by CDF&G based on the condition of market size crabs, and continues until July 15.

Gear type: For each trap fished there is one vertical line in the water, though only in the northern region, is fishing strings illegal. All traps are required to be marked with buoys bearing the commercial fishing license number. The normal operating depth for Dungeness crab is between 35 and 70 m. Traps are typically tended on a daily basis.

Regulations: There is no daily logbook requirement for the commercial Dungeness crab fishery. There is a recreation fishery for Dungeness crab, which allows for 10 crab per day to be harvested except when fishing on a commercial passenger fishing vessel (CPFV) in central California, the limit is 6 crab per person. There is no reliable estimate for the effort or landings in the sport fishery except that CPFVs are required to track catch and effort by species.

Management type: The Dungeness crab pot fishery is managed by the California legislature, CDF&G and also by the tri- state committee for Dungeness, which includes the states of Oregon and Washington.

Comments: Humpback whale entanglements with Dungeness crab gear have not been confirmed, but are suspected as the responsible fishery based on the location and timing of fishing effort and observed humpback entanglements.

**Category II, OR Dungeness crab pot fishery**

Notes: Dungeness crab is the most significant pot/trap fishery in the state of Oregon. Over the long term, the fishery has averaged around 10 million lb of landings per year; although since 2003, annual landings have been approximately 25 to 30 million lb. This fishery requires an Oregon issued limited-entry permit, which is transferable.

Number of permit holders: There were 433 permit holders in 2006.

Number of active permit holders: A total of 364 vessels landed crabs in 2006.

Total effort: In 2006, the fishery made a transition to a three-tiered pot limitation program which allows a maximum of 200, 300, or 500 pots to be fished at any one time depending on previous landing history. The pot limitation is implemented through a buoy tag requirement. All Dungeness crab pots require buoy tags with the identifying associated permit attached. The expected result of the buoy tags and tier limits is to reduce the number of pots in Oregon waters down from 200,000 to approximately 150,000.

Geographic range: Oregon waters.

Seasons: The Dungeness crab season runs from December 1 to August 14. The highest landings are always recorded in December through February, at the beginning of the season.

Gear type: Pots.

Regulations: All Oregon pot/trap gear must be marked on its terminal ends with pole and flag, light, radar reflector, and buoy with the owner/operator number clearly marked. By law, gear may not be left unattended for more than seven days. All vessel operators and deck hands must have a commercial fishing license or crewmembers license.

Management type: State management, Oregon Department of Fish and Wildlife.

Comments: Humpback whale entanglements with Dungeness crab gear have not been confirmed, but are suspected as the responsible fishery based on the location and timing of fishing effort and observed humpback entanglements.

**Category II, CA spot prawn fishery**

Number of permit holders: A three-tiered limited access permit system is used in this fishery to accommodate changes in the fishery that occurred when trawling methods were banned and replaced with trap fishing in 2003. Permits are linked to the vessel owner and only Tier 1 permits are transferable. Tier 1 permits allow a maximum of 500 traps in use at a time. Eighteen vessels had Tier 1 permits in 2007. Tier 2 permits allow 150 traps in use at a time. There were three vessels utilizing Tier 2 permits in 2007. Tier 3 permits were issued to allow vessels that previously used trawl gear to switch to trap gear to target spot prawn. There were nine Tier 3 permits issued in 2007. Information on 2007 license statistics was obtained from the CA Department of Fish and Game.

Number of active permit holders: A total of 30 vessels participated in this fishery in 2007.

Total effort: Landings have increased every year since 2003. The total number of traps set is unknown, although the theoretical maximum number of traps that may be fished annually is approximately 13,000.

Geographic range: The fishery operates from Monterey south. Over half of the landings are made in Los Angeles and San Diego. Traps are typically set in waters of 182 m (100 fathoms) or more. South of Point Arguello, traps must be fished in waters 91 m (50 fathoms) or deeper.

Seasons: North of Point Arguello, the fishery is open from February 1 to October 30. North of Point Arguello, the open season is August 1 to April 30.

Gear type: Strings of 25 to 50 traps are fished in deep waters (>182 m).

Regulations: For all commercial pot and trap fisheries in California, a general trap permit is required, in addition to any specific permits required for an individual fishery. All traps are required to be tended and serviced at least every 96 hours, weather permitting. There is a daily logbook requirement in this fishery. There is no buoy marking requirement and no recreational fishery for this species.

Management type: This fishery is managed under state authority by the California Department of Fish and Game.

Comments: One humpback whale was seriously injured in 2006 as a result of entanglement in spot prawn trap gear.

**Category II, WA/OR/CA sablefish pot fishery**

Notes: Sablefish is likely the most commonly targeted groundfish caught in pot gear in off the U.S. west coast.

Number of permit holders: There are 32 limited-entry permits (LEPs) to catch sablefish with pot gear. Open access privileges are also available to fishermen.

Number of active permit holders: Including all vessels which made landings with an LEP or under open access rules, a total of about 150 vessels participated in this fishery in 2007. This total fluctuates on an annual basis.

Total effort: Estimated annual landings indicate usually over 1 million lbs of sablefish are landed per year in this fishery.

Geographic range: The fishery is well distributed from central California north to the U.S./Canadian border. Most of the effort occurs out in deeper waters (200-400 m).

Seasons: Most fishing effort occurs January through September.

Gear type: Traps <6 ft. in any dimension.

Regulations: A general trap permit is all that is required for open access to this fishery by the states along the U.S. west coast. LEPs are divided into a three-tiered system which allocates annual landing limits to individual permits based on the status of the stock. Daily logbook reporting is required.

Management type: Sablefish is managed under the federal Groundfish Fishery Management Plan. This is the only trap fishery regulated by the federal government; all others are managed by the states.

Comments: One humpback whale was seriously injured in 2006 as a result of entanglement in sablefish trap gear.

**Category III, CA coonstripe shrimp, rock crab, tanner crab pot or trap fishery**

Number of permit holders: There were 134 permits issued in 2007.

Number of active permit holders: Unknown, but it is likely that most issued permits are active.

Total effort: Annual landings averaged approximately 1 million pounds from 2000 to 2005.

Geographic range: The fishery operates throughout California waters. Most landings are made south of Morro Bay, California, with approximately 65% of all landings coming from the Santa Barbara area.

Seasons: There are no seasonal restrictions, though some area closures exist.

Gear type: There is no restriction on the number of traps that may be fished at one time by the vessel but the typical number of traps operated at any given time is less than 200. Traps are usually buoyed singularly or in pairs, but fishing strings (multiple traps attached together between two buoys) is allowed. Buoys are required to be marked with the license number of the operator. The normal working depth of traps in this fishery is 10 to 35 fathoms.

Regulations: There is no daily logbook requirement for the commercial rock crab fishery.

Management type: The fishery is managed by the California Department of Fish and Game.

Comments: The recreational bag limit is 35 crabs per day, but there is no reliable estimate of the effort or landings in the sport fishery.

**Category III, CA halibut bottom trawl fishery**

Notes: This is a newly-listed fishery in the 2007 MMPA NMFS List of Fisheries (Federal Register Volume 72, No. 59, 14466). Information on fishing effort was provided by Stephen Wertz, California Department of Fish and Game.

Number of permit holders: There were 60 permits issued in 2006.

Number of active permit holders: There were 31 active permit holders in 2006.

Total effort: Thirty one vessels made 3,711 tows statewide in 2006, totaling 3,897 tow hours, in 332 days of fishing effort.

Geographic range: The fishery operates from Bodega Bay in northern California to San Diego in southern California, from 3 to 200 nautical miles offshore. Trawling is prohibited in state waters (0 to 3 nmi offshore) and within the entire Monterey Bay, except in the designated “California halibut trawl grounds”, between Point Arguello and Point Mugu beyond 1 nautical mile from shore. Trawls used in this region must have a minimum mesh size of 7.5 in and trawling is prohibited here between 15 March and 15 June to protect spawning adults.

Seasons: Fishing is permitted year-round, except in state waters. State waters are closed between 15 March and 15 June.

Gear type: Otter trawls, with a minimum mesh size of 4.5 inches are required in federal waters, while fishing in state waters has a 7.5 inch mesh size requirement.

Regulations: Fishing in state waters is limited to the period 14 March – 16 June in the 'California halibut trawl grounds' in southern California between Point Arguello and Point Mugu. All other fishing must occur in federal waters beyond 3 nautical miles from shore.

Management type: The fishery is managed by the California Department of Fish and Game.

Comments: No marine mammal interactions have been documented for this fishery, but the gear type and fishing methods are similar to the WA/OR/CA groundfish trawl fishery (also category III), which is known to interact with marine mammals.

**Category III, CA herring gillnet fishery**

The herring fishery is concentrated in four spawning areas which are managed separately by the California Department of Fish and Game (CDFG); catch quotas are based on population estimates derived from acoustic and spawning-ground surveys. The largest spawning aggregations occur in San Francisco Bay and produces more than 90% of the herring catch. Smaller spawning aggregations are fished in Tomales Bay, Humboldt Bay, and Crescent City Harbor. During the early 1990's, there were 26 round haul permits (either purse seine or lampara nets). Between 1993 and 1998, all purse seine fishers converted their gear to gillnets with stretched mesh size less than 2.5 inches (which are not known to take mammals) as part of CDFG efforts to protect herring resources. The fishery is managed through a limited-entry program. The California Department of Fish and Game website lists a total of 447 herring gillnet permits for 2005. Of these, 406 permits exist for San Francisco Bay, 34 in Tomales Bay, 4 in Humboldt Bay, and 3 in Crescent City Harbor. This fishery begins in December (San Francisco Bay) or January (northern California) and ends when the quotas have been reached, but no later than mid-March.

**Category III, WA Willapa Bay drift gillnet fishery**

Number of permit holders: The total number of permit holders for this fishery in 1995 and 1996 was 300, but this number has declined in subsequent years. In 1997 there were 264 total permits and 243 in 1998. The NMFS 2013 List of Fisheries lists an estimate of 82 vessels/persons in this fishery.

Number of active permit holders: The number of active permit holders is assumed to be equal to or less than the number of permits eligible to fish in a given year. The number of permits renewed and eligible to fish in 1996 was 300 but declined to 224 in 1997 and 196 permits were renewed for 1998. The 1996-98 counts do not include permits held on waivers for those years, but do include permits that were eligible to fish at some point during the year and subsequently entered into a buyback program. The number of permits issued for this fishery has been reduced through a combination of State and federal permit buyback programs. Vessels permitted to fish in the Willapa Bay are also permitted to fish in the lower Columbia River drift gillnet fishery.

Total effort: Effort in this fishery is regulated through area and species openings. The fishery was observed in 1992 and 1993 when fishery opening were greater than in recent years. In 1992 and 1993 there were 42 and 19 days of open fishing time during the summer "dip-in" fishery. The "dip-in" fishery was closed in 1994 through 1999. Available openings have also declined in the fall chinook/coho fisheries. In 1992/93 respectively there were 44 and 78 days of available fishing time. There were 43, 45, 22 and 16.5 available open fishing days during 1995 through 1998.

Geographic range: This fishery includes all inland marine waters of Willapa Bay. The waters of the Bay are further divided into smaller statistical catch areas.

Seasons: Seasonal openings coincide with local salmon run timing and fish abundance.

Gear type: Fishing gear used in this fishery is a drift gillnet of single web construction, not exceeding 250 fathoms in length, with a minimum stretched mesh size ranging upward from 5 inches depending on target salmon species. The gear is commonly set during periods of low and high slack tides. It is the intention of the fisher to keep the net off the bottom. The vessel is attached to one end of the net and drifts with the net. The entire net is periodically retrieved onto the vessel and catch is removed. Drift times vary depending on fishing area, tidal condition, and catch.

Regulations: This fishery is a limited-entry fishery with seasonal openings and gear restrictions.

Management type: The salmon drift gillnet fishery is managed by the Washington Department of Fish and Wildlife.

Comments: Observers were placed onboard vessels in this fishery to monitor marine mammal interactions in the early 1980s and in 1990-93. Five incidentally taken harbor seals were recovered by observers in the fishery from 1991through 1993 (3 in '92 and 2 in '93). Two incidentally taken northern elephant seals were recovered by observers from the fishery in 1991 but no takes of this species were observed. The summer fishery (July-August) in Willapa Bay has been closed since it was last observed in 1993 and available fishing time declined from 1996 through 1998.

**Category III, WA Grays Harbor salmon drift gillnet fishery**

Number of permit holders: This commercial drift gillnet fishery does not include Treaty Indian salmon gillnet fishing. The total number of permit holders for this commercial fishery in 1995 and 1996 was 117 but this number has declined in subsequent years. In 1997 there were 101 total permits and 87 in 1998.

Number of active permit holders: The NMFS 2013 List of Fisheries lists a total of 24 vessels/persons operating in this fishery. The number of active permit holders is assumed to be equal to or less than the number of permits eligible to fish in a given year. The number of permits renewed and eligible to fish in 1996 was 117 but declined to 79 in 1997 and 59 permits were renewed for 1998. The 1996-98 counts do not include permits held on waivers for those years but do include permits that were eligible to fish at some point during the year and subsequently entered a buyback program. The number of permits issued for this fishery has been reduced through a combination of State and federal permit buyback programs. Vessels permitted to fish in Grays Harbor are also permitted to fish in the lower Columbia River salmon drift gillnet fishery.

Total effort: Effort in this fishery is regulated through area and species openings. The fishery was observed in 1992 and 1993 when fishery openings were greater than in recent years. In 1992 and 1993 there were 42 and 19 days of open fishing time during the summer "dip-in" fishery. The "dip-in" fishery was closed in 1994 through 1999. Available openings have also declined in the fall chinook/coho fisheries. There were 11, 17.5, 9 and 5 available open fishing days during the 1995 through 1998 fall season.

Geographic range: Effort in this fishery includes all marine waters of Grays Harbor. The waters are further divided into smaller statistical catch areas.

Seasons: This fishery is subject to seasonal openings which coincide with local salmon run timing and fish abundance.

Gear type: Fishing gear used in this fishery is a drift gillnet of single web construction, not exceeding 250 fathoms in length, with a minimum stretched mesh size ranging of 5 inches depending on target salmon species. The gear is commonly set during periods of low and high slack tides and retrieved periodically by the tending vessel. It is the intention of the fisher to keep the net off the bottom. The vessel is attached to one end of the net and drifts with the net. The entire net is periodically retrieved onto the vessel and catch is removed. Drift times vary depending on fishing area, tidal condition, and catch.

Regulations: The fishery is a limited-entry fishery with seasonal openings and gear restrictions.

Management type: The salmon drift gillnet fishery is managed by the Washington Department of Fish and Wildlife.

Comments: Observers were placed onboard vessels in this fishery to monitor marine mammal interactions in the early 1980s and in 1990-93. Incidental take of harbor seals was observed during the fishery in 1992 and 1993. In 1992, one harbor seal was observed entangled dead during the summer fishery and one additional seal was observed entangled during the fall fishery but it escaped uninjured. In 1993, one harbor seal was observed entangled dead and one additional seal was recovered by observers during the summer fishery. The summer fishery (July-August) in Grays Harbor has been closed since it was last observed in 1993. Available fishing time in the fall chinook fisheries declined from 1996 through 1998.

**Category III, WA, OR lower Columbia River (includes tributaries) drift gillnet fishery**

Number of permit holders: The total number of permit holders was 856 (344 from Oregon and 512 from Washington) when the fishery was last observed in 1993. In 1995 through 1998 the number of permits was 747, 693, 675 and 620 respectively. The number of permits issued for this fishery by Washington has been reduced through a combination of State and federal buy-back programs. This reduction is reflected in the overall decline in the total number of permits.

Number of active permit holders: In 1995, 110 vessels (of the 747 vessels holding permits) landed fish in the mainstem fishery.

Total effort: Effort in this fishery is regulated through species related seasonal openings and gear restrictions. The fishery was observed in 1991, 1992 and 1993 during several seasons of the year. The winter seasons (openings) for 1991 through 1993 totaled 13, 9.5, and 6 days respectively. The winter season has subsequently been reduced to remnant levels to protect upriver ESA listed salmon stocks. In 1995 there was no winter salmon season, in 1996 the fishery was open for 1 day. In 1997 and 1998 the season was shifted to earlier in the year and gear restrictions were imposed to target primarily sturgeon. The fall fishery in the mainstem was also observed 1992 and 1993 as was the Young's Bay terminal fishery in 1993, however, no marine mammal mortality was observed in these fisheries. The fall mainstem fishery openings varied from 1 day in 1995 to just under 19.5 days in 1997 and 6 days in 1998. The fall Youngs Bay terminal fishery fluctuated between 60 and 70 days for the 1995 through 1998 period which was similar to the fishery during the period observed.

Geographic range: This fishery occurs in the main stem of the Columbia river from the mouth at the Pacific Ocean upstream to river mile 140 near the Bonneville Dam. The lower Columbia is further subdivided into smaller statistical catch areas which can be regulated independently.

Seasons: This fishery is subject to season and statistical area openings which are designed to coincide with run timing of harvestable salmon runs while protecting weak salmon stocks and those listed under the Endangered Species Act. In recent years, early spring (winter) fisheries have been sharply curtailed for the protection of listed salmon species. In 1994, for example, the spring fishery was open for only three days with approximately 1900 fish landed. In 1995 the spring fishery was closed and in 1996 the fishery was open for one day but fishing effort was minimal owing to severe flooding. Only 100 fish were landed during the one day in 1996.

Gear type: Typical gear used in this fishery is a gillnet of single web construction, not exceeding 250 fathoms in length, with a minimum stretched mesh size ranging upwards from 5 inches depending on target salmon species. The gear is commonly set during periods of low and high slack tides. It is the intention of the fisher to keep the net off the bottom. The vessel is attached to one end of the net and drifts with the net. The entire net is periodically retrieved onto the vessel and catch is removed. Drift times vary depending on fishing area, tidal condition, and catch.

Regulations: The fishery is a limited-entry fishery with seasonal openings, area closures, and gear restrictions.

Management type: The lower Columbia River salmon drift gillnet fishery is managed jointly by the Washington Department of Fish and Wildlife and the Oregon Department of Fish and Wildlife.

Comments: Observers were placed onboard vessels in this fishery to monitor marine mammal interactions in the early 1980s and in 1990-93. Incidental takes of harbor seals and California sea lions were documented, but only during the winter seasons (which have been reduced dramatically in recent years to protect ESA-listed salmon). No mortality was observed during the fall fisheries.

**Category III, WA, OR salmon net pens**

Number of permit holders: There were 12 commercial salmon net pen (“grow out”) facilities licensed in Washington in 1998. There are no commercial salmon net pen or aquaculture facilities currently licensed in Oregon. Non-commercial salmon enhancement pens are not included in the list of commercial fisheries.

Number of active permit holders: Twelve salmon net pen facilities in Washington.

Total effort: The 12 licensed facilities on Washington operate year-round.

Geographic range: In Washington, net pens are found in protected waters in the Straits (Port Angeles), northern Puget Sound (in the San Juan Island area) as well as in Puget Sound south of Admiralty Inlet. There are currently no commercial salmon pens in Oregon.

Seasons: Salmon net pens operate year-round.

Gear type: Net pens are large net impoundments suspended below a floating dock-like structure. The floating docks are anchored to the bottom and may also support guard (predator) net systems. Multiple pens are commonly rafted together and the entire facility is positioned in an area with adequate tidal flow to maintain water quality.

Regulations: Specific regulations unknown.

Management type: In Washington, the salmon net pen fishery is managed by the Washington Department of Natural Resources through Aquatic Lands Permits as well as the Washington Department of Fish and Wildlife.

Comments: Salmon net pen operations have not been monitored by NMFS for marine mammal interactions, however, incidental takes of California sea lions and harbor seals have been reported.

**Category III, WA, OR, CA groundfish trawl fishery**

Number of permit holders: In 1998, approximately 332 vessels used bottom and mid-water trawl gear to harvest Pacific coast groundfish. This is down from 383 vessels in 1995. The NMFS List of Fisheries for 2013 lists 160-180 vessels as participating in this fishery. Groundfish trawl vessels harvest a variety of species including Pacific hake, flatfish, sablefish, lingcod, and rockfish. This commercial fishery does not include Treaty Indian fishing for groundfish.

Note: All observed incidental marine mammal takes have occurred in the mid-water trawl fishery for Pacific hake. The annual hake allocation is divided between vessels that harvest and process catch at sea and those that harvest and deliver catch to shore-based processing facilities. At least one NMFS-trained observer is placed on board each at-sea processing vessel to provide comprehensive data on total catch, including marine mammal takes. In the California, Oregon, and Washington range of the fishery, the number of vessels fishing ranged between 12 and 16 (all with observers) during 1997-2001. Hake vessels that deliver to shore-based processors are issued Exempted Fishing Permits that requires the entire catch to be delivered unsorted to processing facilities where State technicians have the opportunity to sample. In 1998, 13% of the hake deliveries landed at shore-based processors were monitored. The following is a description of the commercial hake fishery.

Number of active permit holders: A license limitation ("limited-entry") program has been in effect in the Pacific coast groundfish fishery since 1994. The number of limited-entry permits is limited to 404. Non-tribal trawl vessels that harvest groundfish are required to possess a limited-entry permit to operate in the fishery. Any vessel with a federal limited-entry trawl permit may fish for hake, but the number of vessels that do is smaller than the number of permits. In 1998, approximately 61 limited-entry vessels, 7 catcher/processors and 50 catcher vessels delivering to shoreside and mothership processors, made commercial landings of hake during the regular season. In addition, 6 unpermitted mothership processors received unsorted hake catch.

Total effort: The hake allocation continues to be fully utilized. From 1997 to 1999 the annual allocation was 232,000 mt/year, this is an increase over the 1996 allocation of 212,000 mt and the 1995 allocation of 178,400 mt. In 1998, motherships vessels received 50,087 mt of hake in 17 days, catcher/processors took 70,365 mt of hake in 54 days and shore-based processors received 87,862 mt of hake over a 196 day period.

Geographic range: The fishery extends from northern California (about $40^{o}$ 30' N. latitude) to the U.S.-Canada border. Pacific hake migrate from south to north during the fishing season, so effort in the south usually occurs earlier than in the north.

Seasons: From 1997 to 1999, season start dates have remained unchanged. The shore-based season in most of the Eureka area (between $42^{O}$- $40^{O}$30' N latitude) began on April 1, the fishery south of $40^{O}$30' N latitude opened April 15, and the fishery north of $42^{O}$ N latitude started on June 15. In 1998, the primary season for the shore-based fleet closed on October 13, 1998. The primary seasons for the mothership and catcher/processor sectors began May 15, north of $42^{0}$ N. lat. In 1998, the mothership fishery closed on May 31, the catcher/processor fishery closed on August 7.

Gear type: The Pacific hake trawl fishery is conducted with mid-water trawl gear with a minimum mesh size of 3 inches throughout the net.

Regulations/Management type: This fishery is managed through Federal regulations by the Pacific Fishery Management Council under the Groundfish Fishery Management Plan.

Comments: Since 1991, incidental takes of Steller sea lions, Pacific white-sided dolphins, Dall's porpoise, California sea lions, harbor seals, northern fur seals, and northern elephant seals have been documented in the hake fishery. From 1997- 2001, 4 California sea lions, 2 harbor seals, 2 northern elephant seals, 1 Pacific white-sided dolphin, and 6 Dall’s porpoise were reported taken in California/Oregon/Washington regions by this fishery.

**Category III, Hawaii inshore gillnet fishery**

Note: Category III fisheries in Hawaii are managed primarily by the State of Hawaii. Some fisheries have undergone many changes in geographic and temporal extent in recent years and complete analyses of fishing effort for recent years are not yet available. For many, fishing season and specific gear types are not well defined. These fishery descriptions will be updated as new information and analyses become available.

Number of active permit holders: In 2011 there were 36 active commercial fishers. In 1995 there were approximately 115.

Total effort: In 2011 there were 495 trips. This fishery operates in nearshore and coastal pelagic regions.

Seasons: This fishery operates year-round with the exception of juvenile big-eyed scad less than 8.5 inches which cannot be taken from July through October.

Gear type: Gillnets are of stretched mesh greater than 2 inches and stretched mesh size greater than 2.75 inches for stationary gillnets. The net dimensions may not exceed 7 feet high and 125 feet long.

Regulations: Stationary nets must be inspected every 2 hours and total soak time cannot exceed four hours in the same location. New restrictions implemented in 2007 include that nets may not: 1) be used more than once in a 24-hour period;
2) exceed a 7 ft stretched height limit; 3) exceed a single-panel; 4) be used at night; 5) be set within 250 ft. of another lay net; 6) be set in more than 80 ft depths; 7) be left unattended for more than ½ hour; 8) break coral

during retrieval, 9) be set in freshwater streams or stream mouths, and nets must be 1) registered with the Division of Aquatic Resources; 2) inspected within two hours after being set; 2) tagged with two marker buoys while fished. Gillnets are prohibited around all of Maui and portions of Oahu and Hawaii Island.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Federal Fishery

Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

Comments: The principle catches include reef fishes and big-eyed scad (akule) and mackerel scad (opelu). Interactions have been documented with bottlenose dolphins and spinner dolphins.

**Category III, Hawaii opelu/akule net fishery**

Number of active permit holders: In 2011 there were 22 active commercial fishers.

Total effort: In 2011 there were 843 trips.

Seasons: unknown.

Gear type: Fishing with a net that captures fish by raising the net from beneath a school of fish. Normally fish are encouraged over and into the new with chum.

Regulations: unknown.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Federal Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Hawaii inshore purse seine fishery**

Number of active permit holders: In 2011 there were less than 3 active commercial fishers.

Total effort: Cannot be reported to protect confidentiality.

Seasons: Year round.

Gear type: Fishing with a net that is used to surround a school of fish and is closed by drawing the bottom of the net together to form a bag.

Regulations: It is unlawful for any person without a valid commercial marine license to take akule with any net that has less than 2-3/4" stretched mesh. It is unlawful to take akule less than 8.5 inches with net from July – October or possess or sell more than 200 lbs of akule less than 8.5 inches per day during July – October. Federal regulations governing this gear can be found in the Code of Federal Regulations, Title 50, Part 665, Subpart C.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. This fishery is also managed under the Federal Hawaiian Archipelago Fishery Ecosystem Plan in waters outside of 3 nmi from shore.

**Category III, Hawaii throw net/ cast net fishery**

Number of active permit holders: In 2011 there were 29 active commercial fishers.

Total effort: In 2011 there were 445 fishing trips.

Seasons: unknown.

Gear type: Fishing with a round or conical shaped net with a weighted outer perimeter that is thrown over fish.

Regulations: Minimum size 2 inch stretched mesh. Possession of thrownets with mesh size less than 2 inches in or near the water where fish may be taken is unlawful. Nets with smaller mesh may be used to take shrimp (`opae), `opelu, and makiawa.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Federal Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

Comments: Targets inshore and reef fish.

**Category III, Hawaii hukilau fishery**

Number of active permit holders: In 2011 there were 26 active commercial fishers.

Total effort: In 2011 there were 227 fishing trips.

Seasons: unknown.

Gear type: Includes hukilau, beach seine, dragnet, pen, surround, etc. Fishing with a net by moving it through the water to surround fish by corralling and trapping them within the walls of the net.

Regulations: Outside of 3nmi from shore, the Federal Fishery Ecosystem Plan for the Hawaii Archipelago requires seine nets be attended to at all times. Federal regulations governing this gear can be found in the Code of Federal Regulations, Title 50, Part 665, Subpart C.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. This fishery is also managed under the Federal Hawaiian Archipelago Fishery Ecosystem Plan outside of 3 nmi from shore.

Comments: Typical species: usually inshore and reef fish.

**Category III, Hawaii trolling, rod, and reel fishery**

Number of active permit holders: In 2011 there were 2,126 active commercial fishers.

Total effort: In 2011 there were 30,020 fishing trips.

Seasons: Year round.

Gear type: Fishing by towing or dragging line(s) with artificial lure(s), dead or live bait, or green stick and dnaglers using a sail, surf or motor-powered vessel underway. Up to six lines rigged with artificial lures may be trolled when outrigger poles are used to keep gear from tangling. When using live bait, trollers move at slower speeds to permit the bait to swim naturally. Pelagic trollers generally fish at an average distance of 5 to 8 miles from shore, with a maximum distance of about 30 miles from shore. Trollers fish where water masses converge and where submarine cliffs, seamounts, and other underwater features dramatically change the bathymetry. Trolls often fish drifting logs, other flotsam, underneath bird aggregations, and near FADs. Typical target species include mahimahi, ono, billfishes (marlin, sailfishes, etc.), kaku, uluas, kamanu, tunas, etc.

Regulations: The Fishery Ecosystem Plan for Pelagic Fisheries of the Western Pacific contains no management regulation applicable to pelagic trolling in Federal waters around Hawaii.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. This fishery is also managed under the Federal Pacific Pelagics Fishery Ecosystem Plan outside of 3 nmi from shore.

**Category III, Hawaii kaka line fishery**

Number of active permit holders: In 2011 there were 17 active commercial fishers.

Total effort: In 2011 there were 46 fishing trips.

Seasons: unknown.

Gear type: Fishing with a gear consisting of a mainline less than one nautical mile in length to which are attached multiple branchlines with baited hooks. Mainline is set horizontally, and fixed on or near the bottom, or in shallow midwater. Typical target species varies spending on set location, e.g., nearshore or pelagics.

Regulations: Managed under State of Hawaii regulations.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Hawaii vertical longline fishery**

Number of active permit holders: In 2011 there were 9 active commercial fishers.

Total effort: In 2011 there were 92 fishing trips.

Seasons: unknown.

Gear type: Fishing using a vertical mainline, less than one nautical mile in length and suspended from the surface with float, from which leaders with baited hooks are attached and ending with a terminal weight.

Regulations: unknown.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Hawaii crab trap fishery.**

Number of active permit holders: In 2011 there were 9 license holders fishing crab traps.

Total effort: In 2011 there were 168 crab traps trips.

Seasons: unknown.

Gear type: Fishing with any of various fishing devices made into the shape of a box, container, or enclosure, with one or more openings that allow marine life to get inside but keep them from leaving.

Regulations: Minimum mesh size: Netting - stretched mesh 2 inches; Rigid material - 2 inches by 1 inch. Entrance cones for traps have no minimum mesh size. Traps must be portable and not exceed 10 feet in length or 6 feet in height or width.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

Comments: From 2007-2011, five humpback whales were reported as entangled in Hawaii trap gear (Lyman 2013, NMFS unpublished data). The gear involved in two entanglements was identified as crab trap gear, one was identified as possibly crab trap gear, and the remaining two could not be identified to a specific trap fishery (NMFS unpublished data). Pre- mitigation injury determinations for the crab trap and possible crab trap entanglements were two serious injuries and one prorated as 0.75 serious injury (Bradford and Lyman 2013, NMFS unpublished data). Humpback serious injury and mortality in the crab trap fishery from 2007-2011 is 2.75, with a 5-year annual average of 0.55 per year.

**Category III, Hawaii fish trap fishery**

Number of active permit holders: In 2011 there were 9 active commercial fishers.

Total effort: In 2011, there were 125 fish trap trips.

Seasons: unknown.

Gear type: Fishing with any of various fishing devices made into the shape of a box, container, or enclosure, with one or more openings that allow marine life to get inside but keep them from leaving.

Regulations: Minimum mesh size: Netting - stretched mesh 2 inches; Rigid material - 2 inches by 1 inch. Entrance cones for traps have no minimum mesh size. Traps must be portable and not exceed 10 feet in length or 6 feet in height or width.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Main Hawaiian Islands lobster trap fishery**

Number of active permit holders: In 2011 there were less than 3 active commercial fishers.

Total effort: Cannot be reported to protect confidentiality.

Geographic range: Lobster fishing is prohibited within the NWHI.

Seasons: In the MHI, open season is from September through April.

Gear type: One string consists of approximately 100-fathom-plus plastic lobster traps. About 10 such strings are pulled and set each day. Since 1987 escape vents that allow small lobsters to escape from the trap have been mandatory. In 1996, the fishery became “retain all”, i.e. there are no size limits or prohibitions on the retention of berried female lobsters. The entry-way of the lobster trap must be less than 6.5 inches to prevent monk seals from getting their heads stuck in the trap. In the MHI, rigid trap materials must have a dimension greater than 1 inch by 2 inches, with the trap not exceeding 10 feet by six feet.

Regulations: The MHI fishery is managed by the State of Hawaii, Division of Aquatic Resources with season and gear restrictions (see above).

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Hawaii shrimp trap fishery**

Number of active permit holders: In 2011 there were 4 active commercial fishers

Total effort: In 2011 there were 69 shrimp trap trips.

Seasons: unknown.

Gear type: Fishing with any of various fishing devices made into the shape of a box, container, or enclosure, with one or more openings that allow marine life to enter but not exit.

Regulations: State regulations specify a minimum mesh size for traps: netting must be a minimum of 2 inches stretched mesh, and rigid material must be a minimum of 2 inches by 1 inch. Entrance cones for traps have no minimum mesh size. Traps must be portable and not exceed 10 feet in length or 6 feet in height or width.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear. *Heterocarpus* shrimp are a federally managed complex caught by traps and are subject to annually set Annual Catch Limits.

**Category III, Hawaii crab net fishery**

Number of active permit holders: In 2011 there were 6 active commercial fishers

Total effort: In 2011 there were 61 crab net trips.

Seasons: unknown.

Gear type: Fishing normally with a small circular lift net that is used to catch crabs. Ring nets set manually from the shoreline, mainly in estuarine areas. The nets are used singly, and are not connected with a ground line. Gear is typically tended.

Regulations: Managed under State of Hawaii regulations.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, Hawaii Kona crab net fishery.**

Number of active permit holders: In 2011 there were 48 active permits.

Total effort: In 2011 there were 179 Kona crab trips.

Seasons: Closed during breeding season May-August

Regulations: Only male crabs of at least 4 inches carapace length may be retained.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

**Category III, aku boat- pole and line fishery**

Number of active permit holders: In 2011 there were 3 active commercial fishers

Total effort: In 2011 there were 86 aku boat trips.

Seasons: unknown.

Gear type: Fishing for aku (skipjack tuna) using live bait (such as nehu or iao) and or artificial lures. Generally live bait and/or water is flung or sprayed out from the stern of the (often drifting) vessel to "chum up the school" and get them feeding. Fishers on the stern of the boat often jig and slap the water with their poles to increase surface feeding behavior. Fish are hooked with pole and line, using a barbless hook (feathered, baited or not).

Regulations: Managed under State of Hawaii regulations. Specific licenses administered by DAR for the taking of baitfish and nehu (Hawaiian anchovy) for baiting purposes may be required. No baitfish may be sold or transferred except for bait purposes and licensees must furnish monthly baitfish catch reports to the DAR.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. This fishery is also managed under the Federal Pacific Pelagics Fishery Ecosystem Plan outside of 3 nmi from shore.

**Category III, Hawaii Main Hawaiian Islands deep sea bottomfish handline fishery**

Note: The Hawaii bottomfish complex is a U.S. fishery management unit comprised primarily of several species of snappers and jacks and a grouper inhabiting waters of the Hawaiian Archipelago. The federal fisheries management regime includes three fishing zones: the main Hawaiian Islands (MHI) Zone, and two zones in the Northwestern Hawaiian Islands, the Mau Zone and the Hoomalu Zone. All bottomfish fishing currently takes place in the MHI zone due to the closure of the Northwestern Hawaiian Islands under Presidential Proclamation 8031. The main Hawaiian Islands bottomfish fishery is managed jointly by NMFS and the State of Hawaii.

Number of permit holders: In 2010 there were 569 active commercial fishers.

Total effort: From 2008 to 2010 in the MHI the reported average annual catch was 221,500 lbs., with an additional 44,300 to 553,700 lbs. estimated to have been caught but not reported[6]

Seasons: Fishing occurs year-round, but effort is concentrated in the late fall and winter and peaks during periods of low wind and sea conditions.

Gear type: This fishery is a hook-and-line fishery that takes place in deep water. In the MHI the vessels are smaller than 30 ft and trips last from 1 to 3 days.

Regulations: In the MHI, the sale of snappers (opakapaka, onaga and uku) and jacks less than one pound is prohibited. In June of 1998, Hawaii Division of Aquatic Resources (HDAR) closed 19 areas to bottomfishing, and regulations pertaining to seven species (onaga, opakapaka, ehu, kalekale, gindai, hapuupuu and lehi) were enacted. Total Allowable Catch (TAC) limits have been established for the "Deep-7" bottomfish species; these are the 7 primary species targeted by the commercial fleet. The TAC applies to both commercial and non-commercial sectors of the fishery. To ensure the TAC is not exceeded, NMFS and the State of Hawaii monitor the catch of Deep-7 bottomfish during the annual fishing season. Annual TAC quota for Hawaii Restricted Bottomfish Species specified in Federal Register by August 31st each year.

Management type: The portion of the fishery in Federal waters is managed under the Fishery Ecosystem Plan for the Hawaiian Archipelago, and operates under an annual catch limit. The fishery is co-managed with the State of Hawaii, which has adopted complementary measures in State waters.

Comments: The deep-slope bottomfish fishery in Hawaii concentrates on species of eteline snappers, carangids, and a single species of grouper concentrated at depths of 30-150 fathoms. These fish have been fished on a subsistence basis since ancient times and commercially for at least 90 years. Effort in this fishery increases significantly around the Christmas season because a target species, a true snapper, is typically sought for cultural festivities.

**Category III, Hawaii inshore handline fishery**

Number of active permit holders: In 2011 there were 378 active commercial fishers

Total effort: In 2011 there were 4,577 inshore handline trips.

Seasons: unknown.

Gear type: Fishing from a vessel using a vertical mainline with single/multiple lures or baited hooks and weight, lowered near the bottom to include drifting for octopus (tako) while using a handline. Fishing tackle usually consists of lighter gear than deep-sea handline. Line can be retrieved manually or by any other powered method. This fishery occurs in nearshore and coastal pelagic regions.

Regulations: Managed under State of Hawaii regulations.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Pacific Pelagics Fishery Ecosystem Plan contains no management measures applicable to this gear.

Comments: The principal catches include reef fishes and big-eyed scad (akule) and mackerel scad (opelu). Bottlenose dolphins and rough-toothed dolphins have been reported as depredating bait or catch from handlines (Shallenberger 1981, Nitta and Henderson 1993). Depredation behavior may increase the risk of marine mammals becoming hooked or entangled.

**Category III, Hawaii tuna handline-fishery**

Number of active permit holders: In 2011 there were 498 active commercial fishers.

Total effort: In 2011 there were 4,619 trips classified as one of the three tuna handline methods, 74 hybrid, 1,626 ika-shibi, and 2,919 palu-ahi.

Seasons: unknown.

Gear type: Palu-ahi tuna handline fishing usually takes place during the daytime. Sometimes instead of using lead weights, the baited hook and cut pieces of bait ("chum") are laid on a stone and the leader is wrapped around the stone and secured with a slipknot. The line wrapped stone is then lowered to the desired depth, where a tug on the line releases the slipknot, dispersing the chum and releasing the baited hook. The stone falls to the bottom, leaving the line free to be worked by the fisherman. This method also includes the use of "danglers" for reporting purposes. Iki-shibi tuna handline fishing occurs mainly at night also using a vertical mainline with high-test monofilament leader, from which is suspended a single baited hook. A weight may be used between the mainline and leader, with four or more lines usually attached to the vessel by breakaway links. A sea anchor is used to control and slow (at times stop) the drift of the vessel. A small light is usually suspended from the boat to attract muhe'e ("true squid") or opelu, typically used as bait. Line may be hauled manually, mechanically or by any powered method. Hybrid tuna handline fishing is a unique mixture of fishing methods used to catch pelagic species primarily on offshore seamounts and near NOAA weather buoys. It is generally a combination of methods which could include handlining, trolling, baiting techniques and other methods which are used simultaneously.

Regulations: Managed under State of Hawaii regulations.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

Comments: This fishery occurs around offshore fish aggregating devices and mid-ocean seamounts and pinnacles. The principal catches are small to medium sized bigeye, yellowfin and albacore tuna. There are several types of handline methods in the Hawaiian fisheries. Baited lines with chum are used in day fishing operations (palu-ahi), another version uses squid as bait during night operations (ika-shibi), and an operation called "danglers" uses multiple lines with artificial lures suspended or dangled over the water. Bottlenose dolphins and rough-toothed dolphins have been reported as depredating bait or catch from handlines (Shallenberger 1981, Nitta and Henderson 1993). Depredation behavior may increase the risk of marine mammals becoming hooked or entangled.

**Category III, Hawaii spearfishing fishery**

Number of active permit holders: In 2011 there were 143 active commercial fishers

Total effort: In 2011 there were 2,142 spearfishing trips.

Seasons: unknown.

Gear type: Fishing with a shaft with one or more sharpened points at one end usually associated with diving. Includes bow and torch fishing.

Regulations: Managed under State of Hawaii regulations.

Management type: A commercial marine license issued by the Hawaii Department of Land and Natural Resources, Division of Aquatic Resources (DAR) is required for all commercial fishing activities in Hawaii State waters. The Fishery Ecosystem Plan for the Hawaii Archipelago contains no management measures applicable to this gear.

Comments: Interactions have been documented with Hawaiian monk seals.

**References:**

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Appendix 2. Pacific reports revised in 2023 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

| Species (Stock) | $N_{est}$ | CV $N_{est}$ | $N_{min}$ | $R_{max}$ | Fr | PBR | Annual Human-Caused Mortality and Serious Injury | Annual Commercial Fishery Mortality and Serious Injury | Strategic Status | Most Recent Abundance Survey | | | Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| California sea lion (U.S.) | 257,606 | n/a | 233,515 | 0.12 | 1 | 14,011 | ≥321 | ≥197 | N | 2008 | 2013 | 2014 | 2018 |
| Harbor Seal (California) | 30,968 | n/a | 27,348 | 0.12 | 1 | 1,641 | 43 | 30 | N | 2004 | 2009 | 2012 | 2014 |
| Harbor Seal (Oregon/Washington Coast) | unk | unk | unk | 0.12 | 1 | undet | 10.6 | 7.4 | N | 1999 | | | 2013 |
| Harbor Seal (Washington Northern Inland Waters) | unk | unk | unk | 0.12 | 1 | undet | 9.8 | 2.8 | N | 1999 | | | 2013 |
| Harbor Seal (Southern Puget Sound) | unk | unk | unk | 0.12 | 1 | undet | 3.4 | 1 | N | 1999 | | | 2013 |
| Harbor Seal (Hood Canal) | unk | unk | unk | 0.12 | 1 | undet | 0.2 | 0.2 | N | 1999 | | | 2013 |
| Northern Elephant Seal (California Breeding) | 187,386 | n/a | 85,369 | 0.12 | 1 | 5,122 | 13.7 | 5.3 | N | 2005 | 2010 | 2013 | 2021 |
| Guadalupe Fur Seal (Mexico) | 34,187 | n/a | 31,019 | 0.137 | 0.5 | 1,062 | ≥3.8 | ≥1.2 | S | 2008 | 2009 | 2013 | 2019 |
| Northern Fur Seal (California)(California) | 14,050 | n/a | 7,524 | 0.12 | 1 | 451 | 1.8 | ≥0.8 | N | 2010 | 2011 | 2013 | 2015 |
| Monk Seal (Hawai'i) | 1,564 | 0.05 | 1,444 | 0.07 | 0.1 | 5.1 | ≥5.4 | ≥2.6 | S | 2019 | 2020 | 2021 | 2023 |
| Harbor Porpoise (Morro Bay) | 4,191 | 0.56 | 2,698 | 0.096 | 0.5 | 65 | 0 | 0 | N | 2008 | 2011 | 2012 | 2021 |
| Harbor Porpoise (Monterey Bay) | 3,760 | 0.561 | 2,421 | 0.058 | 0.5 | 35 | ≥0.2 | ≥0.2 | N | 2011 | 2012 | 2013 | 2021 |
| Harbor Porpoise (San Francisco - Russian River) | 7,777 | 0.62 | 4,811 | 0.061 | 0.5 | 73 | ≥0.4 | ≥0.4 | N | 2014 | 2016 | 2017 | 2021 |
| Harbor Porpoise (Northern CA/Southern OR) | 15,303 | 0.575 | 9,759 | 0.04 | 1 | 195 | unk | unk | N | 2016 | 2021 | 2022 | 2023 |
| Harbor Porpoise (Central Oregon) | 7,492 | 0.421 | 5,332 | 0.04 | 0.5 | 53 | unk | unk | N | 2016 | 2021 | 2022 | 2023 |
| Harbor Porpoise (Northern OR/Washington Coast) | 22,074 | 0.391 | 16,068 | 0.04 | 0.5 | 161 | ≥3.2 | ≥2.8 | N | 2016 | 2021 | 2022 | 2023 |
| Harbor Porpoise (Washington Inland Waters) | 11,233 | 0.37 | 8,308 | 0.04 | 0.4 | 66 | ≥7.2 | ≥7.2 | N | 2013 | 2014 | 2015 | 2016 |
| Dall's Porpoise (California/Oregon/Washington) | 16,498 | 0.61 | 10,286 | 0.04 | 0.48 | 99 | ≥0.66 | ≥0.66 | N | 2008 | 2014 | 2018 | 2021 |
| Pacific white-sided Dolphin (California/Oregon/Washington) | 34,999 | 0.222 | 29,090 | 0.04 | 0.48 | 279 | 7 | 4 | N | 2008 | 2014 | 2018 | 2021 |

Appendix 2. Pacific reports revised in 2023 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

| Species (Stock) | $N_{est}$ | CV $N_{est}$ | $N_{min}$ | $R_{max}$ | Fr | PBR | Annual Human-Caused Mortality and Serious Injury | Annual Commercial Fishery Mortality and Serious Injury | Strategic Status | Most Recent Abundance Survey | | | Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Risso's Dolphin (California/Oregon/Washington) | 6,336 | 0.32 | 4,817 | 0.04 | 0.48 | 46 | ≥3.7 | ≥3.7 | N | 2005 | 2008 | 2014 | 2016 |
| Common Bottlenose Dolphin (California Coastal) | 453 | 0.06 | 346 | 0.04 | 0.48 | 2.7 | ≥2.0 | ≥1.6 | N | 2009 | 2010 | 2011 | 2016 |
| Common Bottlenose Dolphin (California/Oregon/Washington Offshore) | 3,477 | 0.696 | 2,048 | 0.04 | 0.48 | 19.7 | ≥0.82 | ≥0.82 | N | 2008 | 2014 | 2018 | 2021 |
| Striped Dolphin (California/Oregon/Washington) | 29,988 | 0.3 | 23,448 | 0.04 | 0.48 | 225 | ≥4.0 | ≥4.0 | N | 2008 | 2014 | 2018 | 2021 |
| Common Dolphin, short-Beaked (California/Oregon/Washington) | 1,056,308 | 0.21 | 888,971 | 0.04 | 0.5 | 8,889 | ≥30.5 | ≥30.5 | N | 2008 | 2014 | 2018 | 2021 |
| Common Dolphin, long-Beaked (California) | 83,379 | 0.216 | 69,636 | 0.04 | 0.48 | 668 | ≥29.7 | ≥26.5 | N | 2008 | 2014 | 2018 | 2021 |
| Northern right Whale Dolphin (California/Oregon/Washington) | 29,285 | 0.72 | 17,024 | 0.04 | 0.48 | 163 | ≥6.6 | ≥6.6 | N | 2008 | 2014 | 2018 | 2021 |
| Killer Whale (Eastern N Pacific Offshore) | 300 | 0.1 | 276 | 0.04 | 0.5 | 2.8 | unk | unk | N | 2010 | 2011 | 2012 | 2018 |
| Killer Whale (Eastern N Pacific Southern Resident) | 73 | n/a | 73 | 0.035 | 0.1 | 0.13 | unk | unk | S | 2020 | 2021 | 2022 | 2023 |
| Short-finned pilot Whale (California/Oregon/Washington) | 836 | 0.79 | 466 | 0.04 | 0.48 | 4.5 | 1.2 | 1.2 | N | 2005 | 2008 | 2014 | 2016 |
| Baird's Beaked Whale (California/Oregon/Washington) | 1,363 | 0.53 | 894 | 0.04 | 0.5 | 8.9 | ≥0.2 | unk | N | 2008 | 2014 | 2018 | 2021 |
| Mesoplodont Beaked whales (California/Oregon/Washington) | 3,044 | 0.54 | 1,967 | 0.04 | 0.5 | 20 | 0.1 | 0.1 | N | 2005 | 2008 | 2014 | 2017 |
| Cuvier's Beaked Whale (California/Oregon/Washington) | 5,454 | 0.27 | 4,214 | 0.04 | 0.5 | 42 | <0.1 | <0.1 | N | 2008 | 2014 | 2016 | 2022 |
| Pygmy Sperm Whale (California/Oregon/Washington) | 4,111 | 1.12 | 1,924 | 0.04 | 0.5 | 19.2 | unk | unk | N | 2005 | 2008 | 2014 | 2016 |
| Dwarf Sperm Whale (California/Oregon/Washington) | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | 2005 | 2008 | 2014 | 2016 |

Appendix 2. Pacific reports revised in 2023 are highlighted. S=strategic stock, N=non-strategic stock. unk=unknown, undet=undetermined, n/a=not applicable.

| Species (Stock) | $N_{est}$ | CV $N_{est}$ | $N_{min}$ | $R_{max}$ | Fr | PBR | Annual Human-Caused Mortality and Serious Injury | Annual Commercial Fishery Mortality and Serious Injury | Strategic Status | Most Recent Abundance Survey | | | Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sperm Whale (California/Oregon/Washington) | 2,606 | 0.135 | 2,011 | 0.04 | 0.1 | 4.0 | 0.52 | 0.52 | S | 2008 | 2014 | 2018 | 2023 |
| Gray Whale (Eastern N Pacific) | 26,960 | 0.05 | 25,849 | 0.062 | 1 | 801 | 131 | 9.3 | N | 2011 | 2015 | 2016 | 2020 |
| Gray Whale (Western N Pacific) | 290 | n/a | 271 | 0.062 | 0.1 | 0.12 | unk | unk | S | 2014 | 2015 | 2016 | 2020 |
| Humpback Whale (Central America / Southern Mexico - California-Oregon-Washington) | 1,496 | 0.171 | 1,284 | 0.082 | 0.1 | 3.5 | 14.9 | 8.1 | S | 2019 | 2020 | 2021 | 2022 |
| Humpback Whale (Mainland Mexico - California-Oregon-Washington) | 3,477 | 0.101 | 3,185 | 0.082 | 0.5 | 43 | 22 | 11.4 | S | 2016 | 2017 | 2018 | 2022 |
| Blue Whale (Eastern N Pacific) | 1,898 | 0.085 | 1,767 | 0.04 | 0.2 | 4.1 | ≥18.6 | ≥0.61 | S | 2016 | 2017 | 2018 | 2023 |
| Fin Whale (California/Oregon/Washington) | 11,065 | 0.405 | 7,970 | 0.04 | 0.5 | 80 | ≥43.4 | ≥0.41 | S | 2008 | 2014 | 2018 | 2023 |
| Sei Whale (Eastern N Pacific) | 864 | 0.40 | 625 | 0.04 | 0.1 | 1.25 | unk | unk | S | 2005 | 2008 | 2014 | 2023 |
| Minke Whale (California/Oregon/Washington) | 915 | 0.792 | 509 | 0.04 | 0.4 | 4.1 | ≥0.19 | ≥0.17 | N | 2008 | 2014 | 2018 | 2023 |
| Bryde's Whale (Eastern Tropical Pacific) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | n/a | n/a | n/a | 2015 |
| Rough-toothed Dolphin (Hawai'i) | 83,915 | 0.49 | 56,782 | 0.04 | 0.45 | 511 | 3.2 | 3.2 | N | 2002 | 2010 | 2017 | 2023 |
| Rough-toothed Dolphin (American Samoa) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | unk | n/a | n/a | n/a | 2010 |
| Risso's Dolphin (Hawai'i) | 6,979 | 0.29 | 5,283 | 0.04 | 0.5 | 53 | 0 | 0 | N | 2010 | 2017 | 2020 | 2023 |
| Common Bottlenose Dolphin (Hawai'i Pelagic) | 24,669 | 0.57 | 15,783 | 0.04 | 0.5 | 158 | 0 | 0 | N | 2010 | 2017 | 2020 | 2023 |
| Common Bottlenose Dolphin (Kaua'i and Ni'ihau) | 112 | 0.24 | 92 | 0.04 | 0.5 | 0.9 | unk | unk | N | 2016 | 2017 | 2018 | 2023 |
| Common Bottlenose Dolphin (O'ahu) | 112 | 0.17 | 97 | 0.04 | 0.5 | 1.0 | unk | unk | N | 2016 | 2017 | 2017 | 2023 |
| Common Bottlenose Dolphin (Maui Nui) | 64 | 0.15 | 56 | 0.04 | 0.5 | 0.6 | unk | unk | N | 2016 | 2017 | 2018 | 2023 |

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| Species (Stock) | $N_{est}$ | CV $N_{est}$ | $N_{min}$ | $R_{max}$ | Fr | PBR | Annual Human-Caused Mortality and Serious Injury | Annual Commercial Fishery Mortality and Serious Injury | Strategic Status | Most Recent Abundance Survey | | | Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Common Bottlenose Dolphin(Hawai‘i Island) | 136 | 0.43 | 96 | 0.04 | 0.5 | 1.0 | ≥ 0.2 | unk | N | 2016 | 2017 | 2018 | 2023 |
| Pantropical Spotted Dolphin(Hawai‘i Pelagic) | 67,313 | 0.27 | 53,839 | 0.04 | 0.5 | 538 | 0 | 0 | N | 2010 | 2017 | 2020 | 2023 |
| Pantropical Spotted Dolphin (O‘ahu) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2023 |
| Pantropical Spotted Dolphin (Maui Nui) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2023 |
| Pantropical Spotted Dolphin (Hawai‘i Island) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2023 |
| Spinner Dolphin (Hawai‘i Island) | 665 | 0.09 | 617 | 0.04 | 0.5 | 6.2 | ≥ 1.0 | unk | N | 2010 | 2011 | 2012 | 2018 |
| Spinner Dolphin (O’ahu / 4 Islands Region) | n/a | n/a | n/a | 0.04 | 0.5 | undet | ≥ 0.4 | unk | N | 1998 | 2002 | 2007 | 2018 |
| Spinner Dolphin (Kaua’i and Ni’ihau) | n/a | n/a | n/a | 0.04 | 0.5 | undet | unk | unk | N | 1995 | 1998 | 2005 | 2018 |
| Spinner Dolphin (Hawai‘i Pelagic) | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | | 2002 | 2010 | 2018 |
| Spinner Dolphin (Kure / Midway) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | 1998 | 2010 | 2018 |
| Spinner Dolphin (Pearl and Hermes Reef) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | N | | | n/a | 2018 |
| Spinner Dolphin (American Samoa) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | unk | | | n/a | 2010 |
| Striped Dolphin (Hawai‘i) | 64,343 | 0.28 | 51,055 | 0.04 | 0.5 | 511 | 0 | 0 | N | 2010 | 2017 | 2020 | 2023 |
| Fraser’s Dolphin (Hawai‘i) | 40,960 | 0.7 | 24,068 | 0.04 | 0.5 | 241 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Melon-headed Whale (Hawaiian Islands) | 40,647 | 0.74 | 23,301 | 0.04 | 0.5 | 233 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |
| Melon-headed Whale (Kohala Resident) | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | | | 2009 | 2020 |
| Pygmy killer Whale (Hawai‘i) | 10,328 | 0.75 | 5,885 | 0.04 | 0.5 | 59 | 0 | 0 | N | 2002 | 2010 | 2017 | 2020 |

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| Species (Stock) | $N_{est}$ | CV $N_{est}$ | $N_{min}$ | $R_{max}$ | Fr | PBR | Annual Human-Caused Mortality and Serious Injury | Annual Commercial Fishery Mortality and Serious Injury | Strategic Status | Most Recent Abundance Survey | | | | Revised |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| False killer Whale (NW Hawaiian Islands) | 477 | 1.71 | 178 | 0.04 | 0.4 | 1.43 | 0.16 | 0.16 | N | 2002 | 2010 | 2017 | | 2023 |
| False killer Whale (Hawai‘i Pelagic) | 5,528 | 0.35 | 4,152 | 0.04 | 0.44 | 36 | 47 | 47 | S | 2002 | 2010 | 2017 | | 2023 |
| False killer Whale (Main Hawaiian Islands Insular) | 167 | 0.14 | 149 | 0.04 | 0.1 | 0.3 | 0.1 | 0.1 | S | 2013 | 2014 | 2015 | | 2023 |
| False killer Whale (Palmyra Atoll) | 1,329 | 0.65 | 806 | 0.04 | 0.4 | 6.4 | 0.3 | 0.3 | N | | | 2005 | | 2012 |
| False killer Whale (American Samoa) | unk | unk | unk | 0.04 | 0.5 | undet | unk | unk | unk | | | 2006 | | 2010 |
| Killer Whale (Hawai‘i) | 161 | 1.06 | 78 | 0.04 | 0.5 | 0.8 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Pilot Whale, short-finned (Hawai‘i) | 19,242 | 0.23 | 15,894 | 0.04 | 0.5 | 159 | 0.2 | 0 | N | 2010 | 2017 | 2020 | | 2023 |
| Blainville’s Beaked Whale (Hawai‘i Pelagic) | 1,132 | 0.99 | 564 | 0.04 | 0.5 | 5.6 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Longman's Beaked Whale (Hawai‘i) | 2,550 | 0.67 | 1,527 | 0.04 | 0.5 | 15 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Cuvier’s Beaked Whale (Hawai‘i Pelagic) | 4,431 | 0.41 | 3,180 | 0.04 | 0.5 | 32 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Pygmy Sperm Whale (Hawai‘i) | 42,083 | 0.64 | 25,695 | 0.04 | 0.5 | 257 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Dwarf Sperm Whale (Hawai‘i) | unk | unk | unk | 0.04 | 0.5 | undet | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Sperm Whale (Hawai‘i) | 5,707 | 0.23 | 4,486 | 0.04 | 0.2 | 18 | 0 | 0 | S | 2002 | 2010 | 2017 | | 2020 |
| Blue Whale (Central N Pacific) | 133 | 1.09 | 63 | 0.04 | 0.1 | 0.1 | 0 | 0 | S | | 2002 | 2010 | | 2017 |
| Fin Whale (Hawai‘i) | 203 | 0.99 | 101 | 0.04 | 0.1 | 0.2 | 0 | 0 | S | 2002 | 2010 | 2017 | | 2020 |
| Bryde’s Whale (Hawai‘i) | 791 | 0.29 | 623 | 0.04 | 0.5 | 6.2 | 0 | 0 | N | 2010 | 2017 | 2020 | | 2023 |
| Sei Whale (Hawai‘i) | 391 | 0.9 | 204 | 0.04 | 0.1 | 0.4 | 0.2 | 0.2 | S | n/a | 2002 | 2010 | | 2017 |
| Minke Whale (Hawai‘i) | 438 | 1.05 | 212 | 0.04 | 0.5 | 2.1 | 0 | 0 | N | 2002 | 2010 | 2017 | | 2020 |
| Humpback Whale (American Samoa) | unk | unk | 150 | 0.106 | 0.1 | 0.4 | unk | unk | S | 2006 | 2007 | 2008 | | 2009 |